UNIVERSIDAD COMPLUTENSE DE MADRID

FACULTAD DE CIENCIAS GEOLÓGICAS

DEPARTAMENTO DE GEODINÁMICA, ESTRATIGRAFÍA Y PALEONTOLOGÍA

TESIS DOCTORAL

Análisis de las interacciones entre cantidad y calidad en la consecución del buen estado de las aguas continentales del Tajo Medio según la Directiva Marco del Agua

MEMORIA PARA OPTAR AL GRADO DE DOCTOR

PRESENTADA POR

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Madrid, 2020

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Memoria para optar al grado de doctor por:

Antonio Bolinches Quero

Bajo la dirección de los doctores:

Dra. Lucia De Stefano – Dr. Javier Paredes Arquiola

Madrid, 2020

A Laia

Sense tu la tesi s'hauria acabat amb un poc menys de temps, i amb molta menys felicitat

ὅθεν ὕστερον ἐρωτηθεὶς εἰ τοὺς ἀρίστους Ἀθηναίοις νόμους ἔγραψεν,
"ὧν ἄν," ἔφη, "προσεδέξαντο τοὺς ἀρίστους."

Πλούταρχος, Βίοι Παράλληλοι, Σόλων

Cuando le preguntaron si había dado a los atenienses las mejores leyes,
respondió: "de las que estaban dispuestos a aceptar, las mejores"

Plutarco, Vidas Paralelas, Solón

Dosis sola facit venenum

Paracelsus

Agradecimientos

Este estudio no se habría realizado sin el apoyo, tanto profesional como personal, de los directores. Lucia y Javier han estado siempre disponibles para ofrecer orientación científica y consejos prácticos. Su capacidad de trabajo ha sido una referencia durante toda esta etapa, y un ejemplo de lo que se puede aportar a la sociedad desde la función pública.

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En los varios años que dura un doctorado se viven momentos de alegría y ansiedad, al ritmo que se retrasan o se publican los artículos. Yo he tenido la suerte de compartirlos con Kinda y nuestra familia, y con Gabriel, Carlotta, Nuria, Camila, Beatrice y Min. Compañeros y confidentes, gracias a ellos siempre recordaré esta etapa con una sonrisa.

Quiero agradecer a todos los técnicos de la Confederación Hidrográfica del Tajo que han facilitado este estudio con la aportación de bases de datos y con su disponibilidad para consultas. Asimismo, el Ministerio para la Transición Ecológica y en especial la Dirección General del Agua y la Subdirección de Infraestructuras y Tecnología han estado siempre disponibles y han mostrado un especial interés en los resultados del estudio. Igualmente, el Ayuntamiento de Madrid, la Comunidad Autónoma de Madrid y el Canal de Isabel II han compartido sus conocimientos y han participado en los seminarios que hemos organizado a lo largo del estudio, aportando al desarrollo de las ideas principales con su experiencia. Otras aportaciones fundamentales han llegado de organizaciones como WWF España, la Fundación Nueva Cultura del Agua y la Plataforma del Tajo, que han contribuido a que el estudio tenga un enfoque más global.

Finalmente, quiero agradecer a los revisores anónimos de los artículos científicos incluidos en esta Tesis doctoral, que con una aportación desinteresada han contribuido a mejorar la calidad de las publicaciones científicas.

A todos vosotros, y a tantas otras personas que han contribuido al desarrollo de esta Tesis, gracias.

Índice

Resumen

Análisis de las interacciones entre cantidad y calidad en la consecución del buen estado de las aguas continentales del Tajo Medio según la Directiva Marco del Agua

La presente Tesis trata de contribuir a la **mejora de la calidad de las aguas del Tajo Medio aportando un mayor entendimiento sobre las relaciones entre las distintas presiones y el estado de las aguas, así como los mecanismos disponibles en la legislación para implementar las mejoras requeridas**. Todo ello desde una perspectiva integradora, que incluya tanto los aspectos tecnológicos como los legislativos y competenciales.

En una primera instancia, más que buscar respuestas se ha intentado plantear las preguntas adecuadas. A partir de un genérico ¿qué debe cambiar para conseguir una buena calidad de las aguas del área de estudio? se ha empezado por preguntar ¿cuáles son los contaminantes que incumplen los límites establecidos y qué herramientas legislativas podrían aplicarse para limitar su presencia? La Directiva Marco del Agua, junto con las Directivas de tratamiento de las aguas residuales urbanas y de contaminación de las aguas producida por nitratos de origen agrícola nos permiten identificar los **contaminantes físico-químicos (materia orgánica con demanda de oxígeno, compuestos de nitrógeno y fósforo) como objeto de estudio**. Estos contaminantes presentan el mayor número de incumplimientos, y la legislación prevé mecanismos claros para actuar sobre las presiones que los causan. En la Tesis se identifican los artículos de las Directivas europeas y su transposición a la legislación española a disposición de las autoridades para mejorar el estado de las aguas receptoras, destacando **la autorización de vertido** como la herramienta clave en nuestra zona de estudio.

Pero, ¿cómo identificar las presiones detrás de los incumplimientos y cuantificar su efecto? Para ello se han desarrollado modelos matemáticos de evolución de contaminantes. Se han censado las presiones existentes, calibrado los modelos de modo que expliquen satisfactoriamente las variables observadas e identificado las relaciones de causa y efecto entre cada una de las presiones y el estado de las aguas receptoras. Los modelos muestran que **los vertidos de las grandes depuradoras urbanas de la región de Madrid son los principales causantes de los incumplimientos, con un impacto sensiblemente mayor a otras presiones como la contaminación difusa agraria o las detracciones de agua por el Trasvase Tajo-Segura**. Seguidamente, se han generado escenarios de cambio que cuantifican el nivel de presiones compatible con el buen estado de las aguas receptoras. De este modo, la presente Tesis identifica los valores de concentraciones de contaminantes en efluentes de depuradora que permiten cumplir los objetivos ambientales.

Se constata que las medidas necesarias para alcanzar el buen estado requieren un importante esfuerzo económico por parte de las administraciones, en forma de modernización de la infraestructura de depuración existente. En algunos casos, el buen estado de un pequeño tramo de río solo es alcanzable mediante grandes inversiones.

Un análisis realista del problema no puede ignorar estos aspectos, lo cual nos conduce a la siguiente pregunta: **¿existen mecanismos en la legislación para priorizar las medidas de mitigación según los esfuerzos requeridos y el beneficio obtenido?** Un análisis de la Directiva Marco del Agua y del Reglamento de Planificación Hidrológica nos permiten identificar las **exenciones a los objetivos medioambientales por costes desproporcionados** como una figura legislativa fundamental para racionalizar el uso de las medidas y garantizar un nivel de exigencia comparable entre las demarcaciones de la Unión Europea. Al no encontrar en la literatura científica un método comúnmente aceptado a nivel europeo para cuantificar la posible desproporcionalidad de los costes implicados, en la presente Tesis se desarrolla una **metodología que compara el coste de las medidas de mitigación identificadas por los modelos con el beneficio medioambiental obtenido y la capacidad de pago** de la región responsable de las presiones, mediante el cálculo de un índice de desproporcionalidad. El método, aplicado al Tajo Medio, permite asignar un valor cuantitativo de desproporcionalidad de coste a cada escenario de mejora.

Una aplicación equitativa requeriría que el umbral de desproporcionalidad para la aplicación de exenciones sea comparable en toda la Unión Europea, lo cual no está garantizado en la actualidad. Para contribuir a la definición de este umbral, en la presente Tesis se **compara el nivel de presiones a partir del cual se declaran exenciones en el Tajo Medio con otras cinco demarcaciones europeas** (de Portugal, Irlanda, Italia, República Checa y Estonia). Los resultados nos muestran que, aunque el nivel es comparable en la mayoría de las demarcaciones estudiadas, las más septentrionales (en Irlanda y Estonia) apelan a exenciones con niveles de presión sensiblemente más bajos, poniendo en evidencia la **necesidad de consensuar un umbral de desproporcionalidad a nivel europeo**.

A lo largo de la Tesis se ha dado una especial relevancia a la **transferencia de conocimiento** a los actores implicados y la sociedad en general. Para ello se han organizado seminarios y reuniones de seguimiento en las que se ha compartido el avance del estudio con las administraciones (Confederación Hidrográfica del Tajo, Ministerio para la Transición Ecológica, Comunidad Autónoma y Ayuntamiento de Madrid, Canal de Isabel Segunda, etc.) y las empresas del sector. Asimismo, se han organizado jornadas y publicado artículos de divulgación para poner los resultados a disposición del público interesado.

La Tesis se presenta en formato publicaciones, con cuatro artículos publicados en revistas internacionales indexadas.

Abstract

Analysis of the interactions between quantity and quality for the achievement of the good status of the inland surface waters of the Middle Tagus, according to the Water Framework Directive

This PhD Dissertation aims at contributing to the **improvement of the water quality of the Middle Tagus river by exploring the relationship between the different pressures and the status of the receiving waters, as well as the mechanisms available in the legislation to implement the required improvements**. This contribution is made through a holistic approach, which includes technological, legislative and administrative attribution aspects.

To begin with, rather than looking for answers, the focus was set to ask the right questions. From a generic "what must change to achieve good water quality in the study area?" the question derived to asking "which are the pollutants that do not comply with the established limits and what legislative tools could be applied to limit their presence?" The Water Framework Directive, together with the Urban Wastewater Treatment Directive and the Directive against Pollution caused by Nitrates from Agricultural Sources allow us to identify **physico-chemical contaminants (organic matter with oxygen demand, nitrogen and phosphorous compounds) as the object of study**. These pollutants have the highest number of non-compliances, and the legislation provides clear mechanisms to act on the pressures that cause them. The Dissertation identifies the articles of the European Directives and their transposition into the Spanish legislation at the disposal of the authorities to improve the state of the receiving waters. The wastewater plant **discharge permit** is identified as the key tool in our study area.

But how to identify the pressures behind non-compliance and quantify their effect? Mathematical models of the evolution of pollutants have been developed for this purpose. Existing pressures have been registered, models have been calibrated to explain the observed variables, and cause-effect relationships between each of the pressures and the status of the receiving waters have been identified. The models show that **discharges from large urban wastewater treatment plants in the Madrid region are the main causes of non-compliance, since they have a significantly larger impact than other pressures such as non-point pollution from agricultural sources or water detractions through the Tagus-Segura Transfer**. Change scenarios have been generated to quantify the level of pressures compatible with the good status of the receiving waters. This Dissertation identifies the values of concentration of pollutants in wastewater treatment plant effluents that are compatible with the environmental objectives.

The measures needed to achieve the good status require a significant economic effort from the administration, e.g. for the upgrade of existing wastewater treatment infrastructure. In some cases, the good status of a small river stretch is only achievable through large investments. A realistic analysis of the problem cannot ignore these

aspects, which leads us to the following question: **does the legislation offer mechanisms to prioritize mitigation measures according to the efforts required and the benefits obtained?** An analysis of the Water Framework Directive and the Spanish regulation on River Basin Management Plans identifies the **exemptions to environmental objectives due to disproportionate costs** as a fundamental legislative tool to rationalize the use of measures and guarantee a level playing ground among the European Union River Basin Authorities. There is no commonly accepted method in the scientific literature at a European level to quantify the disproportionality of the cost of a proposed measure. Therefore, **a methodology is developed in this Dissertation to compare the cost of a mitigation measure with the implied environmental benefits and the ability to pay** of the region responsible for the pressures, through the calculation of a disproportionality index. The method, applied to the Middle Tagus, assigns a quantitative value of cost disproportionality to each mitigation scenario.

A fair application of cost disproportionality exemptions would require that the threshold of disproportionality is comparable throughout the European Union. This is not currently the case. In order to contribute to the definition of this threshold, this Dissertation **compares the pressure level at which exemptions are declared in the Middle Tagus with five other European River Basins** (in Portugal, Ireland, Italy, the Czech Republic and Estonia). The results show that, although the level is comparable in four out of six River Basins under study, the water authorities in Ireland and Estonia declare exemptions with significantly lower pressure levels, highlighting the **need for an agreement at a European level on the threshold of disproportionality**.

Throughout the development of the PhD research, special relevance has been given to the **know-how transfer** to the actors involved and to society. Seminars and follow-up meetings have been organized, where the progress of the study has been shared with the public sector (Tagus River Basin Authority, Ministry for the Environment, Autonomous Region Administration and City Council of Madrid, Water Utilities Company, etc.) and local companies. Likewise, the results have been made available to the general public through public lectures and published articles in the press.

The results of the PhD work are presented in the form of publications, with four articles published in international indexed journals.

Índice de figuras

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Índice de abreviaturas

ACUAES	Aguas de las Cuencas de España
ACB	Análisis Coste-Beneficio
AEMET	Agencia Española de Meteorología
ATS	Acueducto Tajo-Segura
CAPEX	Coste de inversión inicial
CEDEX	Centro de Estudios y Experimentación de Obras Públicas
CEMAS	Control del Estado de las Masas De Agua Superficiales
CHG	Confederación Hidrográfica del Guadiana
CHT	Confederación Hidrográfica del Tajo
DBO5	Demanda Biológica de Oxígeno a cinco días
DF	Factor de Desproporcionalidad
DGA	Dirección General del Agua
DMA	Directiva Marco del Agua
DQO	Demanda Química de Oxígeno
EC	Comisión Europea
GDPsa	Producto Interior Bruto de la zona de estudio
INE	Instituto Nacional de Estadística
MAPAMA	Ministerio de Agricultura y Pesca, Alimentación y Medio ambiente
NOD	Mejora de los indicadores de calidad de aguas
OECD	Organisation for Economic Co-operation and Development
OPEX	Coste de operación y mantenimiento
PEE	Porcentaje medio de gasto medioambiental
qB	Beneficio cualitativo
QE	Peso relativo de cada tipo de indicador de calidad de aguas
RD	Real Decreto
ROEA	Red Oficial de Estaciones de Aforo
SAICA	Sistema Automático de Información de Calidad de las Aguas
SAIH	Sistema Automático de Información Hidrológica
SCRATS	Sindicato Central de Regantes del Acueducto Tajo Segura
TRLA	Texto Refundido de la Ley de Aguas
TSS	Total de Sólidos en Suspensión
UE	Unión Europea
WSPE	Factor de capacidad presupuestaria
WWF	World Wide Fund for Nature

Relación de publicaciones

Durante el desarrollo de la presente Tesis se han elaborado y enviado a publicación en revistas internacionales indexadas cuatro artículos científicos, cuyo contenido se presenta en los capítulos 5 a 8.

I. Antonio Bolinches, Lucia De Stefano, Javier Paredes-Arquiola (2020) **Adjusting wastewater treatment effluent standards to protect the receiving waters: the case of low-flow rivers in central Spain**. Environmental Earth Sciences 79, 446 (2020). DOI: 10.1007/s12665-020-09184-z

II. Antonio Bolinches, Lucia De Stefano, Javier Paredes-Arquiola (2020) **Designing river water quality policy interventions with scarce data: the case of the Middle Tagus Basin, Spain**, Hydrological Sciences Journal, 65:5, 749-762, DOI: 10.1080/02626667.2019.1708915

III. Antonio Bolinches, Lucia De Stefano, Javier Paredes-Arquiola (2020) **Too expensive to be worth it? A methodology to identify disproportionate costs of environmental measures as applied to the Middle Tagus River, Spain**, Journal of Environmental Planning and Management, DOI: 10.1080/09640568.2020.17267

IV. Antonio Bolinches, Javier Paredes-Arquiola, Alberto Garrido, Lucia De Stefano (2020) **A comparative analysis of the application of water quality exemptions in the European Union: The case of nitrogen**. Science of The Total Environment 139891. DOI: 10.1016/j.scitotenv.2020.139891

1. Introducción general

La gestión y planificación del agua se caracterizan por una gran complejidad, que abarca tanto dificultades científicas y tecnológicas como administrativas y sociales. Por ejemplo, requiere el conocimiento de diferentes fenómenos climáticos e hidrológicos, la comprensión del comportamiento de los ecosistemas acuáticos, el diseño de medidas específicas de mitigación del impacto de las presiones antrópicas o la optimización del consumo de energía en distintos ámbitos del uso del agua. Además, la gestión del agua se desarrolla dentro de un marco competencial complejo y requiere conciliar necesidades y sensibilidades en ocasiones opuestas entre los distintos actores involucrados.

Algunas de las disciplinas implicadas en la gestión y planificación del agua han alcanzado en los últimos dos siglos un nivel de madurez avanzado, como lo demuestra la capacidad tecnológica de abastecer a la población con agua potable sin riesgos para la salud. En cambio, aspectos como el nivel de vertidos antrópicos aceptable para garantizar la sostenibilidad del medio receptor, o el marco legislativo que permita un reparto equitativo de los esfuerzos requeridos para alcanzar los objetivos medioambientales, están en fase de maduración.

La aprobación en el año 2000 de la Directiva Marco del Agua, DMA (European Parliament and Council 2000) de la Unión Europea (UE) supuso un cambio de paradigma en varios aspectos de la gestión hídrica en España. El foco de la planificación hidrológica pasó de la garantía de abastecimiento de las demandas de agua a la consecución de unos objetivos medioambientales fijados siguiendo los criterios de la DMA.

Los análisis desarrollados en la presente Tesis doctoral se centran en la zona central de parte española de la demarcación hidrográfica del Tajo. Las aguas superficiales en esta zona presentan varias particularidades que hacen su estudio especialmente relevante. Por un lado, en esta región es necesario asegurar el abastecimiento de agua para uso doméstico de una población de más de 6 millones de habitantes (INE 2018). La fracción que retorna al medio natural en forma de aguas residuales contribuye a una alta concentración de sustancias contaminantes debido a la elevada carga vertida en relación al bajo caudal circulante. En el caso del eje principal del Tajo, además de recibir importantes cargas contaminantes de origen urbano, los caudales circulantes se ven disminuidos por el efecto del Trasvase Tajo-Segura, lo que afecta la capacidad de dilución del río. El efecto combinado de estas presiones sobre el estado de las aguas superficiales de la región se ha convertido en objeto de controversia a medida que se ha ido consolidando la sensibilidad social por el estado de las aguas del Tajo Medio y que ha ido avanzando el debate sobre la implementación de la DMA.

La gestión de esta zona de la demarcación del Tajo tiene un trasfondo político y social complejo (Gobierno de Castilla-La Mancha 2007; Baeza Sanz et al. 2013), afectando directamente a las Comunidades Autónomas de Madrid, Castilla la Mancha y

Extremadura, e indirectamente (a través de las transferencias de agua del Trasvase Tajo-Segura) a las Comunidades Autónomas de la Región de Murcia y Valenciana. A pesar de la constante mejora de la infraestructura de depuración de la Comunidad de Madrid (Canal de Isabel II 2020b) - llegando incluso a tratar una parte del influente al nivel de agua regenerada (Canal de Isabel II 2020a) - sigue habiendo episodios de baja calidad en las masas de agua que reciben sus efluentes. Lo cual lleva a plantear la pregunta de si dichos efluentes son la causa del mal estado o existen otras presiones que lo causan.

La investigación realizada en esta Tesis se compone de dos partes: una sobre la identificación de las medidas necesarias para alcanzar el buen estado en el Tajo Medio, y otra sobre las implicaciones financieras y legales de los costes de estas medidas.

La **primera parte de la Tesis** se centra en esclarecer la relación entre el estado de las aguas superficiales y las presiones ejercidas por las actividades humanas (contaminación puntual y difusa, abstracción de agua), la relación entre la aparente conformidad de los efluentes de depuración a la normativa específica y la no conformidad de las aguas receptoras, y los instrumentos disponibles en la legislación para alcanzar el buen estado. En esta zona, la elevada presión antrópica (Ministerio de Medio Ambiente 1999) ha conllevado una alta concentración de contaminantes en las aguas superficiales (CHT 2018d). Veinte años después de la entrada en vigor de la DMA, siguen sin alcanzarse los objetivos medioambientales en la zona de estudio. Asimismo, no se dispone de estudios de detalle que proporcionen relaciones cuantitativas entre las presiones y el estado de las aguas. El inventario de presiones y evaluación del estado de las masas de agua del presente plan hidrológico (CHT 2015) se limita a identificar la presión urbana como una fuente de impactos que comprometen el cumplimiento de los objetivos medioambientales.

La literatura previa sobre calidad de las aguas superficiales del Tajo Medio se limita a dos artículos publicados en revistas científicas. Por un lado Cubillo, Rodríguez y Barnwel (1992) presentan una visión general de la interacción entre los efluentes de depuración de aguas urbanas y la concentración de contaminantes físico-químicos en algunos cursos de agua de la Comunidad de Madrid (ríos Henares, Jarama, Manzanares y Guadarrama). Tras más de 25 años de mejora de las infraestructuras de depuración y el traslado del foco de atención del agotamiento del oxígeno disuelto a la acumulación de nutrientes, el estudio ha perdido vigencia. Por otro lado, Paredes-Arquiola et al. (2010) estudian la evolución de contaminantes en el río Manzanares desde la presa del Pardo hasta su confluencia con el río Jarama. Algunas de sus conclusiones siguen vigentes, pero el alcance geográfico del artículo, restringido al bajo Manzanares, no incluye el efecto de presiones como el de la explotación del Acueducto Tajo-Segura o la contaminación difusa que deriva de la actividad agrícola en las vegas del Henares, Jarama y Tajo.

En definitiva, la problemática ha recibido poca atención de la comunidad científica en forma de producción de artículos en revistas indexadas. Por el contrario, la calidad de las aguas del Tajo constituye un tema recurrente en los medios de comunicación generalistas y en los foros especializados de gestión de agua (El País 2017; Agroclm 2018),

No se puede abordar la gestión y planificación hidrológica del Tajo Medio sin entender el efecto de las detracciones de agua de cabecera del Acueducto Tajo-Segura (ATS). La literatura existente se ha centrado en analizar los efectos del trasvase en los recursos hídricos disponibles en la cuenca cedente y en su impacto económico (San Martín 2011; Rey, Garrido, and Calatrava 2016). Pero no hay estudios que analicen el efecto de las detracciones sobre la calidad de las aguas del Tajo. La presente Tesis pretende contribuir a cerrar esta brecha de conocimiento, aportando estimaciones cuantitativas del efecto de las detracciones del ATS sobre las concentraciones de contaminantes en el eje central del Tajo aguas abajo de la confluencia con el Jarama.

Una vez identificadas las medidas requeridas, la **segunda parte de la Tesis** se centra en el desarrollo de las herramientas normativas que comparan el coste de las medidas con el beneficio generado. Cuando dicho coste es particularmente elevado, existen en la normativa mecanismos que permiten flexibilizar los objetivos medioambientales en forma de exenciones bajo el concepto de “coste desproporcionado”. Estos mecanismos se han usado en varias masas de agua del Tajo Medio, aunque con unas justificaciones poco detalladas.

El desarrollo y aplicación de la legislación sobre exenciones a los objetivos medioambientales por costes desproporcionados ha suscitado una importante producción de artículos científicos, especialmente a nivel internacional. En ellos se trata de comparar el coste de una medida mitigadora con el beneficio producido o con la capacidad de pago de los responsables de financiar dicha medida. La principal dificultad reside en la cuantificación del beneficio obtenido en términos monetarios (la llamada monetarización de los beneficios). Los enfoques usados en la literatura existente se pueden dividir en tres ramas metodológicas (Macháč and Brabec 2018; Görlach and Pielen 2007): análisis de coste-beneficio (ACB) con beneficios monetarizados, ACB con beneficios cualitativos, y análisis de coste-capacidad de pago (affordability).

En el primer enfoque se compara el coste de las medidas con el bienestar económico que deriva de la mejora del estado de las masas de agua (Francesc Hernández-Sancho, Molinos-Senante, and Sala-Garrido 2010; Martin-Ortega et al. 2014; Jensen et al. 2013; Galioto et al. 2013). Debido a que la mayoría de los beneficios producidos no se cotizan en el mercado, es muy difícil atribuirles un valor monetario. Para remediarlo, el segundo enfoque propone comparar el coste con un beneficio cualitativo (CHG 2015; Thaler et al. 2014), medido en unidades no monetarias. Finalmente, el método de capacidad de pago (Courtecuisse 2005; OECD 2009; Laurans 2006; Ammermüller et al. 2011) propone comparar el coste anualizado de las medidas con los ingresos disponibles de los responsables de su financiación. Por ejemplo, la OECD propone que en el caso de medidas financiadas directa o indirectamente por las familias, el gasto asociado no debería superar el 5% de sus ingresos.

A pesar de la multitud de metodologías propuestas sigue sin haber un consenso a nivel de la Unión Europea para fijar un método común, lo que dificulta la comparación entre los criterios de desproporcionalidad aplicados entre las distintas demarcaciones hidrográficas de los Estados Miembros. Esta Tesis pretende contribuir a este debate proporcionando una metodología para justificar las exenciones por coste desproporcionado basada en el uso de datos ya disponibles a nivel europeo para el cálculo de beneficios. El método desarrollado ordena las medidas propuestas según un índice de desproporcionalidad común para toda la Unión Europea, de manera que se facilite la comparación entre demarcaciones y se acuerde un umbral común a partir del cual se acepten las exenciones.

Para contribuir a definir dicho umbral común, en esta Tesis se ha realizado una comparativa entre la política de exenciones en el Tajo Medio y la de otras cinco demarcaciones de la UE (situadas en Portugal, Irlanda, Italia, República Checa y Estonia), para identificar los niveles de impactos de las presiones a partir del cual se han declarado exenciones.

Finalmente, en la presente Tesis se ha dado una especial relevancia a la transferencia a la sociedad del conocimiento generado durante el desarrollo de la misma. Se ha mantenido un diálogo fluido con técnicos e investigadores de las administraciones responsables (Confederación Hidrográfica del Tajo, Ministerio de Transición Ecológica, Canal de Isabel II, Ayuntamiento y Comunidad de Madrid, Directorado General de Medio Ambiente de la Comisión Europea), con asociaciones ecologistas (WWF, Fundación Nueva Cultura del Agua, Plataforma de Toledo en Defensa el Tajo), industria del sector y sociedad civil. Los resultados de la investigación se han ido compartiendo con los actores interesados mediante seminarios, conferencias y artículos de divulgación.

En resumen, esta Tesis pretende identificar y cuantificar las presiones significativas que actúan sobre las aguas superficiales del Tajo Medio, y las medidas necesarias para alcanzar los objetivos medioambientales de la DMA. Asimismo, propone una metodología para poner el esfuerzo requerido por estas medidas en el contexto de otras demarcaciones europeas para asegurar un reparto equitativo del esfuerzo requerido a los Estados Miembros de la Unión Europea para alcanzar los objetivos medioambientales. Todo ello desde un enfoque integral que abarque tanto los aspectos técnicos como los legislativos y administrativos del problema.

La presente Tesis está estructurada en 10 capítulos. Después de esta introducción general, se presentan los objetivos principales y parciales de la investigación (capítulo 2) y el marco metodológico general utilizado para alcanzarlos (capítulo 3). A continuación, se caracteriza la zona de estudio y se presentan aspectos de la normativa vigente de especial relevancia para nuestra investigación (capítulo 4). Los capítulos 5 a 8 presentan los cuatro artículos científicos desarrollados. Seguidamente (capítulo 9) se presentan las actividades realizadas a lo largo del doctorado para transmitir los resultados a los actores implicados. Finalmente (capítulo 10), se exponen las conclusiones generales con las aportaciones de la Tesis y las propuestas para futuras líneas de investigación.

2. Objetivos de la Tesis doctoral

2.1. Objetivo principal

El objetivo principal de esta Tesis doctoral consiste en **identificar las medidas requeridas para alcanzar los objetivos de calidad del agua superficial del Tajo Medio según la Directiva Marco del Agua**. Dicho objetivo se entiende desde una perspectiva integral, donde se incluyen tanto los aspectos científico-tecnológicos que relacionan cuantitativamente las presiones ejercidas con el estado de las aguas y los cambios de infraestructuras requeridos, como los aspectos económicos e institucionales que regulan la flexibilización de los objetivos en función del esfuerzo financiero requerido para alcanzarlos. En el primer aspecto se pretende calcular los valores admisibles de las presiones, mientras que en el segundo el objetivo consiste en proponer una metodología que garantice unos criterios de racionalización del uso de recursos y de equidad en la aplicación de la DMA.

2.2. Objetivos parciales

A continuación, se enumeran los objetivos parciales que se han fijado como pasos intermedios para la consecución del objetivo principal:

- **Identificar el nivel de presiones compatible con el buen estado de las aguas superficiales del área de estudio**. Para ello se desarrollan unos modelos de calidad del agua adecuados para cada tramo de río. Esto permite estudiar el efecto de la contaminación puntual y difusa, así como del trasvase Tajo-Segura sobre la calidad del agua en el tramo medio del Tajo. En los capítulos 5 y 6 de la Tesis se trata este objetivo parcial.
- **Identificar y priorizar las medidas necesarias para conseguir el buen estado**. Se requiere determinar los cambios en la infraestructura de depuración en los ríos Manzanares, Henares y Jarama y cuantificar los volúmenes de agua que se pueden transferir a través del ATS sin comprometer los valores de los indicadores de calidad de agua. La definición de medidas también se cubre en los capítulos 5 y 6 de la Tesis.
- **Definir una metodología para determinar si los costes de las medidas son asumibles según lo establecido en la DMA**. Para ello se desarrolla un método para la justificación de la aplicación de exenciones por costes desproporcionados y se aplica a las masas de agua del Tajo Medio. El capítulo 7 trata sobre la metodología propuesta.
- **Comparar las decisiones sobre exenciones en la demarcación del Tajo con la experiencia europea en otras demarcaciones**. En el capítulo 8 de la Tesis se ofrece una comparación con otras cinco demarcaciones europeas.
- **Desarrollar un marco de trabajo entre administraciones** con competencia sobre la gestión del agua en la zona de estudio, que facilite la definición de los límites de

concentraciones físico-químicos actuales y el diseño de unas inversiones en tratamiento eficaces y consensuadas. Todo ello aportando al debate los resultados de la investigación y creando espacios de diálogo entre los actores implicados. Las acciones de transferencia del conocimiento se exponen en el capítulo 9 del estudio.

3. Metodología

Para efectuar un análisis de las interacciones entre cantidad y calidad en la consecución del buen estado de las aguas continentales del Tajo Medio según la Directiva Marco del Agua se han seguido varios pasos, que han permitido abordar la problemática de manera escalonada (Figura 1).

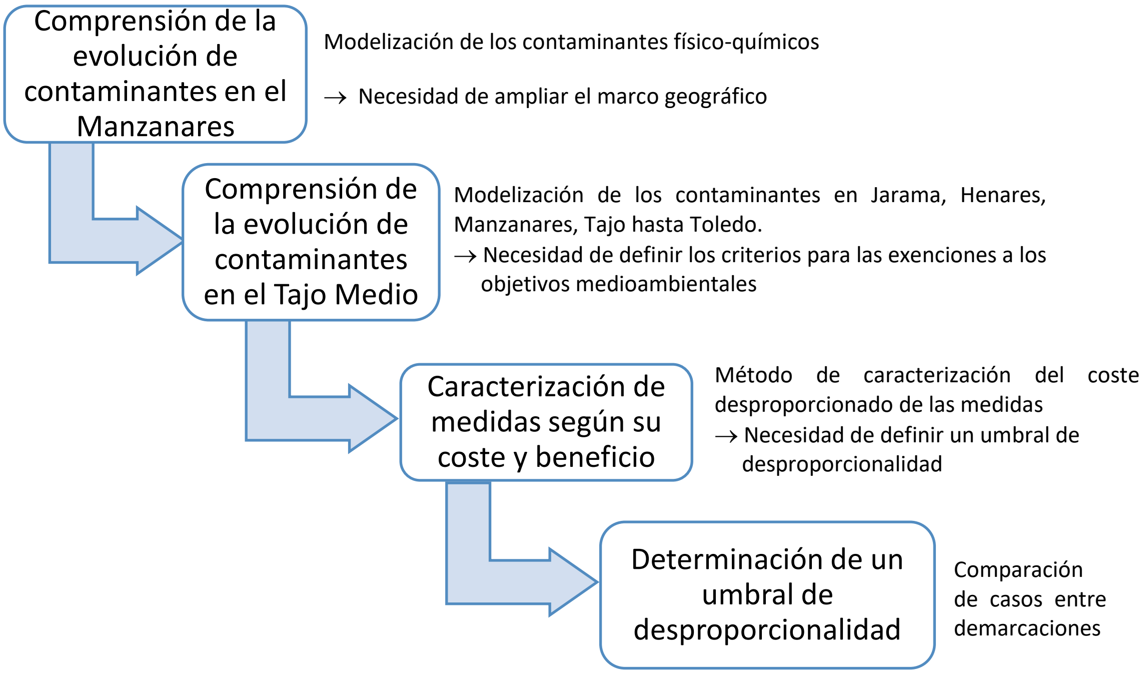

Figura 1. Secuencia de pasos del desarrollo de la Tesis

El primer paso consiste en **relacionar las presiones que se ejercen sobre el tramo medio-bajo del Manzanares con el estado de las aguas receptoras**, así como familiarizarse con las herramientas que permiten adaptarse a los datos disponibles. Se desarrolla con un modelo matemático de calidad de aguas que permite la creación de escenarios en los que se investigan los cambios requeridos en las presiones para conseguir el buen estado de las masas receptoras. Con ello se da cumplimiento (en un marco geográfico restringido) a los dos primeros objetivos parciales de esta Tesis, de identificación de las presiones y las medidas necesarias para alcanzar el buen estado.

Con las lecciones aprendidas del primer modelo, el siguiente paso consiste en la creación de un segundo modelo. El marco geográfico se amplía para incluir el Jarama aguas abajo de la confluencia del Guadalix, el Henares desde aguas arriba de Alcalá de Henares, el Manzanares desde el Pardo y el tramo del Tajo entre la confluencia con el Jarama y Toledo. Se incluye también la presión de la contaminación difusa y el efecto de las detracciones del Acueducto Tajo-Segura.

El segundo paso es, por tanto, **la creación de un modelo matemático que relacione las presiones sobre el Tajo Medio (efluentes de depuración, contaminación difusa,**

abstracciones, etc.) con el estado de las aguas receptoras. Con ello se pretende identificar las causas fundamentales de la degradación de la calidad del agua en la zona del Tajo Medio.

Dicho modelo debe adaptarse a la tipología de los datos disponibles, para lo que se desarrolla un protocolo de calibración que optimiza los coeficientes del modelo teniendo en cuenta las características estadísticas de las mediciones. Por lo tanto, se realiza una aportación a la metodología de calibración de los coeficientes de evolución de los modelos de calidad de aguas. Con ello se amplía para toda el área de interés del Tajo Medio el cumplimiento de los dos primeros objetivos parciales del estudio.

Una vez concluido el primer bloque de modelización, se constata que algunos de los objetivos medioambientales para los que se han identificado las medidas están rebajados por la aplicación de exenciones en el presente plan hidrológico . La legislación ampara esta medida, pero los mecanismos de justificación están poco desarrollados.

Con esta problemática, el tercer paso consiste en **desarrollar una herramienta que identifique en qué casos se podría justificar una flexibilización de los objetivos medioambientales de la DMA**. Para ello se enfrentan los costes de las medidas requeridas con los beneficios esperados, en términos de mejora de los indicadores que contribuyen al estado de las masas de agua. La herramienta debe facilitar la identificación de medidas con un coste desproporcionado. Este paso permite dar cumplimiento al tercer objetivo parcial de determinar si los costes de las medidas son asumibles en el contexto definido por la DMA. La herramienta se define de manera que pueda aplicarse con datos fácilmente accesibles y disponibles para todo el ámbito geográfico de aplicación de la Directiva.

Dada la inherente relatividad del límite de desproporcionalidad del coste de una medida, y la necesidad de que dicho límite sea coherente entre los Estados Miembros de la Unión Europea, el cuarto paso consiste en **hacer una comparativa de la aplicación de exenciones entre diferentes demarcaciones europeas**. Con ello se cumple el cuarto objetivo parcial de poner en perspectiva las decisiones sobre exenciones en la demarcación del Tajo, conocida la experiencia europea en otras demarcaciones. Se han estudiado los niveles de impacto de los distintos tipos de presiones (puntuales y difusas) en cada demarcación del estudio, y se ha comparado el umbral a partir del cual cada demarcación ha declarado preferentemente las exenciones.

4. Área de estudio y marco normativo

4.1. Delimitación geográfica del área de estudio

A continuación se hace una descripción de las características de la parte española de la demarcación hidrográfica del Tajo con información obtenida de la Confederación Hidrográfica del Tajo y la Agencia Española de Meteorología (AEMET 2018). La demarcación del Tajo en su parte española abarca 55 781 km^2, y está delimitada por el Sistema Central al norte, el Sistema Ibérico al este y los Montes de Toledo al sur. Se caracteriza por ser la demarcación más poblada de España. De sus 7,8 millones de habitantes (INE 2018), 6,5 millones están situados en la Comunidad Autónoma de Madrid, lo que implica una importante concentración de presiones puntuales urbanas sobre el sistema de drenaje superficial.

El clima está caracterizado como "Mediterráneo típico de verano cálido" (Csa) en el sistema de clasificación Köppen–Geiger (Kottek et al. 2006). La pluviometría está sujeta a una gran variabilidad temporal intra-anual y su distribución geográfica está notablemente influenciada por la orografía, con precipitaciones que superan los 1800 mm anuales en las cumbres del Sistema Central (Figura 2), que alimenta los afluentes de la margen derecha del Tajo. La cabecera del río Tajo a su vez recibe las precipitaciones del Sistema Ibérico (del orden de 1000 mm anuales), mientras que la margen izquierda comprende zonas con una precipitación media por debajo de los 400 mm anuales. La ciudad de Madrid, con una precipitación media de 421 mm anuales, presenta unos veranos muy secos (10 mm mensuales de precipitación) e inviernos relativamente húmedos (hasta 60 mm mensuales). La temperatura atmosférica media mensual varía de 2.7 grados centígrados en invierno a los 32.1 grados en verano.

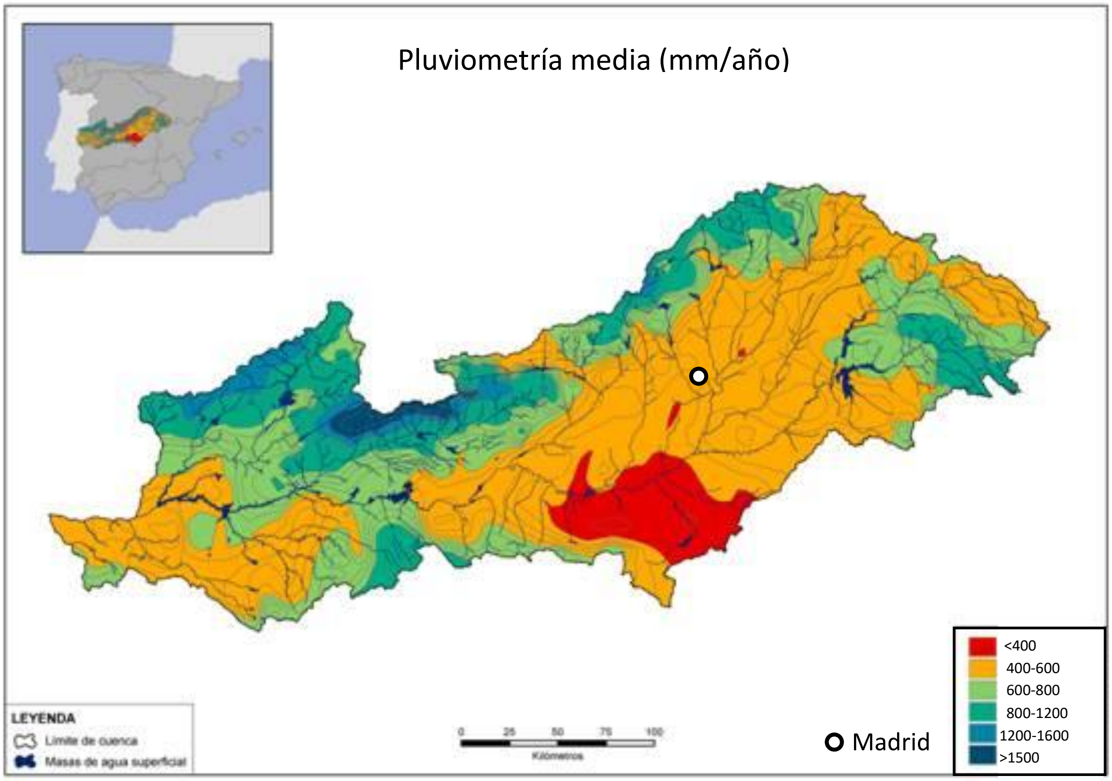

Figura 2. Pluviometría de la cuenca del Tajo. Modificado a partir de la página de Rasgos Climáticos de la Confederación Hidrográfica del Tajo (CHT 2020b)

La geología de la cuenca juega un papel importante a la hora de transformar la precipitación en escorrentía superficial. Las rocas carbonáticas del Sistema Ibérico en la cabecera del Tajo (Figura 3) tienen cierta capacidad de almacenamiento y laminación de los picos de precipitación, lo que resulta en que las aportaciones en el eje principal del Tajo aguas arriba de la confluencia del Jarama sean bastante uniformes a lo largo del año. Por el contrario, las rocas ígneas y metamórficas del Sistema Central en las cabeceras de Jarama y Manzanares responden a la precipitación con picos de escorrentía más pronunciados. Estos efectos se ven amortiguados por las importantes obras de regulación existentes en el cauce alto de estos ríos: presas de Entrepeñas y Buendía en la cabecera del Tajo, Pinilla, Riosequillo, Puentes Viejas, el Villar y el Atazar en el Lozoya (afluente del Jarama), así como el Vado en el propio Jarama, Navacerrada, Manzanares el Real y el Pardo en el Manzanares. De manera que los caudales circulantes dependen no solo de las precipitaciones sino también de las decisiones de explotación de las presas, que toman en consideración aspectos como la garantía de suministro de agua y la producción hidroeléctrica.

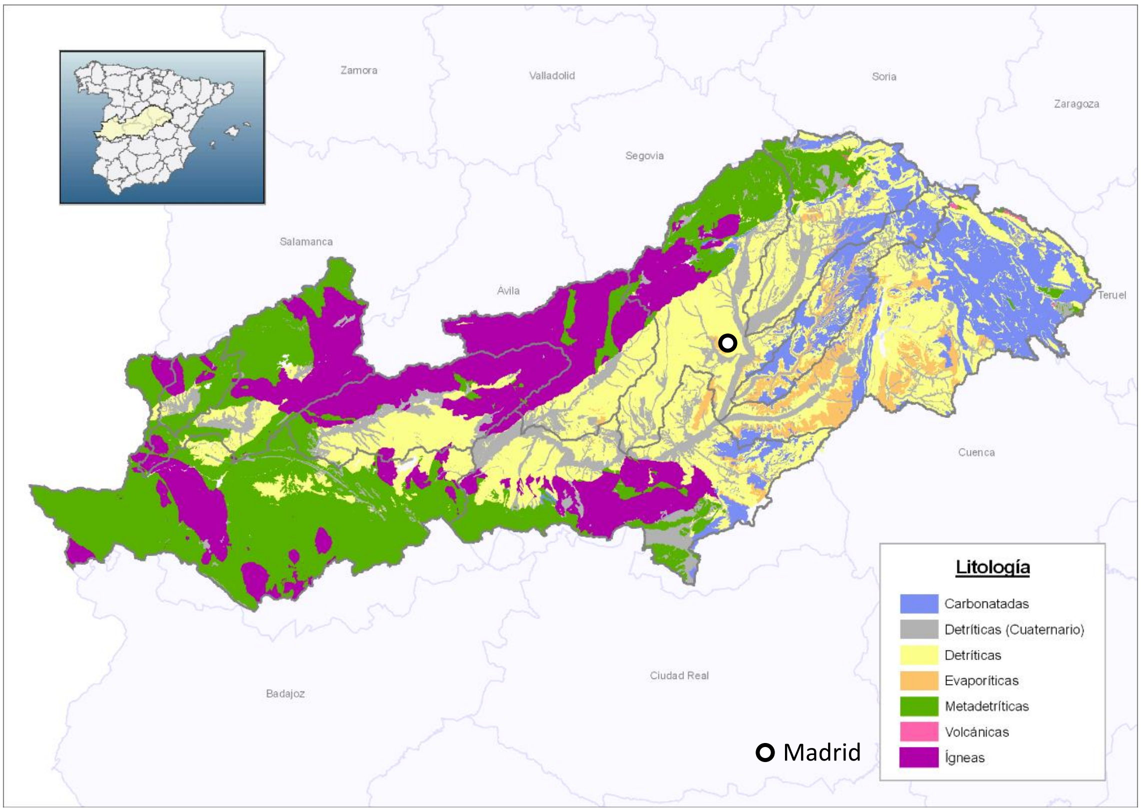

Figura 3 Principales grupos litológicos de la cuenca del Tajo. Modificado a partir de la página de Contexto Geológico de la Confederación Hidrográfica del Tajo (CHT 2020a)

La variabilidad meteorológica también afecta a la calidad de las aguas superficiales aguas abajo de las zonas urbanas. Con una carga contaminante poco variable a lo largo del año - ya que depende de la población residente -, la disminución de los caudales durante el estiaje supone un aumento considerable de la concentración de contaminantes en las aguas superficiales.

Los indicadores publicados por la Confederación Hidrográfica del Tajo relativos al estado de las masas de agua superficial nos muestran que, en general, los ríos de la demarcación conservan un estado bueno en sus cabeceras y que esta calidad tiende a deteriorarse en las cuencas bajas o al atravesar las grandes concentraciones urbanas (Figura 4). Es el caso del Jarama, Henares y Manzanares, que presentan un buen estado en sus cuencas altas y ven aumentar la concentración de contaminantes y disminuir la biodiversidad de sus hábitats (CHT 2018b) según atraviesan las grandes poblaciones de la Comunidad Autónoma de Madrid. En el caso del eje del Tajo, el buen estado se conserva hasta la confluencia con el Jarama en Aranjuez. A partir del embalse de Castrejón, aguas abajo de Toledo, ya no hay grandes aportaciones de contaminantes a las masas de agua superficiales y las presiones son más bien sobre la hidromorfología del río, con grandes presas de producción hidroeléctrica.

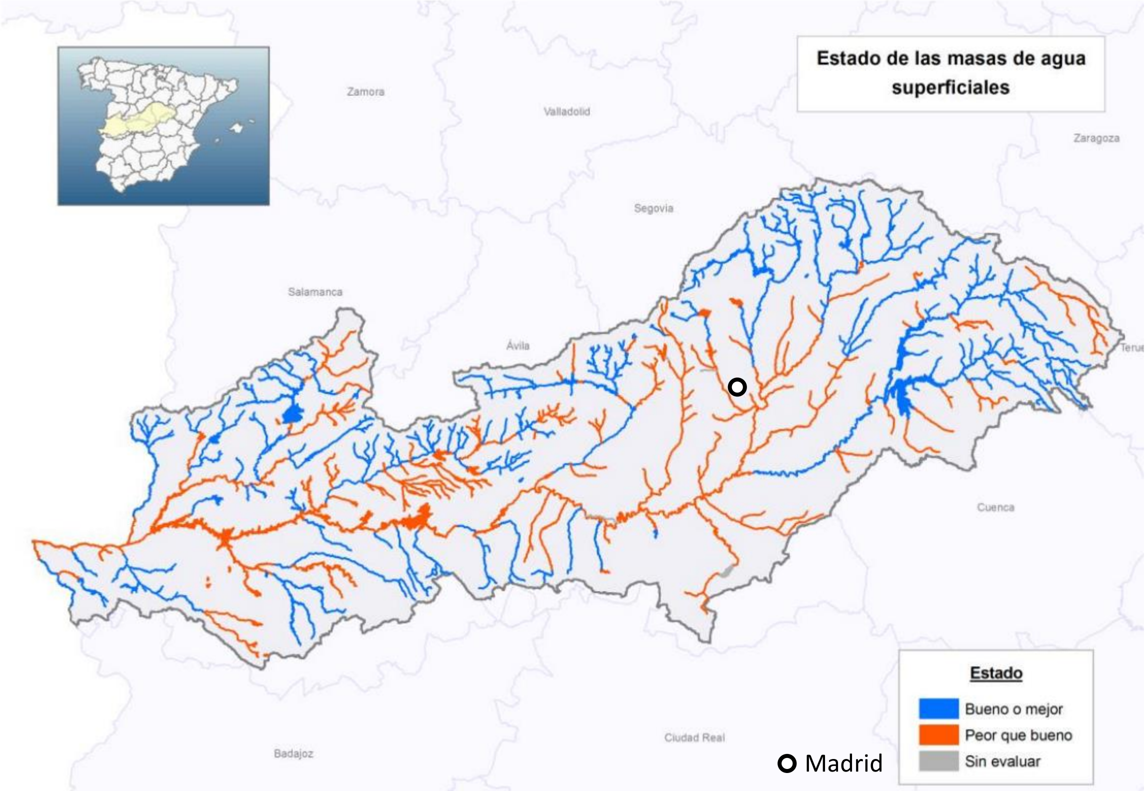

Figura 4. Estado de las masas de agua superficiales de la Demarcación del Tajo. Modificado a partir del informe del estado ecológico y químico de los ríos en la cuenca hidrográfica del Tajo (CHT 2018b)

Teniendo en cuenta estas diferencias entre las cabeceras y los tramos medios y bajos de los ríos considerados se delimita el área de estudio de la modelización matemática de contaminantes a la zona media de la cuenca, incluyendo los ríos a partir del punto donde empieza a deteriorarse su estado. Así para la modelización se considera el Jarama desde la confluencia del Guadalix hasta su desembocadura, el Henares desde aguas arriba de Alcalá y el Manzanares desde la presa del Pardo hasta su confluencia con el Jarama, y el Tajo desde Aranjuez hasta Toledo (Figura 5).

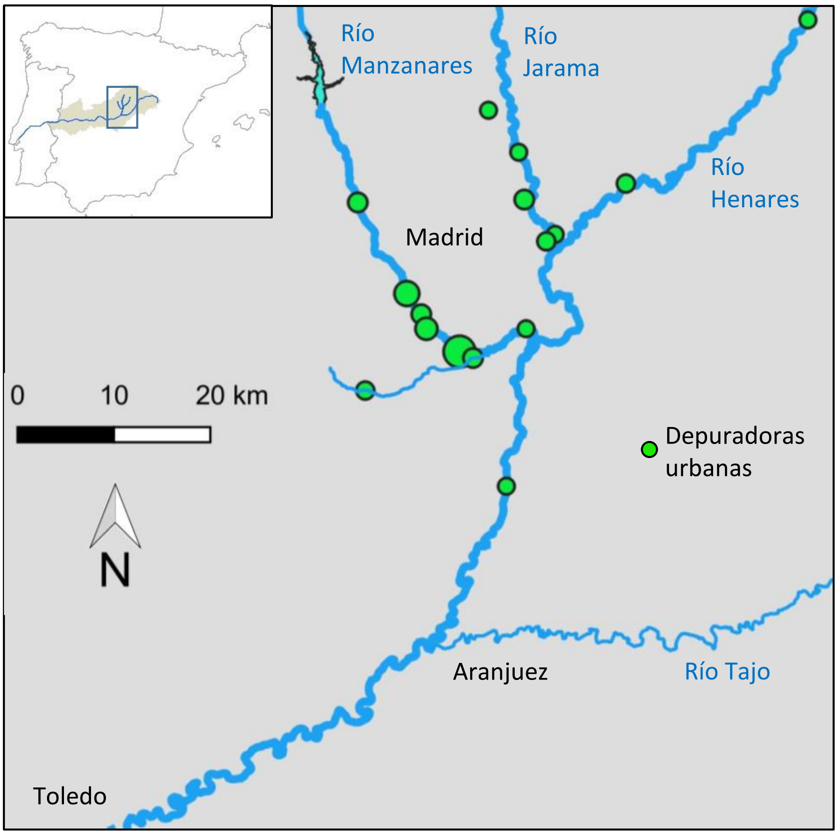

Figura 5. Área de estudio. El diámetro del círculo de cada depuradora es proporcional al volumen medio vertido.

Demarcaciones consideradas en la comparativa de exenciones

En el desarrollo de la comparativa de demarcaciones para identificar tendencias entre los umbrales de las exenciones (cuarto objetivo parcial, capítulo 8), se amplía el área de estudio para incluir varias demarcaciones europeas (Figura 6). La selección de las demarcaciones se basa en dos criterios. Por un lado, debe existir suficiente información disponible para caracterizar las presiones existentes y las exenciones declaradas. Por otro, el análisis tiene que abarcar cuencas hidrográficas con características topográficas, geológicas, climatológicas y demográficas lo suficientemente distintas para reflejar la diversidad del ámbito de aplicación de la DMA.

Figura 6. Cuencas comparadas en el modelo de presiones, impactos y exenciones.

4.2. La legislación aplicable a la calidad de las aguas superficiales continentales

La legislación aplicable a la calidad de las aguas superficiales emana de la Directiva Marco del Agua, piedra angular de la política europea de aguas desde su aprobación en el año 2000. La Directiva marcó un cambio de paradigma al pasar de una gestión basada en el aprovechamiento humano del recurso (para uso agrícola, urbano, etc.) a otra basada en la conservación del patrimonio medioambiental asociado al agua.

La Directiva requiere en sus considerandos 25, 26, 33, y en su Artículo 4.1.a.ii la consecución y conservación del buen estado de las aguas superficiales y subterráneas. En el caso de las aguas superficiales, el estado se determina (DMA Art. 2.17) como el peor entre el estado químico y el ecológico (Figura 7).

Figura 7. Valoración del estado de las aguas superficiales (Fuente: Confederación Hidrográfica del Tajo).

El estado químico se caracteriza por la concentración de sustancias prioritarias (metales pesados, hidrocarburos, pesticidas, etc.). Por su parte, para el estado ecológico se deben considerar los elementos de calidad biológica, hidromorfológica y físico-química (Figura 8). El Anejo V de la DMA nos indica las mediciones a tener en cuenta para caracterizar cada uno de estos elementos..

La implementación de la DMA conlleva una revisión de los objetivos de la planificación hidrológica. Su Artículo 40 establece que “la planificación hidrológica tendrá por objetivos generales conseguir el buen estado y la adecuada protección del dominio público hidráulico y de las aguas objeto de esta ley, la satisfacción de las demandas de agua, el equilibrio y armonización del desarrollo regional y sectorial, incrementando las disponibilidades del recurso, protegiendo su calidad, economizando su empleo y racionalizando sus usos en armonía con el medio ambiente y los demás recursos naturales”.

La regulación europea se transpone a la legislación española en varias normativas, entre las que destaca el Texto Refundido de la Ley de Aguas (TRLA) publicado en el Real Decreto Legislativo 1/2001 (Ministerio de Medio Ambiente 2001).

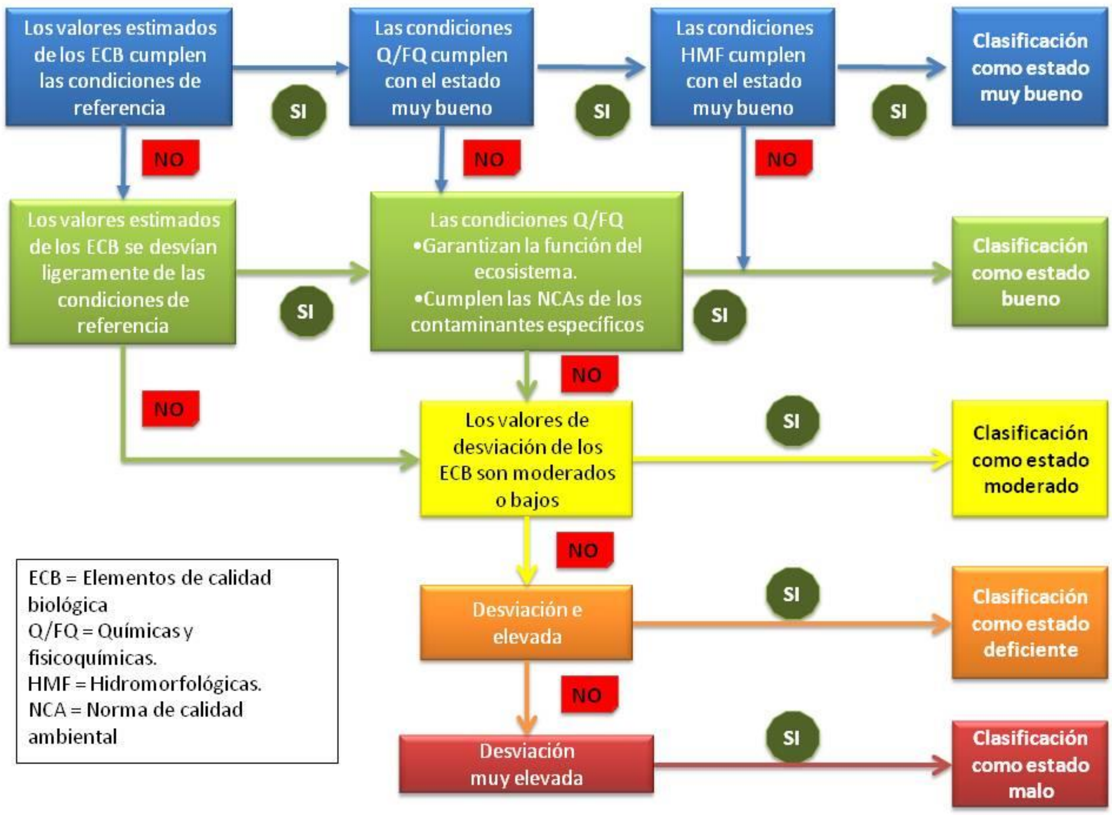

Figura 8. Determinación del estado ecológico de las aguas superficiales a partir de los elementos de calidad biológica, hidromorfológica y físico-química. Fuente: Ministerio de Transición Ecológica (2020a)

Por otro lado, la DMA requiere dividir las aguas superficiales en “masas de agua”, que constituyen la unidad de análisis y gestión. Cada masa de agua es clasificada según su tipología (EC 2003d). En particular, la tipología de las masas de agua del Tajo Medio se ha definido en el RD 1/2016 (MAPAMA 2016), Anexo V.

El RD 817/2015 (MAPAMA 2015) establece los criterios de seguimiento y evaluación del estado de las aguas superficiales y las normas de calidad ambiental. El Decreto identifica los indicadores que describen los diferentes elementos de calidad, así como los valores umbral a partir de los cuales se considera un cambio de clase de estado (Anexo II, Apartado A.2). Si como resultado del análisis el estado de una masa de agua es peor que bueno, se deberán definir medidas correctoras o justificar exenciones (apartado 4.4 de la DMA). En el caso de masas de agua definidas como “muy modificadas” el término de “estado ecológico” se sustituye por el de “potencial ecológico” (Artículo 2 de la DMA).

Como ejemplo, la Tabla 1 muestra los límites para una masa de agua tipo R-T15 (ecotipo de las masas de agua “Manzanares a su paso por Madrid”, “Jarama entre la confluencia del Henares y el Embalse del Rey” y “Jarama ente el Embalse del Rey y la confluencia del Tajuña”, entre otras). En particular, se aprecia cómo una concentración de amonio superior a los 0,6 mg/L se considera asociada a un estado peor que bueno. En los capítulos 5 y 6 se compararán los valores de las observaciones de las estaciones de medición con estos límites para caracterizar el estado actual de las masas de agua del área de estudio.

	Indicador	Unidades	Límite
Biológicos	IBMWP	-	72,2
	IMMi-T	-	0,7
	IBMR	-	6,3
	IPS	-	12,9
Hidromorfológico	QBR	-	100 (Muy bueno/Bueno)
Físico-Químicos	Ph	-	6-9
	Oxígeno	mg/l	5
	% Oxígeno	%	60-120
	Amonio	mg NH4/l	0,6
	Fosfato	mg PO4/l	0,5
	Nitrato	mg NO3/l	25

Tabla 1. Límites de cambio de clase de estado Bueno/Moderado para una masa de agua de tipo RT-15 (Fuente: RD817/2015).
IBMWP: Iberian Biomonitoring Working Party. IMMi-T: Índice Multimétrico de intercalibración 10. IBMR: Índice Biológico de Macrófitos en Ríos en España. IPS: Índice de Polusensibilidad Específica. QBR: Índice de Calidad del Bosque de Ribera.

4.3. El marco temporal de la gestión hidrológica: los ciclos de planificación

Los planes hidrológicos de demarcación son un instrumento clave en la implementación de la DMA. El Artículo 13 y el Anejo VII de la DMA describen la información que debe incluirse en estos planes y exigen su revisión cada 6 años. El Real Decreto 907/2007 (Ministerio de Medio Ambiente 2007) del Reglamento de la Planificación Hidrológica, establece los objetivos y criterios de la planificación, en armonía con la DMA. Dichos aspectos se ven desarrollados en la Orden ARM/2656/2008 (Ministerio de Medio Ambiente 2008), de Instrucción de Planificación Hidrológica.

Según esta legislación, los planes hidrológicos deben contener una serie de apartados definidos previamente. Primero, deben describir los recursos hídricos existentes, así como los usos y las presiones significativas. Seguidamente, se deben describir las zonas protegidas, así como las redes de control de seguimiento de estado. Deben incluir los objetivos medioambientales y los plazos previstos para su consecución, además de un Programa de Medidas para alcanzar dichos objetivos. Finalmente, se debe incluir un análisis económico del uso del agua y definir una política de precios teniendo en cuenta la recuperación de los costes relativos a los servicios del agua.

El plan hidrológico vigente en la Demarcación del Tajo corresponde al segundo ciclo de planificación (2015-2021) aprobado por el Real Decreto 1/2016 (MAPAMA 2016).

4.4. Las exenciones a los objetivos medioambientales

La legislación vigente considera que existen casos en los que el alcanzar los objetivos ambientales generales (buen estado o, en su caso, buen potencial) podría requerir medidas con costes desproporcionados o técnicamente inviables, entre otros supuestos. Para ello, los Artículos 4.4 y 4.5 de la DMA y los Artículos 36 y 37 del Real Decreto 907/2007 del Reglamento de la Planificación Hidrológica permiten extensiones temporales (es decir un plazo mayor para que se alcance el buen estado) o el establecimiento de objetivos menos rigurosos (que se permita excepcionalmente fijar otro valor objetivo de los indicadores en vez de llegar al buen estado). Dichas exenciones deben ir acompañadas de la correspondiente justificación.

En nuestra área de estudio, el apéndice 8.3 del Anexo V del Real Decreto RD1/2016 presenta la aplicación de exenciones a diversas masas de agua del Tajo Medio en el plan hidrológico vigente. La Tabla 2 enumera las masas de agua con objetivos menos rigurosos, así como el valor de los nuevos objetivos a cumplir.

Concentración en mg/l	*Amonio*	*Nitrato*	*DBO5*	*Fósforo total*
Jarama desde Tajuña a Tajo	10	25	10	1
Jarama desde E. Rey a Tajuña	10	25	8	1
Jarama desde Henares haste E. Rey	8	25	8	1
Jarama desde Valdebebas hasta Henares	10	25	8	1
Manzanares a su paso por Madrid	10	25	10	1
Arroyo Culebro	2	40	15	1

Tabla 2. Objetivos menos rigurosos en las masas de agua del Tajo Medio, en el plan hidrológico vigente (Fuente: RD1/2016)

Por tanto, en una masa de agua como el Manzanares a su paso por Madrid, la concentración máxima permitida de amonio pasa de ser 0,6 mg/l a 10 mg/l. La justificación de estas exenciones no es particularmente extensa en el plan hidrológico actual.

4.5. La legislación aplicable a los vertidos puntuales

La legislación vigente establece que cualquier vertido a las aguas superficiales debe contar con una autorización de vertido. El TRLA en su Artículo 100 especifica que "la autorización de vertido tendrá como objeto la consecución de los objetivos medioambientales establecidos."

Los vertidos de depuradoras urbanas, además, tienen una legislación propia que emana de la Directiva 91/271/CEE de Tratamiento de las aguas residuales urbanas (Council of the European Communities 1991c), actualizada con la Directiva 98/15/CE (Commission of the European Communities 1998). Dichas Directivas han sido transpuestas y desarrolladas en los Reales Decretos RDL 11/1995 (Ministerio de Medio Ambiente 2001), RD 509/1996 (Ministerio de Obras Públicas 1996) y RD 2116/1998

(Ministerio de Medio Ambiente 1998). La legislación establece explícitamente unos límites máximos de concentración de los siguientes contaminantes: demanda biológica y química de oxígeno, sólidos en suspensión (Tabla 3) y, en el caso de vertidos a áreas sensibles a eutrofización, nitrógeno y fósforo (Tabla 4). Es decir, de los contaminantes que afectan el estado físico-químico de las masas de agua (Anexo V de la DMA).

Concentración en mg/l	*Concentración*	*Porcentaje mínimo de reducción*
DBO5	25	70-90
Demanda Química de Oxígeno (DQO)	125	75
Total de Sólidos en Suspensión	35	90

Tabla 3. Concentración máxima de demanda de oxígeno y sólidos suspendidos en efluente de depuradora de más de 10 000 habitantes equivalentes (Fuente: RD 509/1996)

En el área de estudio, la Resolución de 6 de febrero de 2019 de la Secretaría de Estado de Medio Ambiente (Ministerio para la Transición Ecológica 2019) establece que el Embalse del Rey, en la confluencia del Manzanares con el Jarama, es una zona sensible a eutrofización por fósforo, lo cual implica la aplicación de los límites de fósforo en los efluentes de las depuradoras que vierten a su zona de captación (Manzanares a su paso por Madrid, Jarama entre el río Henares y el propio embalse).

Dichos límites de fósforo se describen en la Tabla 4. Por tanto, los efluentes de las grandes depuradoras de la zona metropolitana de Madrid (por encima de 100 000 habitantes equivalentes) tienen restringida explícitamente la concentración de fósforo total a 1 mg/l. Por el contrario, al no estar declarada la zona como sensible a eutrofización por nitrógeno, estos artículos no aplican para restringir la concentración de nitrógeno en los efluentes de depuradora. En los siguientes subapartados se exponen los otros artículos de la legislación que sí la restringen implícitamente.

Concentración en mg/l	*Habitantes equivalentes*	*Concentración*	*Porcentaje mínimo de reducción*
Fósforo total	De 10 000 a 100 000	2	80
	Más de 100 000	1	80
Nitrógeno total	De 10 000 a 100 000	15	70-80
	Más de 100 000	10	70-80

Tabla 4. Límites de concentración de nutrientes en efluentes de depuradoras vertiendo a zonas sensibles a eutrofización por fósforo y/o nitrógeno (Fuente: RD 2116/1998)

4.6. Integración de la legislación de vertidos con la legislación de aguas receptoras

Un aspecto fundamental de la legislación de vertidos es que independientemente de las restricciones explícitas expuestas, el objetivo principal de la infraestructura de depuración es la consecución del buen estado de las aguas receptoras expuesto en el apartado 4.2, como requieren los siguientes artículos de la legislación.

a) RDL11/1995 (actualizado y desarrollado en RD 509/1996, RD 2116/1998). Art 2i: "las aguas receptoras cumplan después del vertido, los objetivos de calidad previstos en el ordenamiento jurídico aplicable."
b) RD 1/2001 (TRLA), Art. 100 et al., y RD 849/86 (Ministerio de Obras Públicas y Urbanismo 1986), RD 606/2003 (Ministerio de Medio Ambiente 2003), RD1290/2012 (Ministerio de Agricultura 2012).
c) Directiva 91/271/CEE, Anexo I, Sec. B.4: "Se podrán aplicar requisitos más rigurosos para garantizar que las aguas receptoras cumplen con directivas aplicables."

Por tanto, vista la legislación de vertidos en su conjunto, la concentración de contaminantes físico-químicos en efluente de depuradora debe cumplir con los límites descritos en la Tabla 5.

mg/l	*Límite explicito en efluente*	*Límite implícito teniendo en cuenta la calidad del medio natural*	*Límite en efluente*
DBO5	25	El que permita alcanzar los objetivos de Oxígeno Disuelto en las aguas receptoras	El mínimo de los dos anteriores
DQO	125	El que permita alcanzar los objetivos de Oxígeno Disuelto en las aguas receptoras	El mínimo de los dos anteriores
SS	35	No aplica	35
Fósforo Total	1	El que permita alcanzar los objetivos de fosfatos en las aguas receptoras	El mínimo de los dos anteriores
Nitrógeno Total	No aplica	El que permita alcanzar los objetivos de amonio, nitrito y nitrato en las aguas receptoras	El que permita alcanzar los objetivos de amonio y nitrato en las aguas receptoras

Tabla 5. Concentración máxima de contaminantes físico-químicos en efluente de depuradora

Como se ha mencionado anteriormente, el instrumento que permite a las autoridades de cuenca definir unos límites de contaminantes en el efluente de cada depuradora es la autorización de vertido, según el Artículo 100 del TRLA.

4.7. La legislación aplicable a la contaminación difusa

No existen normas en la legislación que limiten explícitamente la cantidad de contaminantes físico-químicos emitidos a las aguas superficiales continentales por fuentes de contaminación difusa.

En el caso del nitrógeno, la Directiva 91/676/CEE (Council of the European Communities 1991a) relativa a la protección de las aguas contra la contaminación producida por nitratos procedentes de fuentes agrarias se limita a requerir un seguimiento de la cantidad de nitratos usados como fertilizantes en actividades agrarias, sin establecer un límite superior a partir del cual se podría incurrir en infracciones. En el caso de los demás contaminantes (como puedan ser el fósforo o la materia orgánica con demanda de oxígeno), ni siquiera existe una legislación para su seguimiento en el caso de fuentes difusas.

4.8. Los caudales circulantes: interacción entre cantidad y calidad de las aguas. Efecto del Acueducto Tajo-Segura

Otra presión que puede actuar sobre la calidad de una masa de agua es la detracción de parte de su caudal. Como hemos visto en el apartado 4.2, los límites de los contaminantes se definen en términos de concentraciones. Es decir, como carga de contaminante por volumen de agua circulante (por ejemplo, miligramos de amonio por litro de agua). Por tanto, para una cantidad dada de contaminante por unidad de tiempo, una reducción de caudal circulante implica un aumento de la concentración.

Un estudio de la calidad de las aguas de un área determinada debe englobar los distintos regímenes de caudales circulantes. En el caso de España y la demarcación del Tajo, se dispone de información diaria de los caudales mediante la Red Oficial de Estaciones de Aforo (ROEA). Además, se dispone de información con una mayor frecuencia temporal de las estaciones del Sistema Automático de Información Hidrológica (SAIH), aunque la densidad geográfica de esta red es mucho menor.

El río Manzanares se caracteriza por un desequilibrio entre los caudales circulantes y los efluentes de depuración, debido a la combinación de una gran cantidad de población que necesita verter sus aguas residuales tratadas y el relativamente pequeño caudal natural del río. Esto situación contribuye de manera sustancial a las altas concentraciones de contaminantes observadas.

En el Tajo Medio la principal detracción de agua corresponde al trasvase intercuencas canalizado por el Acueducto Tajo-Segura. Dicha infraestructura deriva una parte de las aguas de la cabecera del Tajo (Figura 9), aguas arriba de la confluencia con el Jarama, a la cuenca del Segura, provocando una disminución de los caudales circulantes en el eje del Tajo y consecuentemente una mayor concentración de contaminantes.

El ATS se construyó en la década de los 70 del siglo pasado, y empezó a trasvasar agua en 1979 (Ministerio para la Transición Ecológica 2020b). El objetivo de la infraestructura era derivar las aguas consideradas excedentarias de la cuenca del Tajo a la del Segura, que a pesar de su aridez (Confederación Hidrográfica del Segura 2020) cuenta con una agricultura intensiva que aporta más de 5 000 millones de euros anuales de exportación a la balanza comercial (SCRATS 2020).

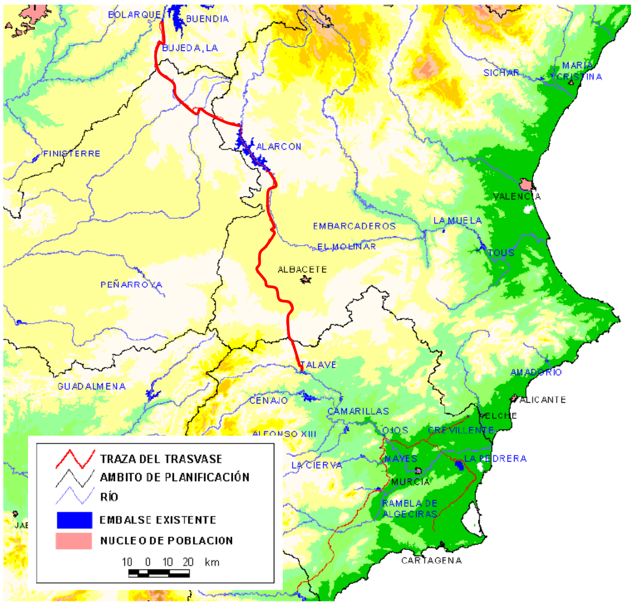

Figura 9. Plano de situación del Acueducto Tajo-Segura (Fuente: Plan Hidrológico Nacional, 2000)

En el Real Decreto 773/2014 (Ministerio de Agricultura 2014) se revisaron las normas de explotación del trasvase. Dichas normas solamente toman como referencia (Artículo 1, reglas de explotación del trasvase Tajo-Segura) la cantidad de agua embalsada en cabecera del Tajo para decidir los volúmenes máximos transvasables, no teniendo en cuenta el efecto causado sobre el estado de las aguas del Tajo aguas abajo de la detracción.

No existen estudios previos que relacionen los volúmenes transvasados con las concentraciones de contaminantes en la cuenca cedente. En el capítulo 6 de la presente Tesis se muestra la evolución esperada de la concentración de los contaminantes físico-químicos para distintos escenarios de volumen transvasado.

4.9. Contaminantes objeto del estudio

Una vez expuestas las diferentes normas que afectan la calidad de aguas superficiales, estamos en disposición de seleccionar los indicadores y contaminantes que serán objeto de nuestro análisis. Se han seleccionado los contaminantes físico-químicos (demanda biológica de oxígeno, amonio, nitrato, fosfato) por las siguientes razones:

- Son los únicos contaminantes para los que existen leyes que limitan su concentración tanto en las aguas receptoras como en los vertidos de las depuradoras urbanas.
- Aunque no existen límites para los vertidos de contaminación difusa, al menos existe una norma de seguimiento de nitratos.
- Las altas concentraciones de contaminantes físico-químicos son motivo continuo de no conformidad con los límites establecidos en las masas receptoras, en al menos diez de las doce masas de agua del área de estudio.
- Se puede modelizar su evolución en las aguas receptoras (Chapra 2008; Thomann and Mueller 1987). Lo cual facilita el establecimiento de relaciones causa-efecto entre las presiones aplicadas y el estado de las aguas receptoras.
- Existen mediciones, no exhaustivas pero sí suficientes, para calibrar los modelos de evolución. Tanto las redes de Control del Estado de las Masas De Agua Superficiales (CEMAS) de calidad de aguas (con mediciones trimestrales) como la Red Oficial de Estaciones de Aforo ofrecen información sobre el estado de las aguas receptoras. Por otro lado, se dispone de datos del caudal y concentración de contaminantes de los principales vertidos de depuradoras urbanas.

Por tanto, un estudio de estos contaminantes puede establecer recomendaciones sobre los cambios requeridos en las presiones para alcanzar el buen estado de las aguas receptoras.

Las mismas razones que nos habilitan para estudiar los contaminantes físico-químicos son las que aconsejan descartar otro tipo de indicadores y contaminantes. Los contaminantes químicos solamente han sido causa de incumplimientos temporales en dos de las doce masas de agua del área de estudio (capítulo 6) en el período medido (CHT 2018b). Se trata de casos puntuales con una afección limitada sobre la cuenca. En el caso de los indicadores biológicos, la dificultad de análisis radica en el insuficiente conocimiento de las relaciones causa-efecto entre las medidas tomadas y la evolución de los indicadores de biodiversidad del ecosistema (Palmer, Menninger, and Bernhardt 2010), así como de los periodos de tiempo requeridos para que las medidas surtan efecto. Lo cual dificulta la construcción de escenarios de mejora con suficientes garantías. Por otro lado, existe una serie de contaminantes cuya evolución en las aguas receptoras y efecto sobre los ecosistemas aún son escasamente conocidos, englobados bajo la etiqueta de “contaminantes emergentes” (Rosal et al. 2010). Además de las dificultades técnicas, la falta de desarrollo de la legislación aplicable imposibilita un estudio sistemático de presiones e impactos que conduzca a la emisión de recomendaciones de mejora.

El estudio de los contaminantes físico-químicos no está exento de dificultades. La primera está relacionada con la escasez de observaciones, tanto en las aguas receptoras como en las presiones. En el caso de las aguas superficiales del área de estudio, se dispone de las mediciones de la red CEMAS, con una buena distribución geográfica y diversidad de contaminantes medidos, pero una escasa resolución temporal con solamente una medición cada tres meses, de media. Otra red de medición, el Sistema

Automático de Información de Calidad de las Aguas (SAICA) ofrece una resolución temporal más alta pero mide un número limitado de contaminantes y la distribución geográfica de las estaciones es más dispersa. Respecto a las presiones, se dispone de mediciones mensuales (de media) de las concentraciones de contaminantes en los efluentes de las depuradoras. Las fechas de las mediciones no corresponden a las fechas en que se miden las concentraciones en las aguas receptoras, lo que obliga a modificar los métodos de calibración de los modelos (capítulo 6). En el caso de la contaminación difusa, no se dispone de mediciones de las emisiones que alcanzan las aguas superficiales continentales. Para el nitrógeno, se dispone de balances (DGA 2017) de cantidades anuales de nitratos agrícolas (cantidad de fertilizantes aplicadas al suelo, parte absorbida por las plantas, etc.). Todo ello nos obliga a trabajar con medias en estado estacionario, ya que no se dispone de suficientes mediciones para caracterizar los efectos transitorios.

4.10. Análisis integrador de la calidad de las aguas superficiales

Una vez conocidos los condicionantes normativos de las presiones, la calidad resultante en el medio receptor y los esfuerzos de mitigación exigibles, se constata la necesidad de aplicar un enfoque integrador que tenga en cuenta todos los aspectos implicados (Figura 10).

Primeramente, se analiza el efecto de las presiones (contaminación puntual y difusa, gestión de caudales) sobre la calidad del medio natural. Si los efluentes de depuración cumplen con los límites explícitos presentados en el apartado 4.5 y las aguas receptoras con los requerimientos del apartado 4.2, se considera cumplida esta parte de la legislación.

Si, por el contrario, se incumple alguna de las dos condiciones, se deben definir las medidas correctoras necesarias, que serán diseñadas, financiadas e implementadas por las autoridades competentes. Una vez analizado el coste, si se considera no desproporcionado (apartado 4.4) se procede a la implementación de las medidas y se reinicia el ciclo de monitorización de efluentes y aguas receptoras. Si el análisis revela un coste desproporcionado, se deben establecer unos objetivos menos rigurosos debidamente justificados, así como las medidas necesarias para alcanzarlos.

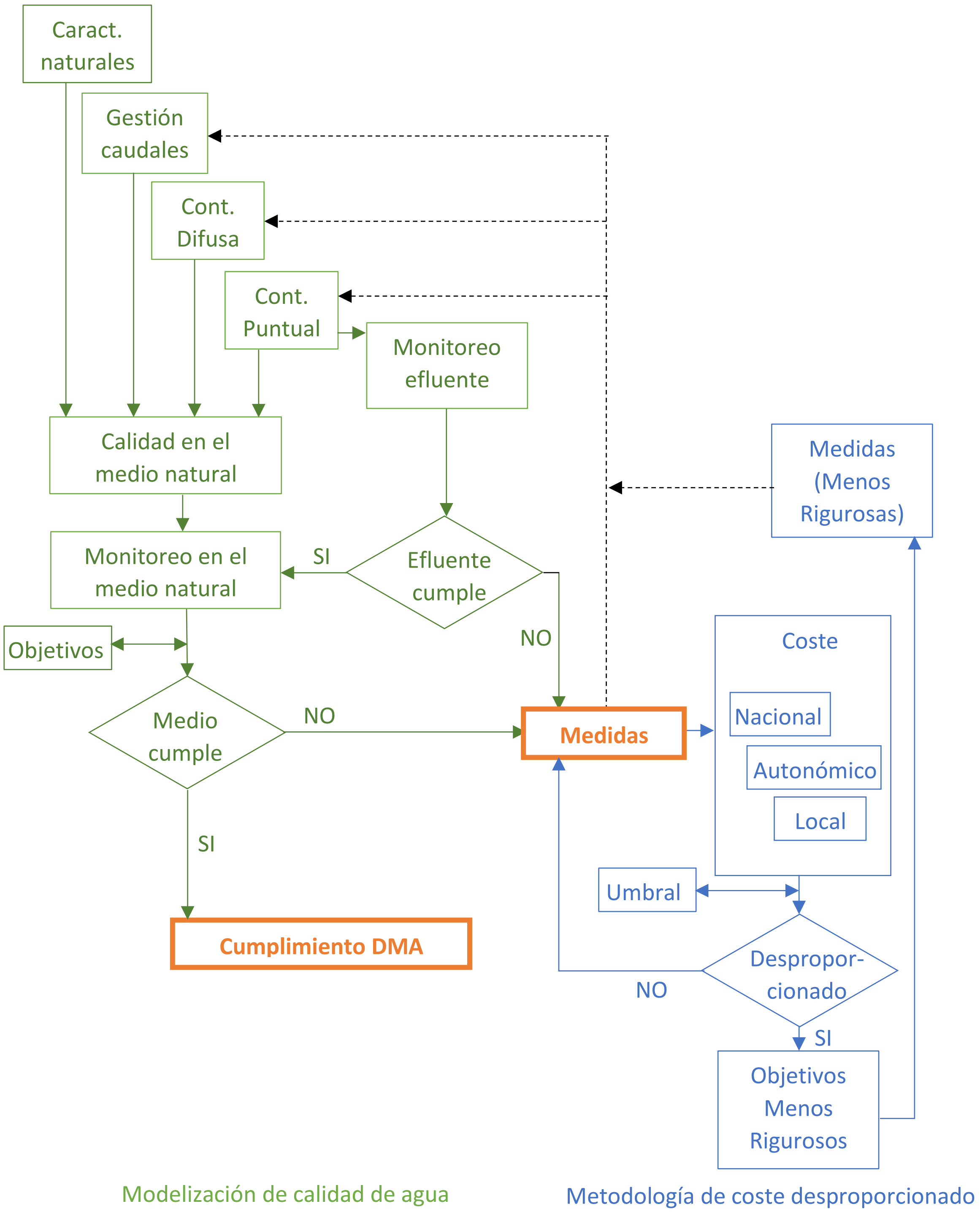

Figura 10. Diagrama simplificado de la gestión integral de calidad de aguas superficiales

5. Modelización de la calidad físico-química del Manzanares

5.1. Introducción

En este capítulo se expone la parte de la investigación que trata sobre la evolución de los contaminantes físico-químicos en el tramo del Manzanares entre la presa del Pardo y su desembocadura al Jarama. Los resultados se recogen en un artículo publicado en la revista Environmental Earth Sciences. El área de estudio se ha seleccionado bajo el criterio que fuera lo suficientemente pequeña para facilitar la comprensión de los procesos implicados, y lo suficientemente representativa para que las conclusiones fueran útiles para la gestión del agua en la zona. El límite del modelo aguas arriba se sitúa en Presa del Pardo, donde las aguas aún no han sufrido presiones significativas. El límite aguas abajo se ha fijado en la confluencia con el Jarama, donde el río Manzanares ha drenado la mayor parte de aguas depuradas de la ciudad de Madrid. El río Manzanares es particularmente importante por atravesar la ciudad más poblada de España, y porque la calidad de sus aguas afecta a decenas de kilómetros de aguas superficiales situadas aguas abajo (en los ríos Jarama y Tajo). Aproximadamente el 90% de las aguas que circulan aguas abajo de la ciudad de Madrid son efluentes de depuradora (CHT 2018a; CEDEX 2016), lo que (con la infraestructura de depuración actual) implica unas concentraciones de nutrientes (amonio, fosfatos) extremadamente altas (CHT 2018d).

5.2. Metodología

Se realiza primeramente un análisis estadístico de los datos de calidad físico-química de las aguas y de caudal disponibles, para conocer los valores de concentración media y variabilidad de los contaminantes en las aguas superficiales, así como en los efluentes de depuración. Estos valores se comparan con los límites legales para conocer el estado actual del área de estudio respecto a sus objetivos medioambientales.

Seguidamente se crea en varios pasos un modelo de evolución de los contaminantes físico-químicos. En primer lugar, se procede a una discretización de las masas de agua superficiales por tramos (Figura 11).

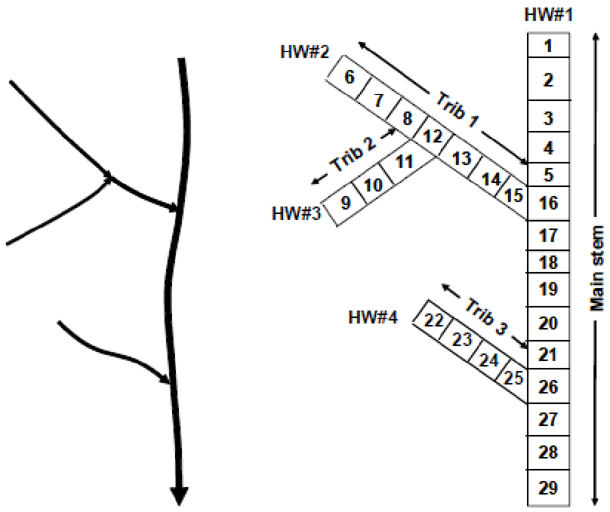

Figura 11. Ejemplo del proceso de discretización de las masas de agua (Fuente: Chapra 2008)

En cada uno de los tramos se establece un balance de masas para cada contaminante, permitiéndose su evolución química (Figura 12).

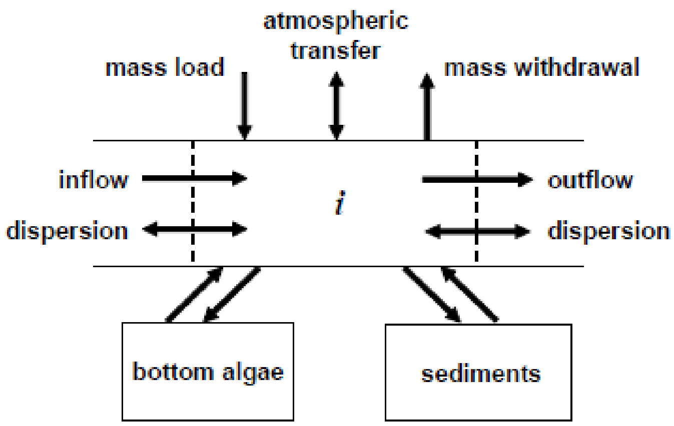

Figura 12. Balance de masas y evolución química en cada tramo de discretización del modelo (Fuente: Chapra 2008)

Se desarrolla un modelo de evolución de contaminantes usando el software Gescal (Paredes-Arquiola and Solera 2013) que incluye las aportaciones de cabecera y de cada una de las depuradoras del sistema (Figura 13).

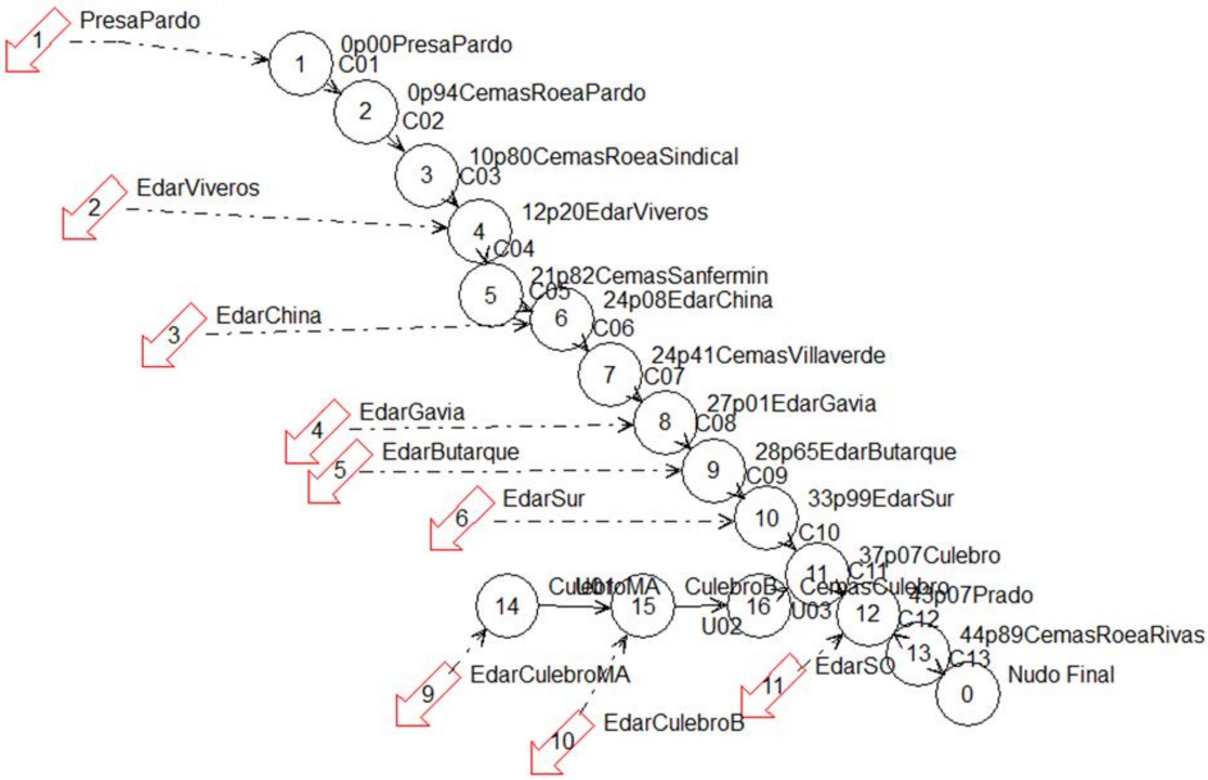

Figura 13. Modelo Gescal del Manzanares entre el Pardo y su desembocadura

La Figura 14 muestra las evoluciones químicas permitidas en el modelo (Paredes-Arquiola and Solera 2013):

- La materia orgánica, caracterizada por su Demanda Biológica de Oxígeno a 5 días puede descomponerse con consumo del oxígeno disuelto disponible en el agua.
- El nitrógeno orgánico puede mineralizarse a amonio. Por ejemplo, la urea (CH_4N_2O) presente en la orina puede liberar dos moléculas de amonio. El amonio a su vez puede nitrificarse con consumo de oxígeno disuelto, y el nitrato resultante puede desnitrificarse a nitrógeno atmosférico en condiciones de anoxia.
- El fosfato por su parte puede sedimentar de manera autónoma.
- Finalmente, el oxígeno disuelto disponible en el agua puede consumirse en los procesos descritos previamente (descomposición de materia orgánica y nitrificación del amonio), y puede reponerse hasta su concentración de equilibrio por intercambio gaseoso con la atmósfera.

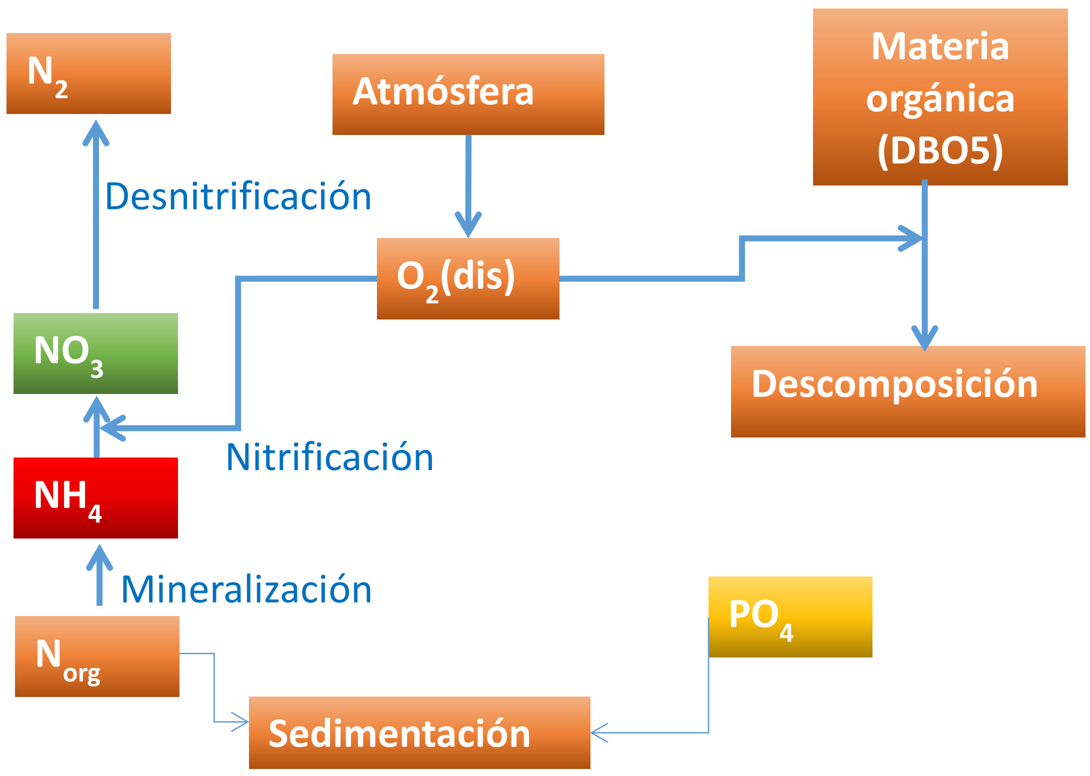

Figura 14. Evolución química de los contaminantes (Fuente: Manual Gescal)

Para cada una de las reacciones descritas y para los fenómenos de difusión y transporte se definen unos factores de evolución de primer orden que se deben calibrar en cada tramo. El conjunto de ecuaciones diferenciales se integra resultando en relaciones exponenciales de evolución en los tramos que transcurren entre puntos de introducción de presiones externas. Los coeficientes se calibran utilizando las observaciones existentes.

Una vez calibrado el modelo, se procede a cuantificar dos cuestiones fundamentales. En primer lugar, qué peso relativo tiene cada una de las presiones en la cantidad de contaminantes que encontramos en las aguas superficiales. Seguidamente, qué cambios se deben implementar en las presiones para que las aguas receptoras alcancen el buen estado requerido por la Directiva Marco del Agua y su transposición a la legislación española.

5.3. Resultados y discusión

El primer resultado del estudio es la constatación de que las concentraciones de algunas sustancias observadas en las aguas receptoras son muy superiores a los límites de buen estado definidos en la legislación. Y ello, a pesar de que los efluentes de las depuradoras se ciñen a los límites numéricos de las tablas de la legislación de depuración (Tabla 3 y Tabla 4 en el caso de las grandes depuradoras del área de estudio). Por tanto, se necesita la aplicación de medidas adicionales para alcanzar el buen estado requerido por la DMA.

El análisis indica que para alcanzar el buen estado de las aguas receptoras en el curso bajo del Manzanares se debería requerir en los efluentes de las depuradoras del Manzanares a su paso por Madrid una concentración de 11 mg/l de Demanda Biológica de Oxígeno (DBO5), 0,65 mg/l de amonio, 26 mg/l de nitrato y 0,51 mg/l de fosfato. Madrid y el Manzanares se presentan como un caso en el que para cumplir con la legislación medioambiental en materia de agua se necesitan límites más estrictos que los genéricos descritos por la legislación de depuración. Además, estos límites están definidos para las fracciones de interés de cada nutriente (amonio, nitrato, fosfato) en vez de las métricas de nitrógeno total y fósforo total usadas por la Directiva de tratamiento de aguas residuales urbanas.

5.4. Conclusiones

La primera conclusión que arroja el estudio es la apreciación de cierta falta de coordinación entre las dos Directivas principales que relacionan los vertidos urbanos y la calidad de las aguas superficiales. Por un lado, se utilizan métricas distintas para los contaminantes de interés (nitrógeno y fósforo totales en los vertidos cuando corresponde, amonio, nitrato y fosfato en las aguas receptoras). Por el otro, aunque la directiva de tratamiento de aguas residuales urbanas cita explícitamente la necesidad de que la calidad de los vertidos sea compatible con el cumplimiento de la legislación de las aguas receptoras, la cláusula no se cumple en la aplicación práctica. Se requeriría un cambio de la Directiva de Tratamiento de aguas residuales urbanas para alinear los contaminantes limitados a los definidos por la DMA, y afianzar el criterio de cumplimiento de la calidad de las aguas receptoras.

El estudio permite determinar los límites de concentración que deberían respetar los efluentes de las depuradoras para que se pudiera alcanzar el buen estado de la masa de agua receptora. Con ello se solucionarían los problemas físico-químicos del tramo estudiado del Manzanares. Seguirían requiriéndose acciones de mitigación en otras masas de agua cercanas con grandes impactos (Jarama, Tajo) por lo que la continuación natural de este análisis consiste en ampliar el marco geográfico del estudio. Asimismo, la existencia de exenciones a los objetivos medioambientales en el plan hidrológico de demarcación vigente requiere un estudio más detallado de las condiciones bajo las cuales estas exenciones están justificadas. Ambos aspectos son el objeto de los siguientes capítulos.

5.5. Artículo

A continuación se reproduce, con el permiso de los coautores, el contenido del artículo:

Antonio Bolinches, Lucia De Stefano, Javier Paredes-Arquiola (2020) Adjusting wastewater treatment effluent standards to protect the receiving waters: the case of low-flow rivers in central Spain. Environmental Earth Sciences 79, 446 (2020). DOI: 10.1007/s12665-020-09184-z

Adjusting wastewater treatment effluent standards to protect the receiving waters: the case of low-flow rivers in central Spain

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ABSTRACT

Freshwater quality is deteriorating worldwide. In populated areas, urban pollution is the main pressure on surface continental waters, but intensive wastewater treatment is costly. Setting standards for treatment of wastewater before discharge is a major policy instrument for water authorities, balancing environmental gains and operational costs. Discharge permits usually define concentration limits at the discharge point of the plant effluent. This approach, however, may not guarantee the good status of the receiving waters. Discharge permits should be directly linked to pollutant concentration in the river. Our paper develops an approach to adaptively adjust discharge permits, and applies it to Madrid and the Manzanares river, a city of more than 3 million inhabitants discharging its treated wastewater to a stream having less than 2 $m^3 s^{-1}$ average flow. Stricter limits to 5-day biological oxygen demand (11 mg O_2 L^{-1}), ammonium (0.5 mg N-NH_4 L^{-1}), nitrate (5.9 mg N-NO_3 L^{-1}) and phosphate (0.17 mg P-PO_4 L^{-1}) at plant effluent are required to meet the river environmental objectives. The results can be generalized to assess wastewater management decisions in other geographical areas.

KEYWORDS:

Wastewater, Water Framework Directive, Water quality, Tagus basin

1. Introduction

Freshwater quality is degrading worldwide due to increased human pressure (United Nations Environment Program 2015). Among the stressors, wastewater treatment plant (WWTP) effluents are a major contributor of pollutants especially in densely populated areas (Carey and Migliaccio 2009). Considerable research attention has been devoted to the quality of water streams in urban environments. Apart from the changes in hydrology and geomorphology, Paul and Meyer (2001) identified a general increase in the concentration nutrients, metals, pesticides, and other contaminants in urban rivers, associated with city growth. Similar conclusions were drawn by McGrane (2016), who further acknowledged that our understanding of dynamics of pollutants remain limited and a priority area for continued research. Nevertheless, an empirical study on the world's largest cities (Duh et al. 2008) found that urban growth is not necessarily linked to increasing pollution, since technological advances and environmental policy can mitigate the effects.

The degree of urban pollution that is acceptable for the society is a political decision to be settled in the legislation. On the one hand, excessive pollution negatively affects freshwater ecosystems (Kloas et al. 2009; Bahamonde et al. 2015; Jasinska et al. 2015). On the other, reduction of pollutant concentration at WWTP requires an economic effort (F. Hernández-Sancho, Molinos-Senante, and Sala-Garrido 2011).

This decision is often implemented through the conditions of the WWTP operating permit that the competent authority grants to the agency responsible for sanitation. Due to technical and jurisdictional issues, these permits often define pollutant concentration at WWTP effluent (Morris et al. 2017). There are cases where standard concentration limits at WWTP effluents result in a poor state of the receiving waters. In those cases, effluent concentrations limits linked to the status of the receiving waters should be defined instead (Corominas et al. 2013).

In the European Union (EU), WWTP effluent limits were defined in the Urban Waste Water Treatment Directive (UWWTD) (Council of the European Communities 1991c). Official standard limits are set in Annex I tables, and the clause that "more stringent requirements (…) shall be applied where required to ensure that the receiving waters satisfy any other relevant Directives" (Annex I, B.4) remains void in many contexts. One decade later, the Water Framework Directive (WFD) (European Parliament and Council 2000) presented a more holistic approach to water management. It set the focus on the good status of water ecosystems. This implied that WWTP effluent limits should ensure the good status of the receiving waters. After almost 20 years of implementation, however, only 38% of the surface water bodies in the European Union are in good status (European Commission 2019a).

In the case of densely populated areas, WWTP permits need to be adjusted to ensure that the effluents do not pose a threat to the receiving waters. More stringent effluent limits imply higher infrastructure and operating costs. The competent water authorities may therefore encounter opposition from WWTP operators, and need sound scientific evidence to support changes to discharge permits. Some approaches are based on a statistical description of the water quality observations (Soares Cruz et al. 2019; Asad Ismaiel et al. 2018). These methods are very powerful to identify trends but are not physically-based models, which constrains their ability to predict the effect of changes in WWTP effluents on the receiving waters.

For physico-chemical pollutants, the scientific evidence is built on water quality models. Developed in the last 40 years (Thomann and Mueller 1987; Chapra 2008), these models are a useful tool to predict the evolution of pollutants like Biological Oxygen Demand (BOD5) Ammonium (NH_4^+), Nitrate (NO_3^-) and Phosphate (PO_4^-). These pollutants, largely present in urban sewage effluents, may produce anoxia and eutrophication in the receiving waters (Dodds and Smith 2016; V. H. Smith, Tilman, and Nekola 1999; Anderson, Glibert, and Burkholder 2002) and the subsequent stress on the ecological status of the waters. Such water quality models have been successfully implemented to analyze pollutant evolution in the Seine river in Paris (Even et al. 2007; Sferratore et al. 2005), in the Thames river in London (Cox and Whitehead 2009), in the

Rhine basin (Loos et al. 2009), and in urban rivers in UK (Astaraie-Imani et al. 2012; Hutchins and Bowes 2018), Taiwan (Chang et al. 2015) and India (Griffiths et al. 2017), to cite a few examples. Recent studies in EU rivers (Alexakis, Kagalou, and Tsakiris 2013; Jin et al. 2016) have applied such models in the context of the WFD. But to date, scant attention has been paid to the particular limits to be imposed to WWTP effluents in order to ensure that receiving waters comply with WFD requirements (Corominas et al. 2013). There are very few case studies in the scientific literature that may serve as reference for management decisions.

In complex scenarios with multiple WWTPs the measures required by the WFD to achieve its environmental objectives may be phased in to ease the financial effort. In that case, the assumptions made for the water quality model may need revision after each investment phase. For this reason, adaptive permits with shorter renewal time are preferred (Morris et al. 2017). That is, river response should be monitored after each plant upgrading, so that it is possible to revise the model assumptions and adjust the conditions of the discharge permits.

This study explores how more stringent limits in WWTP effluents can be defined to ensure a good status of the receiving waters, according to the WFD standards. This is addressed for the case of the city of Madrid and the Manzanares River, in the Tagus River Basin (Spain). The aggregate discharged flow of Madrid WWTPs can amount to 90% of the flow of the river (MAPAMA, 2018). Measurements of physico-chemical elements in the Manzanares River downstream of Madrid show systematic non-compliance of pollutant concentration objectives. In particular, average measurements of ammonium in the river are consistently above 15.6 mg N-NH_4 L^{-1}. This compares to the good status limit of 0.5 mg N-NH_4 L^{-1} and the current less stringent concentration objective of 7.8 mg N-NH_4 L^{-1}(MAPAMA, 2016). The high concentration of pollutants in this stretch of the Manzanares affects not only the local ecosystem, but also the quality of the water bodies downstream. Figure 1 shows how Manzanares waters affect Tagus river through the Jarama tributary. The effect of this pollution can be observed in water quality stations in the Tagus river in the vicinity of Toledo city (CHT Confederación Hidrográfica del Tajo 2018d), 100 km downstream of Madrid, where episodes of foam formation on the river`s surface have been documented (Gallego Bernad and Sánchez Pérez 2006).

To date, few studies have sought to examine the causes of this non-compliance and its relationship with the composition and volume of WWTP effluents from Madrid. Cubillo et al. (1992) and Paredes et al. (2010) developed water quality models for the Madrid region and Manzanares, respectively. While their theoretical frameworks remain valid, contour conditions have evolved with the construction of new WWTPs and the approval of new legislation.

In this paper the current pollutant concentration is characterized, both for Madrid WWTP effluents and at the Manzanares river waters. Then, they are compared with those established in the legislation with the aim of identifying potential noncompliance with the official standard limits (defined in UWWTD Annex I table) and the policy goal (achievement of good status) of the law. A water quality model is then presented to understand and quantify the pressure-status relation of WWTP effluents and river water quality. Finally, some change scenarios are introduced to identify the WWTP discharge characteristics that would be compatible with the good status of the receiving waters. Arguably, these should be the limits defined in WWTP permits.

2. Materials and Methods

2.1. Study Area

The study area comprises the Manzanares River from the outlet of El Pardo reservoir upstream of Madrid until the river junction with the Jarama River (figure 1). Jarama is a major tributary to the Tagus River. This stretch of the river receives more than 85% of the treated waste water of the city of Madrid (Madrid City Council 2018).

The city of Madrid has a population of 3.2 million inhabitants (Madrid City Council 2017). Average water consumption in 2014 for the region of Madrid was 217 L d^{-1} per inhabitant (INE, 2018). The local climate is classified as hot-summer Mediterranean (Csa) in the Köppen–Geiger climate classification system (Kottek et al. 2006). Mean annual precipitation is 421 mm yr^{-1} (AEMET 2018), with dry summers (average precipitation of 10 mm $month^{-1}$) and wet autumns (60 mm $month^{-1}$ on average). Mean monthly atmospheric temperature ranges from 2.7 to 32.1 degrees Celsius, with a mean annual temperature of 15.0 degrees Celsius (AEMET 2018). Potential evapotranspiration is in average 1100 mm yr^{-1} (IGN 2020). The annual average of days with a precipitation greater than 1 mm is 59 days, meaning that 84% of the time WWTPs work in dry weather conditions.

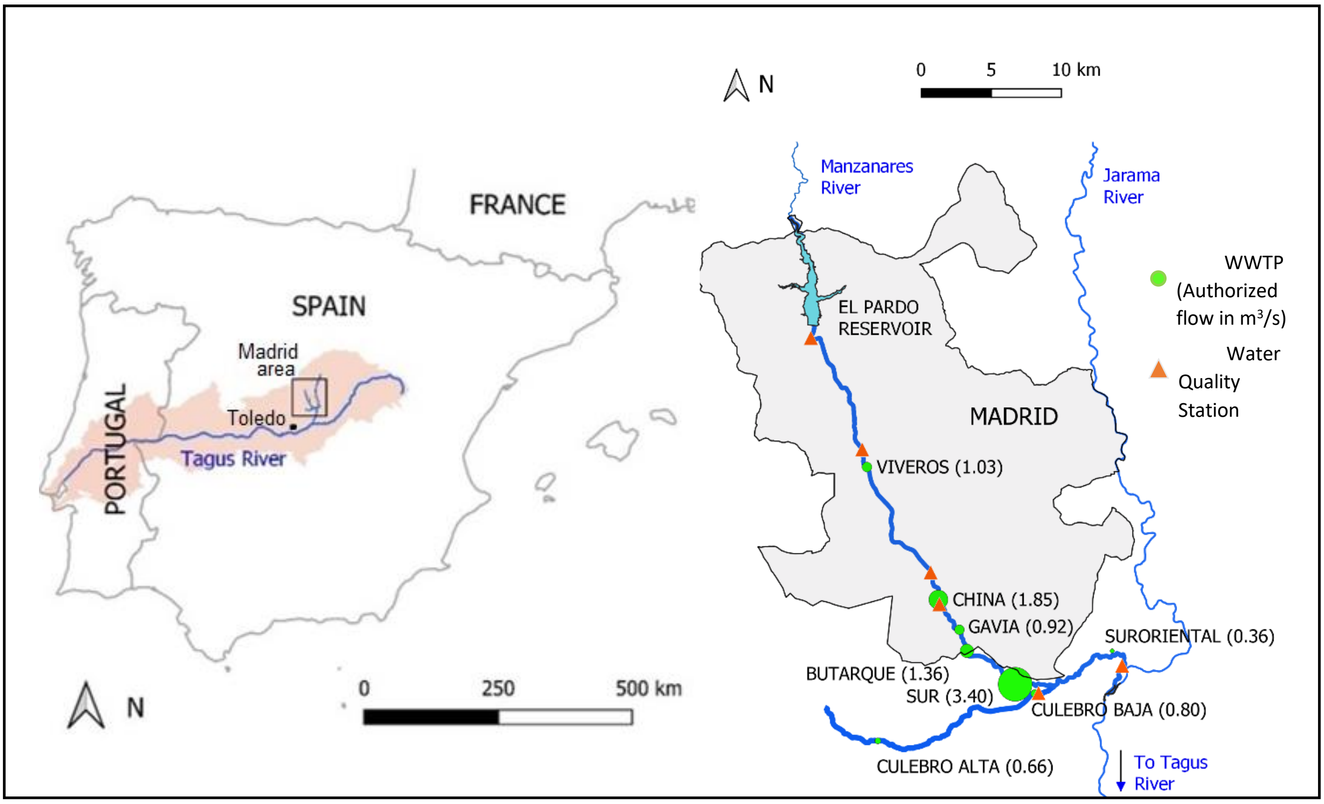


Fig. 1 Location of study area

The Tagus River Basin Authority (TRBA) is responsible for the river quality management and for issuing the WWTP discharge permits. The management of WWTPs is under the responsibility of Madrid City Council.

The stretch of Manzanares under study has a length of 45 km. The river evolves from an almost pristine mountain stream to a collector of WWTP effluents. El Pardo dam controls the flow upstream of Madrid city. This results in a highly regulated regime with an average flow of 1.5 $m^3 s^{-1}$. The combined flow of the authorized WWTPs discharges is 10.4 $m^3 s^{-1}$ (see figure 1).

The population of Madrid has been growing steadily from over 157,000 inhabitants in 1842 to the current figure above 3 million inhabitants (INE 2018). Wastewater treatment infrastructure has developed accordingly, starting with the first WWTP built in 1940. In 1977 an Integral Plan of Sewage Treatment was declared to apply a secondary treatment to the existing plants (CYII Foundation, 2015). Facing increased population and a growing concern about eutrophication, subsequent plans were implemented to build new WWTPs and add nutrient reduction cycles to some of the existing ones.

2.1.1. WWTP Effluent Quality Requirements

Table 1 shows the requirements set by Annex I of the UWWTD for the total suspended solids (TSS), biological and chemical oxygen demand (BOD5 and COD), and total P and nitrogen in WWTP effluents. We refer to these as the “standard” values since they must be followed by default. Since the Manzanares watershed was declared as a catchment area of sensitive areas to eutrophication by phosphorous (MAPAMA, 2011), the standard requirement of 1 mg P L^{-1} applies to all the WWTP effluents. On the contrary, the total N limit does not apply to Madrid WWTPs since the area has not been declared sensitive to eutrophication by N.

Parameter	*TSS*	*BOD5*	*COD*	*Total P*	*Total N*[a]
Requirement (mg L^{-1})	35	25	125	1	10

Table 1 UWWTD standard requirements for effluents from agglomerations above 100,000 h-e. [a] Does not apply in the area of study

TSS: Total Suspended Solids, BOD5: 5-day Biological Oxygen Demand, COD: Chemical Oxygen Demand

According to the Spanish Water Act, any discharge to surface water bodies must be authorized through a specific permit that sets the quality requirements for the effluent. Table 2 summarizes the specific requirements set by the TRBA in the discharge permits for the WWTP effluents in the study area (we refer to these emission controls as “specific” requirements, as opposed to the “standard” requirements of the previous paragraph).

WWTP	*Issue date*	*TSS*	*BOD5*	*COD*	*Total P*	*Total N*	*N-NH_4*
Viveros	31-12-2006	20	20	-	1	-	-
China *Butarque*	31-12-2021	35	25	125	1	10	-
Gavia	15-11-2010	35	25	125	1	10	-
Sur	23-08-2010	35	25	125	1	-	-
	31-12-2021	35	25	125	1	10	-
Culebro Alta	14-03-2007	35	25	125	1	-	0.8
Culebro Baja	31-12-2021	35	25	125	1	10	-

Suroriental	31-12-2006	20	20	-	1	-	-
	31-12-2021	35	25	125	1	10	-

Table 2 Current and future WWTP requirements of maximum pollutant concentration (mg L^{-1}). TSS: Total Suspended Solids, BOD5: 5-day Biological Oxygen Demand, COD: Chemical Oxygen Demand

As shown in Table 2, current discharge requirements set by the TRBA for all WWTPs except one (Gavia) will change in 2021. The new requirements will be aligned with the UWWTD standard values.

2.1.2. River Water Quality Objectives

The attention is now set to the concentration objectives established by the WFD and its transposition to the Spanish legislation for the Manzanares River. Table 3 reports the pollutant limits associated with a good ecological status or potential for water bodies having the ecological type of Manzanares River. These limits are currently overridden by the less stringent requirements set by the Tagus River Basin Management Plan.

	DO	*BOD5*	*N-NH_4*	*N-NO_3*	*P-PO_4*	*Total P*	*pH*
Ecotype RT15, Low mineralization Continental-Mediterranean rivers	5	-	0.5	5.6	0.16	-	6 to 9
Less stringent requirements for Manzanares River	-	10	7.8	5.6	-	1	-

Table 3 Concentration objectives for the physico-chemical elements for Manzanares River at Madrid according to baseline RD817/2018 for the river ecotype and less stringent requirements set by RD1/2016 (mg L^{-1}). Values shown are

maximum acceptable concentration, except for DO (minimum acceptable concentration) and pH (acceptable range)

DO: Dissolved Oxygen, BOD5: 5 day Biological Oxygen Demand

2.2. Data collection

The data used in this study were obtained from the following data sets:

- Digital Elevation Model, obtained from the National Geographical Institute (IGN, 2018) and remote sensing imagery (Google Earth 2018).
- River flow historical data collected and published by the Ministry for environmental affairs (MAPAMA, 2018).
- River water quality measurement data published by the TRBA and additional measurements collected by the municipality on a 3-day basis. They consist of laboratory analysis of water samples measuring water temperature, pH, conductivity, suspended solids, dissolved oxygen, ammonium, nitrate, total P and phosphate concentrations. The measurement error of the methods used to determine pollutant concentration (spectrophotometry, inductively coupled plasma mass and ion chromatography), is below 6%.
- Madrid WWTP effluent flow and pollutant concentration collected by TRBA (CHT Confederación Hidrográfica del Tajo 2018d) and Madrid’s water utility company on a daily basis.

The analysis is conducted for the most recent period with enough data available (January 2016 to August 2017), in order to work with wastewater treatment levels and river dynamics that reflect current conditions as much as possible.

2.3. Methodological approach

The methodological approach includes three steps: a) the analysis of compliance of the observed concentrations in the WWTP effluents and in the receiving waters with the legal standards presented in the previous pages; b) the construction of an hydrological model to simulate the river flow according to the point source and rainfall contribution; and c) the development of a water quality model to identify the changes in the quality of WWTP effluents needed to comply with WFD river quality objectives.

The parameters of the hydrological and water quality models are calibrated with the observed data. The model is run with different scenarios to predict the maximum WWTP effluent concentration that is compatible with compliance of WFD in the receiving waters.

2.3.1. Analysis of compliance with water quality standards

The raw data describing water quality in WWTP effluents and in river streams is analyzed statistically to calculate the yearly averages and quartiles, and compare them with the prescribed limits in the legislation.

2.3.2. Hydrological Modelization

It is assumed that in-stream flow responds to the following inputs: point source discharges (El Pardo reservoir, WWTPs), rainfall contribution and water losses through evaporation and infiltration in the riverbed.

Point source discharge is known but rainfall contribution and water losses need to be assessed. A model is built, based on the SCS method (Soil Conservation Service 1972) for the rainfall-runoff relation (linearized for simplification):

$$Q_{downstream} = \sum Q_{point\ source} - U + [P - I_0] \cdot F \qquad \text{(Eq.1)}$$

U evaporation and infiltration loss (m^3/s)

P precipitation ($mm\ yr^{-1}$)

I_0 average initial abstraction ($mm\ yr^{-1}$)

F Flow to net precipitation coefficient (m^3 s^{-1} mm^{-1})

Coefficients (U, I_0 and C) in the hydrologic model (Eq. 1) are calibrated to maximize the NS coefficient (Eq. 2) (Nash and Sutcliffe 1970). PBIAS coefficient (Eq. 3) will also be used for the validation. Since there is more hydrological data available than water quality records, the period from January 2016 to June 2017 is used for the calibration and the period from July to November 2017 is used for the validation (figure 6).

$$\mathrm{NS} = 1 - \frac{\sum_i^n (Q_{oi} - Q_{si})^2}{\sum_i^n (Q_{oi} - \overline{Q_o})^2} \qquad \text{(Eq. 2)}$$

$$PBIAS = \frac{\sum_i^n (Q_{oi} - Q_{si})}{\sum_i^n Q_{oi}} 100 \qquad \text{(Eq. 3)}$$

Where: Q_{oi} observed flow $\overline{Q_o}$ mean of observed flow

Q_{si} simulated flow

2.3.3. Water Quality Modelization

Well-established water-quality kinetics (Chapra 2008; Thomann and Mueller 1987) are applied to the river stream. This allows inferring the evolution of pollutants under different conditions (see Scenarios in the following section).

The river is modeled as a one-dimensional stream with perfect horizontal and vertical mixing in the cross section. It is discretized into finite longitudinal sections. Then, mass balance and reaction kinetics for each contaminant are applied to calculate the equilibrium concentrations (Eq. 4).

$$\frac{d}{dx}\left(E\frac{dC}{dx}\right) - \frac{d}{dx}(uC) + \frac{S_d}{V} + \sum W_i = 0 \qquad \text{(Eq. 4)}$$

E dispersion (m^2 d^{-1})

u stream average speed (m d^{-1})

C concentration (g m^{-3})

S_d load (g d^{-1})

W_i in-flow reactions (g m^{-3} d^{-1})

Software Aquatool/Gescal (Andreu, Capilla, and Sanchís 1996) is used for the calculations with a 10-day time step. This is consistent with the available data of one observation in the stream every 3 days. Gescal has been successfully used to assess the evolution of several water courses (Momblanch et al. 2015; J. Paredes-Arquiola et al. 2014; Javier Paredes-Arquiola et al. 2010).

The transport and fate of physico-chemical pollutants in each discrete section is modeled assuming first-order kinetics for the following components:

- Dissolved oxygen in the water column. It may be lost through degradation of organic mass or nitrification of ammonia. It may be replenished through aeration (reaeration constant K_a) at the water-air interface (Arora and Keshari 2018; Bowie et al. 1985).
- Biological oxygen demand. The presence of degradable carbonaceous organic matter is modeled through its BOD. It can precipitate to the sediment (modeled with a settling speed v_{sc}) or degrade to inorganic compounds (kinetic constant K_d) with oxygen consumption.
- Nitrogen compounds (Organic N, ammonium, and nitrate). Organic N can precipitate or be broken down to ammonia (Dojlido and Best 1993), modeled with first-order kinetic constant K_{norg}. The ammonia/ammonium may be nitrified (kinetic constant K_{nitr}) with oxygen consumption, and nitrate may be reduced to gaseous N_2 in anoxic conditions.

- Phosphate and total P. Both pollutants are modeled independently, being able to degrade at a first-order rate without interaction with the other elements.

Further explanations on in-flow reactions can be found in Gescal manual (Paredes-Arquiola and Solera 2013).

The model is calibrated using available time series of the pollutant input and river water quality measurements. Model parameters are allowed to change within ranges established in the literature (Paredes-Arquiola and Solera 2013; Bowie et al. 1985) in order to maximize model performance, evaluated through the percent bias (PBIAS, Eq. 5) and coefficient of determination (R^2, Eq. 6) (Moriasi et al. 2007; Santhi et al. 2001; Singh et al. 2005; Donigian 2002).

$$PBIAS = \frac{\sum_i^n (o_i - s_i)}{\sum_i^n o_i} 100 \qquad \text{(Eq. 5)}$$

$$R^2 = \left[\frac{\sum_i^n (o_i - \bar{o})(s_i - \bar{s})}{\left[\sum_i^n (o_i - \bar{o})\right]^{0.5} \left[\sum_i^n (s_i - \bar{s})\right]^{0.5}} \right]^2 \qquad \text{(Eq. 6)}$$

o_i observed concentration $\quad$ $\bar{o}$ mean of observed concentrations

s_i simulated concentration $\quad$ $\bar{s}$ mean of simulated concentrations

The percent bias measures the goodness of fit of the simulated time series mean with respect to the observed. For N and P concentrations, a value below 25% is considered 'very good', below 40% is considered 'good 'and below 70% is 'satisfactory'. The coefficient of determination measures how the simulated values capture the observed variations with respect to average (a value above 0.5 implies good correlation). Model performance in water quality is often evaluated in monthly time step outputs (Fonseca et al. 2014). Here a 10-day step output is chosen to maximize the time steps since only 2 years of data is available.

3. Results

3.1. Analysis of current WWTP effluents

This subsection assesses the compliance of the observed WWTP effluent data with the quality requirements listed in Table 2.

During the period of analysis (January 2016 to August 2017), TSS concentrations in effluents of all the WWTPs in the study area complied with the requirement of 35 mg TSS L^{-1} set by the UWWTD. Similarly, BOD5 concentrations were consistently below the standard UWWTD requirement of 25 mg O_2 L^{-1} (figure 2).

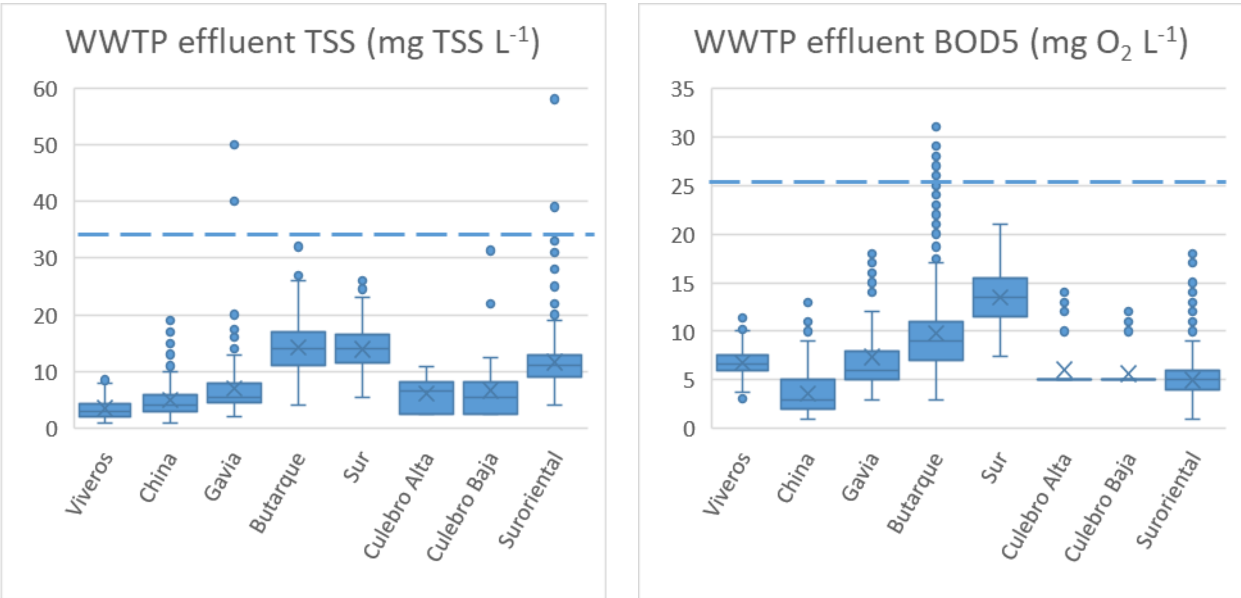


Fig. 2 Total Suspended Solids (TSS) and 5 day Biological Oxygen Demand (BOD5) concentration at effluent of WWTP (dashed horizontal line representing the UWWTD requirement). Second and third quartiles (box), first and fourth quartiles (whiskers), outliers (dots) and average (cross)

Nitrogenous compounds concentration measured in WWTP effluent (figure 3) show two different trends. Some WWTPs (Viveros, Gavia, Culebro Alta, Culebro Baja, and

Suroriental) apply N reduction cycles. Total N concentration in those plants was relatively low (average ranges between 9 and 16 mg N L^{-1}), and most of this N appears in the form of nitrate (80% on average). These results in very low concentrations of ammonium (5% on average). The other group of WWTPs (Butarque and Sur) had high concentration of total N (50 mg N L^{-1}), relatively low concentration of nitrate, and high concentration of ammonium. La China WWTP is a special case. Until May 2017, its effluent had a high concentration of total N and ammonium but since June 2017 a N treatment line is operative and effluent concentration was drastically reduced.

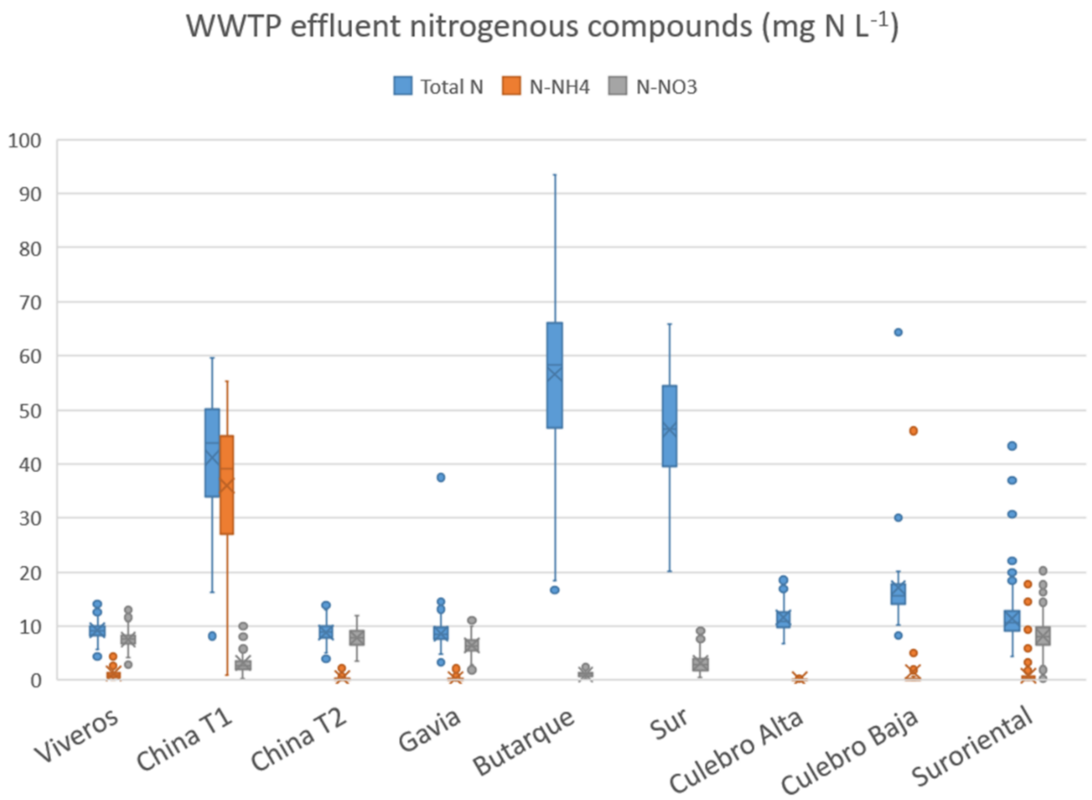


Fig. 3 Nitrogenous compounds concentration in effluent of WWTP. La China WWTP data has been divided in two time series: T1 before June 2017 and T2 after June 2017. Ammonium measurements for Butarque and Sur WWTPs, and nitrate measurements for Culebro Alta and Baja WWTPs are not available

Total P concentration in WWTP effluent (figure 4) shows compliance with UWWTD requirements.

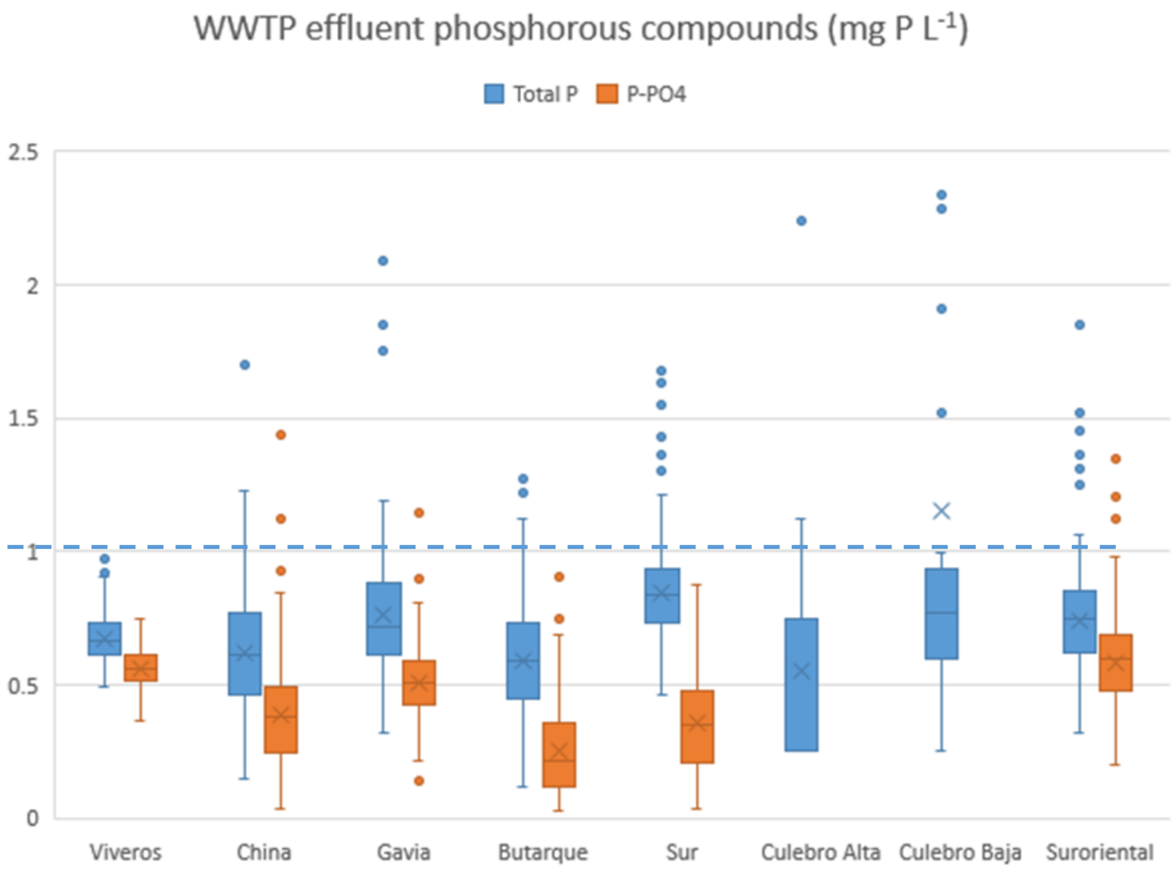


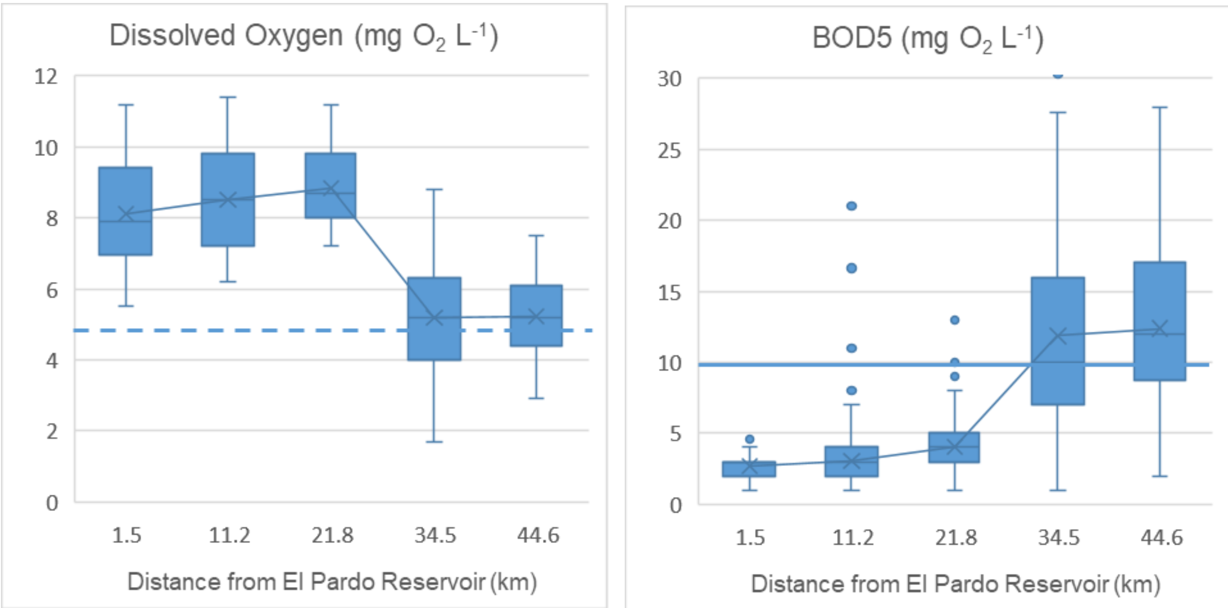
Fig. 4 Phosphorous compounds concentration in effluent of WWTP. Dashed horizontal line represents UWWTD requirement for total P.

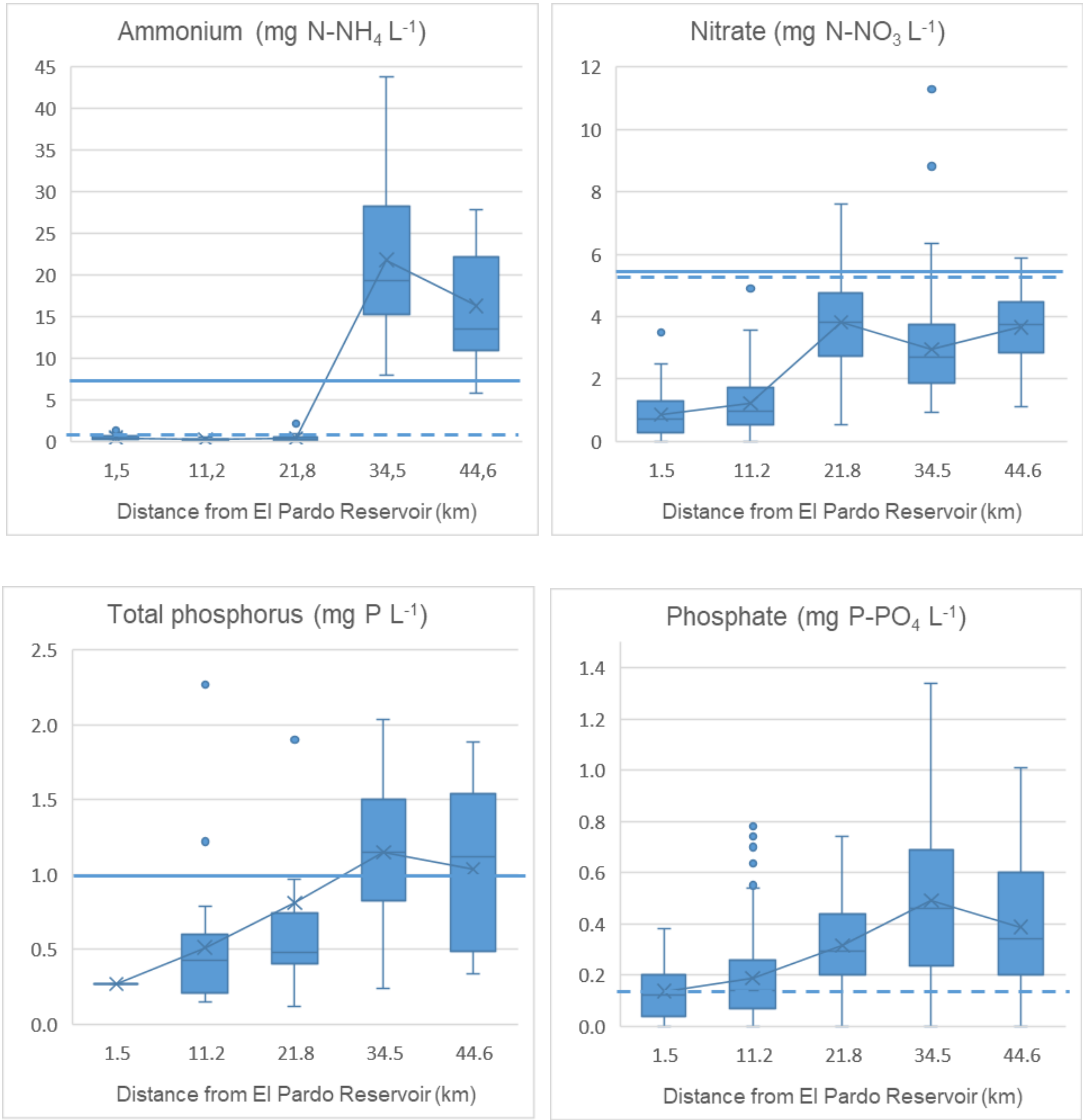
Phosphate measurements for Culebro Alta and Baja WWTPs were not available for this study

3.2. Analysis of current river quality

This subsection assesses the compliance of the observed data in the river with the admissible quality standards in the receiving waters (Table 3).

When analyzing the observed data in the river, figure 5 shows that dissolved oxygen average remains above the 5 mg O_2/l objective for Ecotype RT15 (Table 3). In the lower stretch, the river presents occasional non-compliance. BOD5 values measured in the river exceed the allowed limits (associated to less stringent objectives in Table 3) in the downstream end of the river. Therefore, BOD5 values in the river are non-compliant even if concentrations in the WWTP effluent are less than 60% of the permitted value (figure 2). Ammonium concentration observed in the river downstream of Madrid is above 16 mg N-NH_4 L^{-1}, i.e. twice the less stringent objective, and 30 times the limit associated to good ecological status (Table 3). Nitrate concentrations remain below the 5.6 mg N-NO_3 L^{-1} objective. Total P average concentration is above the less stringent objective by a close margin. Phosphate presents concentrations 1.9 times the good status objective (Table 3). Regarding the pH, the measured values fall within the acceptable range for the good status (pH between 6 and 9).




Ammonium (mg N-NH4 L-1)
45
40
35
30
25
20
15
10
5
0
1,5
11,2
21,8
34,5
44,6
Distance from El Pardo Reservoir (km)
Nitrate (mg N-NO3 L-1)
12
10
8
6
4
2
0
1.5
11.2
21.8
34.5
44.6
Distance from El Pardo Reservoir (km)
Total phosphorus (mg P L-1)
2.5
2.0
1.5
1.0
0.5
0.0
1.5
11.2
21.8
34.5
44.6
Distance from El Pardo Reservoir (km)
Phosphate (mg P-PO4 L-1)
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
1.5
11.2
21.8
34.5
44.6
Distance from El Pardo Reservoir (km)

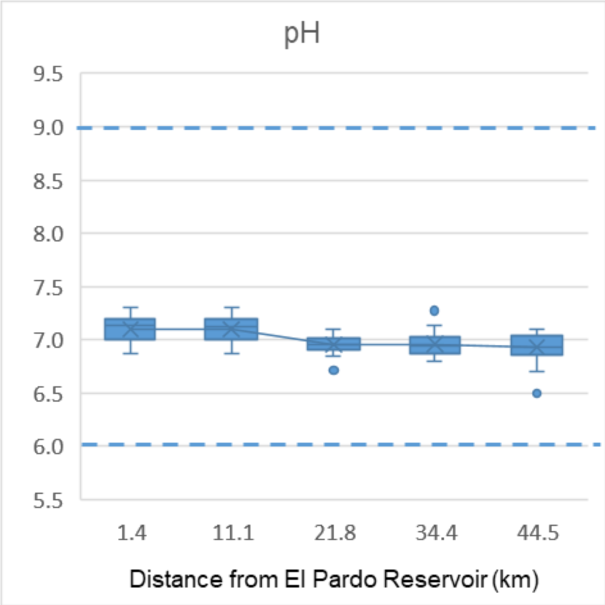


Fig. 5 Observed concentrations in Manzanares River

(environmental objective corresponding to the Manzanares Ecotype is shown in dashed line and less stringent objectives are shown in continuous line)

3.3. Validation of the hydrological model

The calibration and validation of the hydrological model (equation 1) optimizing the Nash-Sutcliffe and PBIAS coefficients (equations 2 and 3) yields the result: U=0.8m^3 s^{-1}, I_0=2.0mm, C=0.6m^3 s^{-1} mm^{-1}. Nash-Sutcliffe coefficient is respectively 0.78 for the calibration period and 0.80 for the validation period, which is considered acceptable, and PBIAS coefficient is 1% in calibration and -7% in validation, considered very good (Moriasi et al. 2007).

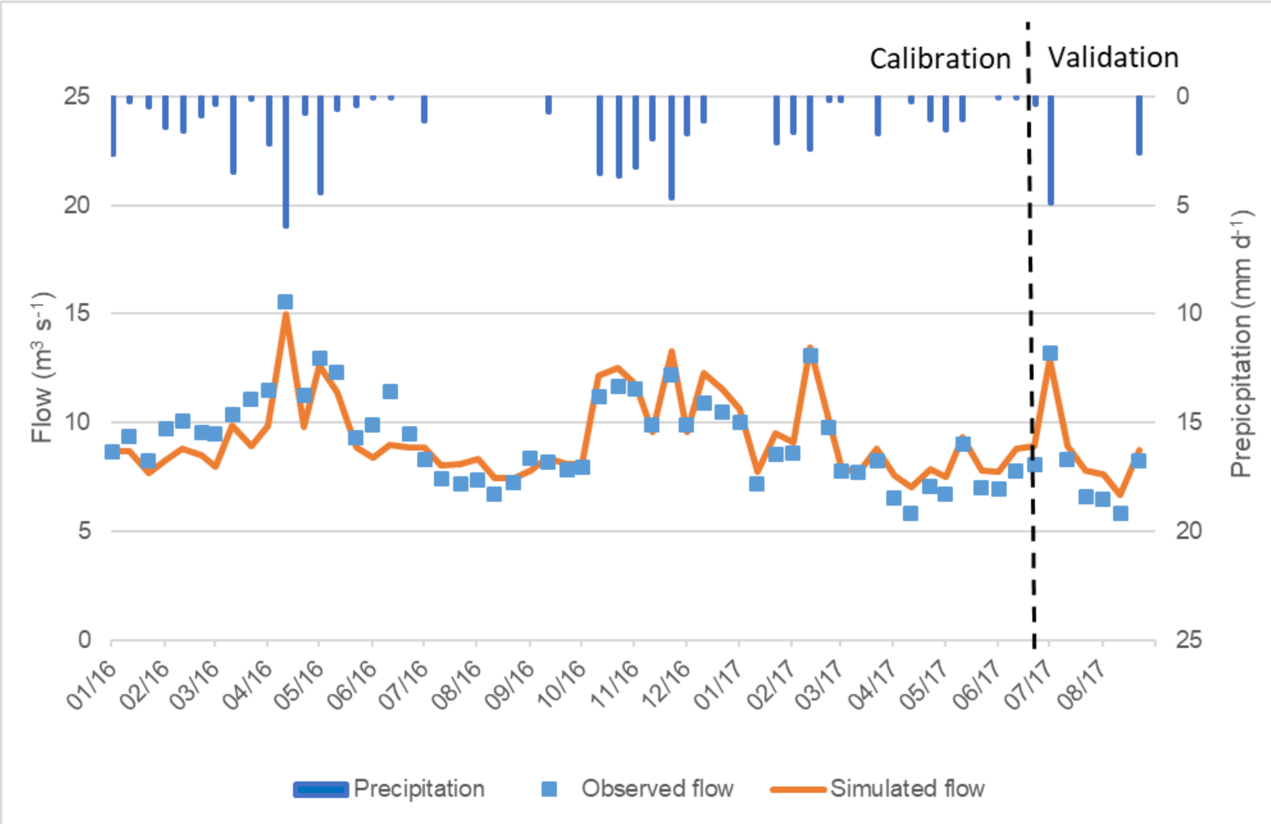


Fig. 6 Observed and simulated flow for Manzanares River at its junction with Jarama River

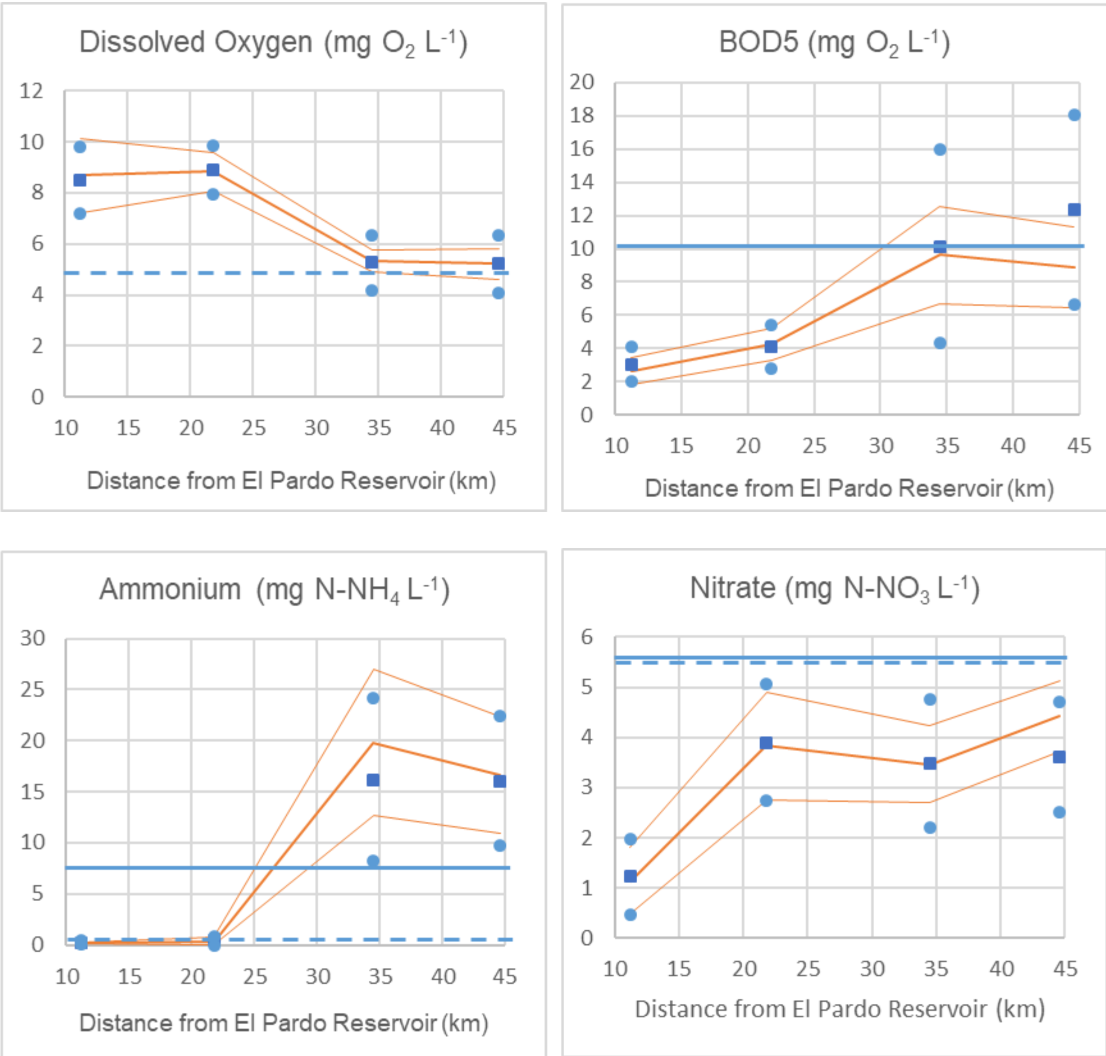
3.4. Validation of the water quality model

The values of the calibrated coefficients as calculated using equations 5 and 6, and presented in Table 4 are consistent with previous literature (Paredes-Arquiola, Andreu, and Solera 2010) and show very low reaction constants for all the processes modeled except for the reaeration (atmosphere to water oxygen transfer) and nitrification (ammonium to nitrate reaction) constants.

River stretch, Distance from Pardo Reservoir (km)	0 to 11.2	11.2 to 21.8	21.8 to 24.9	24.9 to 37.1	37.1 to 44.9
Reaeration coef. Ka (d^{-1})	1.8	4.5	4.5	0.3	3
Nitrification coef. Knitr (d^{-1})	2.6	2.6	2.6	0.01	0.2

Table 4 Calibrated coefficients

Calibration and validation data (Table 5) indicates that the percentage bias simulated values are good except for three total P values that are only satisfactory. With respect to the coefficient of determination, relatively low values for all the pollutants except for ammonium imply that the model can predict average values (figure 7) with acceptable accuracy but has limited capacity to predict temporal variations for these contaminants.



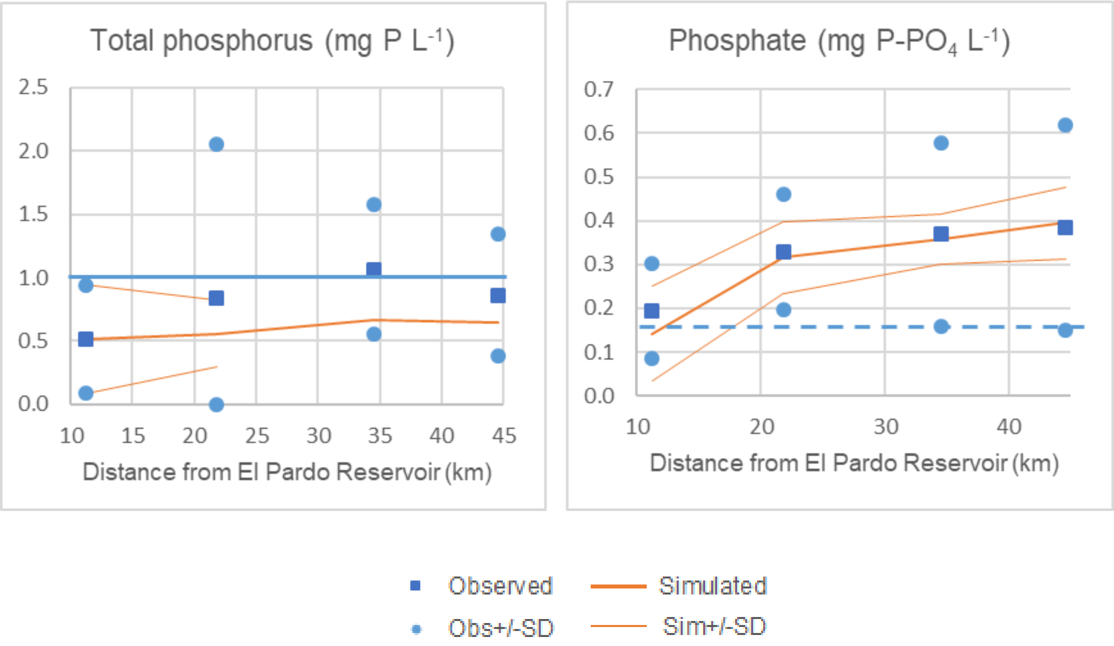


Fig. 7 Observed and Simulated concentrations: average +/- 1 standard deviation at each station (objective corresponding to the Manzanares ecotype is shown in blue dashed line and less stringent objectives are shown in blue continuous line)

		Distance from El Pardo Reservoir (km)			
		11.2	*21.8*	*34.5*	*44.6*
Dissolved oxygen	PBIAS	-2% / -2%	3% / -2%	2% / -6%	0% / 0%
	R^2	0.93 / 0.94	0.73 / 0.76	0.1 / 0.34	0.16 / 0.47
BOD5	PBIAS	8% / 22%	-9% / 3%	2% / 10%	29% / 26%
	R^2	0.16 / 0.08	0.14 / 0.03	0.52 / 0.12	0.34 / 0.11
Ammonium	PBIAS	23% / 16%	45% / 15%	-19% / -28%	-32% / -5%
	R^2	0.03 / 0.45	0.81 / 0.61	0.51 / 0.84	0.39 / 0.75
Nitrate	PBIAS	11% / 1%	3% / 0%	-15% / 18%	-30% / -12%
	R^2	0.53 / 0.83	0.41 / 0.64	0.15 / 0.23	0.13 / 0.04
Total P	PBIAS	0% / -	23% / 42%	49% / -	37% / -45%
	R^2	- / -	0.28 / 0.86	0.8 / -	- / -
Phosphate	PBIAS	32% / 17%	3% / 5%	-18% / 27%	-11% / 8%
	R^2	0.26 / 0.06	0.05 / 0.26	0.12 / 0.05	0.02 / 0.01

Table 5 Model performance indicators, water quality (Calibration/Validation)

BOD5: 5 day Biological Oxygen Demand

With respect to the concentration of pollutants in the precipitation-generated runoff, a significant effect is found in the BOD5. The model shows that river water quality observations in rainy conditions are consistent with a BOD5 value of 50 mg O_2 L^{-1} for runoff water. As a reference, this concentration is twice the maximum value allowed in the WWTP effluent. This highlights the importance of managing urban storm waters to contain river pollution (Lastra 2017).

Several amelioration scenarios are presented in the discussion chapter.

4. Discussion

4.1. Analysis of calibrated coefficients

The value of the calibrated coefficients show that for all the contaminants, the effect of dispersion is found to be negligible with respect to transport and reactivity.

First, a very low BOD5 degradation constant is consistent with the discharge of advanced WWTPs. Most degradable mass has been treated in the WWTPs and a high percentage of remaining BOD5 may be refractory organic compounds. The model systematically underestimates the BOD5 present in the stream, although the percentage bias remains acceptable and the difference is within one standard deviation of the observed data. This mismatch may respond to different laboratory analysis techniques in the measurement of the BOD5. UWWTD explicitly requires for inhibition of nitrification when measuring BOD5 in the WWTP effluent. There is no such explicit requirement in the stream observations.

Secondly, the model shows very limited ammonification of organic N, the most likely cause being that the only remaining organic N is strongly bonded to the organic matter (after the advanced treatment in the WWTP). The values of the nitrification constant are in the high end of the expected range (Bowie et al. 1985), which is consistent with the high concentration of ammonium and oxygen in the stream. This nitrification must be considered when defining the nitrate requirements of the WWTP discharge, since additional nitrate will be formed as the process advances. The model predicts negligible values of denitrification reaction constant in the column of water. Concentration of dissolved oxygen remains high enough to avoid anoxic conditions needed for denitrification.

Thirdly, both the total P and the phosphate measured in the river show negligible reactivity and sedimentation. This makes them behave as a conservative pollutant responding only to mass balances at each discharge point.

The low reactivity of the remaining organic matter detected by the model has strong implications on the WWTP discharge concentrations that are compatible with a good status of the river. High concentrations in the river can only be tackled by reducing their emissions at the source. In the case of nitrate and phosphoric compounds, the low value of the evolution coefficients may indicate that these have precipitated as insoluble compounds in the solids (TTS).

4.2. Scenarios

Once the model is calibrated, different scenarios with changes in WWTP effluents are built. The model predicts the evolution in river pollutant concentration for these scenarios.

The first two scenarios explore the compatibility of the UWWTD effluent concentration requirements (subsection 2.1.1) with the achievement of the good status of the receiving waters. In the third scenario, the model explores what WWTP discharge limits would be needed to achieve the good status.

- Scenario 1 (S1): All the WWTPs are set to discharge the maximum permitted effluent flow and the maximum allowed concentrations for TSS, BOD5, total P and N; and with the current ammonium/nitrate repartition for each WWTP.
- Scenario 2 (S2): All the WWTPs are set to discharge at the maximum permitted effluent flow and the maximum allowed concentrations for TSS, BOD5, total P and N but with nitrifying ammonium/nitrate repartition (5% ammonium, 80% nitrate as calculated in subsection 3.1). In case scenario S1 is not compatible

with a good status due to high ammonium concentration, scenario S2 is built to assess whether the use of the current nitrification technology in all the WWTPs would lead to a good status.

- Scenario 3 (S3): This scenario explores what WWTP discharge concentrations are needed to achieve the physico-chemical conditions in the river associated to a good status. Maximum allowed WWTP discharge flow is used. In this scenario the input concentration of each pollutant in the WWTP effluent is progressively reduced until the simulated concentration in the stream complies with the values associated to good status.

The concentration of pollutants for each scenario is analyzed in the following sub-sections in order to draw conclusions about the behavior of the system. Some caveats are needed in the interpretation of these results. First and utmost, the model accuracy relies on the density of the data, which is limited by the temporal and spatial sampling density in the case study. Moreover, our model (paragraph 3.2) is a simplification of the river behavior.

Biological oxygen demand

Scenarios S1 and S2 explore what the BOD5 values in the stream would be if the WWTPs were discharging the maximum concentration allowed by the UWWTD standard values and the discharge permit, i.e. 25 mg O_2 L^{-1} (see Tables 1 and 2). Figure 8 shows that this would entail BOD5 values in the stream above the less stringent objective of 10 mg O_2 L^{-1}. This reveals a discrepancy between the direct implementation of sewage treatment legislation (UWWTD) and environmental legislation (WFD).

Scenario S3 estimates that 11 mg O_2 L^{-1} of BOD5 at plant effluent is the maximum value that meets the limit set by the WFD in the river.

Nitrogenous compounds

Scenario S1 examines all the WWTPs discharging at the maximum concentration allowed of total N (10 mg N L^{-1}), with the current percentage of ammonium and nitrate for each WWTP. Ammonium concentrations above 2.3 mg N-NH_4 L^{-1} are to be found in the stream (figure 8). While this would comply with the less stringent objective of 10 mg N L^{-1}, it is still in excess of the 0.6 mg N-NH_4 L^{-1} required for good ecological status.

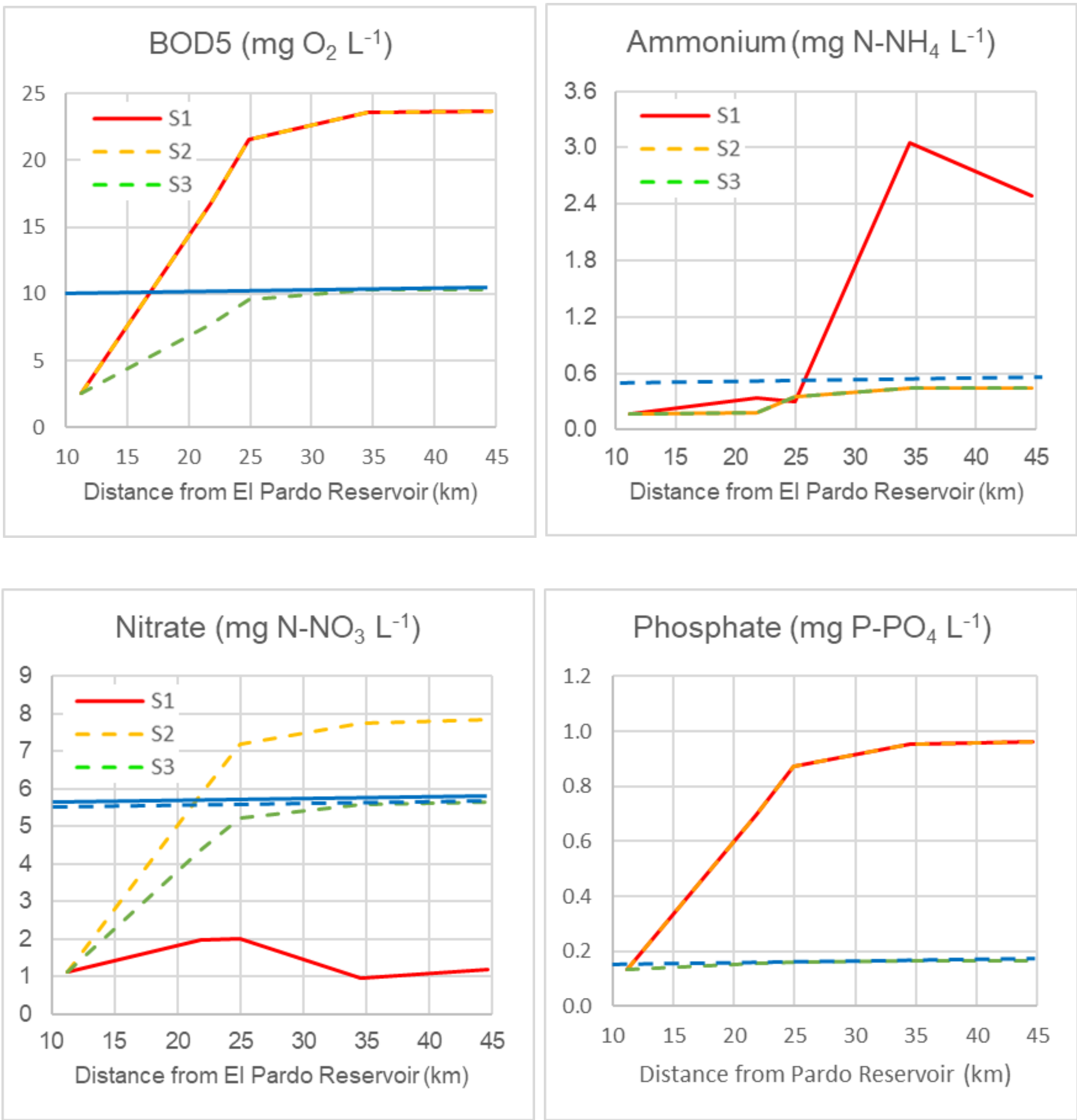


Fig. 8 Scenario output concentrations

Objective corresponding to the ecotype is shown in horizontal blue dashed line and less stringent objectives are shown in blue continuous line

In Scenario S2 all the WWTPs comply with the requirement of 10 mg N L^{-1}/l of total N. Since this scenario requires the installation of a nitrifying line in all WWTPs. Figure 8 shows that the expected ammonium concentrations in the river would be slightly below the 0.5 mg N-NH_4 L-1 limit. However, in this scenario the concentration of N-NO_3 in the river would be 7.7 mg L^{-1} (figure 8), which is above the maximum allowed value of 5.6 mg N-NO_3 L^{-1}. This result suggests that setting a requirement for total N concentration at WWTP effluent does not guarantee the good status of the receiving water body and highlights the need of further harmonization between the WFD and the UWWTD. In order to improve consistency, specific requirements for ammonium and nitrate concentrations should be set at WWTP effluent.

Scenario S3 estimates the concentration of ammonium and nitrate in the WWTP effluent that would be compatible with a good status of the river in terms of nitrogenous compounds. It finds that a concentration of 0.51 mg N-NH_4 L^{-1} and 5.9 mg N-NO_3 L^{-1} at WWTP discharge would be needed. Discharge permit should be modified accordingly.

With the current nitrogenous compound distribution of the nitrifying WWTPs effluents (and considering the contribution of organic N), scenario S3 implies a maximum total N concentration of 7 mg N L^{-1}. Therefore, it is technically feasible to achieve the good status of nitrogenous indicators through the application of current nitrifying techniques.

Although ionized ammonium (NH_4^+) has low toxicity, unionized ammonia (NH_3) presents high levels of toxicity to the fauna in the stream. Ammonium and ammonia cohabitate in water dissolution in an acid-base equilibrium resulting in negligible ammonia concentration for a pH below 7.5 (Dojlido and Best 1993). Current pH measured in the river is below 7.3 (figure 5), but pH concentration allowed in the river is 6 to 9. Thus, evidence suggests that pH values below 7.5 in the river and in WWTP discharge should be prescribed at least until the level of ammonium in the river is effectively reduced.

Phosphate

Scenario S3 foresees that a phosphate concentration of 0.17 mg P-PO_4 L^{-1} in WWTP effluents would be needed to achieve good status in the stream. The model shows that WWTP effluent represents 91% of the total water content downstream. Therefore, phosphate concentration in the receiving waters will be very close to the concentration in the discharge. The phosphate concentration identified by S3 represents 36% of the current average phosphate discharge (0.46 mg P-PO_4 L^{-1}). This points to the need for a leap of improvement in the performance of WWTPs in the study area. A study with a broader geographical scope is needed to assess the implications of the current high concentrations of phosphate downstream. Such high concentrations increase the risk of eutrophication in lakes and reservoirs especially in the warmest months of the year and can be reduced through costly treatment processes (Ostace et al. 2013; Genkai-Kato and Carpenter 2005). In order to optimize operation costs in the WWTPs, it could be advisable to set seasonal requirements to the phosphate concentrations in WWTP effluents.

Complementary research is needed to analyze the economic impact of the proposed measures. If the costs of the required measures were found to be disproportionate, WFD allows the use of exemptions to the environmental objectives. Since the effect of Madrid WWTP effluents may affect bodies of water downstream of Manzanares River, further investigation is also needed to analyze the effect on a broader geographical scale.

The results are in line with findings described in the extant literature. The prominence of ammonium as a major contaminant in urban rivers is in consonance with Paul and Meyer (2001) results, and the appeal to technological solutions through normative requirements agrees with Duh et al. (2008) considerations. The length of the water stream impaired by the urban pollution in the case of Manzanares is above other cases documented in the literature. Even et al. (2007) report a length of 50 km of low quality waters in the Seine river through Paris and downstream, a similar distance is reported for London and the Thames (Cox and Whitehead 2009). In our study area, the effect of Madrid WWTP effluent is detected 100 km downstream of the city (CHT 2018d; Gallego Bernad and Sánchez Pérez 2006). This effect underpins the relevance of the imbalance between the volume of the wastewaters and the low natural flow of the receiving rivers, highlighting the need for stricter limits for WWTP effluents.

5. Conclusions

Our study highlights the need to adapt WWTP discharge permits to the ultimate objective of good status in the receiving waters. In the context of the EU, it also underlines the need for a stronger integration between the sewage treatment legislation (UWWTD) and environmental legislation (WFD), which perhaps warrants reform of UE legislation. Science-based mechanisms to define maximum concentrations at WWTP effluent need to be developed, and the publication of further case studies will support this effort.

In the specific case of the Manzanares River and the treated wastewaters of the city of Madrid, current BOD5 requirement (25 mg O_2 L^{-1}) for plant effluent is not compatible with the BOD objective for receiving waters. A more stringent requirement of 11 mg O_2 L^{-1} of BOD5 is proposed for plant effluents. Current ammonium levels in the river are above good status limits and less stringent objectives. A WWTP ammonium requirement of 0.51 mg $N\text{-}NH_4$ L^{-1} is proposed instead. Nitrate requirement of 5.9 mg $N\text{-}NO_3$ L^{-1} at plant effluent is also recommended. Finally, a limit of 0.17 mg $P\text{-}PO_4$ L^{-1} of phosphate in WWTP effluents is needed to achieve the good status in the river. Discharge permits should align to these values. Usual caveats on model assumptions apply. The need for the wastewater treatment infrastructure improvements that can achieve these effluent concentrations is highlighted not only by the current high pollutant concentrations in the Manzanares river, but also by the tens of kilometers of impaired waters in the rivers downstream.

Infrastructure adaptation of WWTPs to comply with these limits may be gradually phased in. After each phase, model assumptions should be reviewed and TRBA should adapt WWTP permits accordingly.

Conflict of Interest: The authors declare that they have no conflict of interest.

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6. Modelización de la calidad físico-química del Tajo Medio

6.1. Introducción

El artículo presentado en este capítulo se titula “Designing river water quality policy interventions with scarce data: the case of the Middle Tagus Basin, Spain” y ha sido publicado en la revista Hydrological Sciences Journal. Trata sobre la evolución de los contaminantes físico-químicos en un área ampliada que engloba los cursos bajos de los ríos Manzanares, Jarama, Henares, y del río Tajo entre las ciudades de Aranjuez y Toledo (Figura 5). En total se estudian 12 masas de agua con la siguiente denominación en el plan demarcación vigente: Henares desde Arroyo del Sotillo hasta el Torote; Henares desde el Torote hasta el Jarama; Manzanares desde el Pardo hasta la Trofa; Manzanares a su paso por Madrid; Arroyo Culebro; Jarama desde el Guadalix hasta Valdebebas; Jarama desde Valdebebas hasta el Henares; Jarama desde el Henares hasta el Embalse del Rey; Embalse del Rey; Jarama desde el Embalse del Rey hasta el Tajuña; Jarama desde el Tajuña hasta el Tajo; y Río Tajo desde el Jarama hasta Toledo. El área de estudio se define de esta manera para englobar el efecto de las grandes depuradoras de la región de Madrid, así como el de las detracciones del Acueducto Tajo-Segura.

6.2. Metodología

La metodología aplicada es similar a la expuesta en el capítulo anterior, añadiendo el efecto de la contaminación difusa, principalmente por agricultura de regadío en las vegas del Henares y del Jarama.

Debido a la escasez de datos, que dificulta una calibración clásica, se desarrolla un método de calibración propio que adapta el valor de los coeficientes para minimizar las diferencias entre los valores simulados y los observados teniendo en cuenta sus propiedades estadísticas.

Una vez calibrado el modelo, se construyen escenarios para identificar los cambios en las presiones (vertido de contaminantes, abstracción de aguas en cabecera) requeridos para alcanzar el buen estado de las aguas receptoras.

6.3. Resultados y discusión

El método de calibración de los coeficientes del modelo queda validado por el análisis estadístico de comparación entre las simulaciones y las observaciones disponibles.

El modelo arroja varios resultados destacables. En primer lugar, identifica las depuradoras de las grandes poblaciones de Madrid como la principal causa de contaminación físico-química del Tajo Medio, siendo la aportación de contaminantes por otras fuentes como la contaminación difusa de un orden de magnitud menor. El estudio calcula las concentraciones de contaminantes en el efluente de cada depuradora de la zona de estudio que permitirían alcanzar los objetivos medioambientales (Tabla 6).

Depuradoras vertiendo al:	*NH_4 (mg/l)*	*NO_3 (mg/l)*	*PO_4 (mg/l)*
Henares	4.00	60	0.65
Manzanares	0.65	30	0.55
Jarama aguas arriba de la confluencia del Henares	1.00	50	0.55
Jarama aguas abajo de la confluencia del Henares	8.00	60	1.00

Tabla 6. Concentraciones de contaminantes en efluente de depuradora compatibles con el buen estado de las aguas receptoras

Por otro lado, se cuantifica el efecto del Trasvase Tajo-Segura y se constata que ni siquiera con un volumen trasvasado nulo se alcanzaría el buen estado de las aguas en la masa de agua Tajo entre la confluencia del Jarama y Toledo.

Respecto a los escenarios de mejora, se caracteriza el estado de las masas de agua tras la implementación de cada medida de mitigación, y se concluye que se necesita al menos incluir un ciclo de nitrificación-desnitrificación en cinco de las grandes depuradoras objeto de estudio para alcanzar el buen estado en el eje principal del Tajo. El estudio también fija una prioridad a las actuaciones, proponiendo actuar en cada fase sobre la masa de agua con mayor brecha respecto a sus objetivos medioambientales. Con ello se define una lista ordenada de depuradoras sobre las que habría que actuar para conseguir los objetivos medioambientales.

6.4. Conclusiones

El estudio apunta a la modernización de infraestructuras de depuración como una medida ineludible para alcanzar el buen estado de las aguas superficiales del Tajo Medio. El impacto sobre la calidad de las aguas de otras presiones que actúan sobre el sistema (contaminación difusa y detracciones del Acueducto Tajo-Segura) es sensiblemente menor, y cualquier medida de mitigación que actuara sobre ellas no podría por sí sola producir la consecución de los objetivos medioambientales. Al igual que en el capítulo anterior, debe resaltarse que en el plan hidrológico de demarcación vigente se aplican unas exenciones que flexibilizan los objetivos medioambientales.

Las implicaciones financieras de dichas medidas y su posible intervención en la flexibilización de los objetivos medioambientales obligan a realizar un estudio específico en el que se comparen los costes de las medidas con los beneficios que aportan. Esta caracterización es el objeto de los dos siguientes capítulos.

6.5. Artículo

A continuación se reproduce, con el permiso de los coautores, el contenido del artículo:

Antonio Bolinches, Lucia De Stefano, Javier Paredes-Arquiola (2020) Designing river water quality policy interventions with scarce data: the case of the Middle Tagus Basin, Spain, Hydrological Sciences Journal, DOI: 10.1080/02626667.2019.1708915

Designing river water quality policy interventions with scarce data: the case of the Middle Tagus Basin, Spain

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Abstract

Anthropic pressures deteriorate river water quality, so authorities need to identify their causes and define corrective actions. Physically based water quality models are a useful tool for addressing physicochemical pollutants, but they must be calibrated with an amount of data that is often unavailable. In this study, we explore the characterization of a model to design corrective interventions in a context of sparse data. A calibration indicator that is both simple and flexible is proposed. This approach is applied to the Middle Tagus Basin in central Spain, where the physicochemical concentration of pollutants is above legal standards. We quantify the effects of the main existing pressures (discharge from wastewater treatment plants, agricultural diffuse pollution and a major inter-basin water transfer) on the receiving waters. In particular, the study finds that wastewater treatment plant effluent concentrations should be reduced to up to 0.65 mg/L of ammonium and 0.55 mg/L of phosphate to achieve the environmental goals. We propose and prioritize a set of policy actions that would contribute to the good status of surface water bodies in the region.

Keywords: Water Framework Directive; water quality; Tagus Basin; data scarcity; modelling

1. Introduction

River banks have been a preferential area for human settlement since the early civilizations (Macklin and Lewin 2015). Suitable conditions such as water availability, land fertility due to nutrientrich floods and ease of transport of goods (Vega et al. 1998, Di Baldassarre et al. 2013) have attracted high population densities in the vicinities of rivers. They have also entailed an increase in river pollution (Ward and Elliot 1995) and, as a result, continental water quality is worsening globally (Allaoui et al. 2015).

In places where deteriorating water quality threatens ecosystem sustainability, water authorities need to identify the causes and prescribe corrective actions. This is often defined through water quality models. There is a growing body of scholarly literature on the modelling of the ecological status of rivers and the effect of pressures caused by polluting agents (Thomann and Mueller 1987, Genkai-Kato and Carpenter 2005, Chapra 2008, Momblanch et al. 2015, Dodds and Smith 2016, Shrestha et al. 2016). Among other elements (biological and hydromorphological), physicochemical indicators are used to describe the concentrations of oxygen and nutrients that are compatible with the long-term sustainability of freshwater ecosystems. In the case of continental surface waters, the main pressures on water quality are present in the form of the urban wastewater, industrial pollution and nutrient-rich agricultural fertilizers. While point source pressures are easier to locate and quantify through direct measures at the discharge, diffuse pollution characterization poses complex challenges (Strömqvist et al. 2012, Epelde et al. 2015), leading to data demanding models, or qualitative output studies (Munafò et al. 2005, Zhang and Huang 2011).

Physically based models use water quality observations to understand the behaviour of river stretches (Fonseca et al. 2014, Keupers and Willems 2017, Hutchins and Bowes 2018). Typically, observations on a regular basis (daily, weekly) of flow and pollutant concentration for rivers and pollutant discharges have been used to characterize the model through performance indicators. For a given study area, these indicators compare river observations downstream with model output (simulated from river observations upstream and pollutant discharges along the study area). However, the available observations are often too sparse to allow such approach, and there is limited research investigating how to calibrate the models when data is not dense enough to correlate pressures and observations on similar dates. The first approach is to move away from physically based models and describe the process with empirically based regressions tailored for small sample sizes (Cohn et al. 1989). Due to the difficulty of interpreting the parameters in physically meaningful terms, other authors combine the two approaches and start from mechanistic models to which statistical methods are applied (Romanowicz et al. 2004, Wellen et al. 2014). Some scholars maintain the existing indicators, widening the data scale (monthly instead of daily) to adapt to the existing data (Tarawneh et al. 2016), while others combine this approach with annual mass balances (Zhao et al. 2010).

In this paper, we explore options to offer science-based support to water management decisions when the data observations are scarce. We propose a calibration technique that does not require a pair-wise comparison between pressures and receiving water status. This is achieved through the development of a goal function that exploits the statistical properties of the existing data.

The approach is applied to the Middle Tagus Basin, Spain, where surface waters fail to comply with the established quality standards. Adapting the methodology to the few observations available (four water quality observations per year), the study quantifies the effect of the existing pressures on river water quality. It also defines the infrastructure changes and the management decisions that are required to achieve or maintain the good status of the surface waters.

Understanding river water quality dynamics in this region is especially interesting for several reasons. Firstly, this area receives the wastewater of a highly populated urban region – the metropolitan region of Madrid – and the diffuse pollution from fertilized irrigated land, and has relatively low-flow rivers with limited capacity to dilute pollution.

Wastewater treatment plants (WWTP) in the study area present a high efficiency in biological oxygen demand reduction but have an uneven record in nutrient reduction. The study thus explores the degree of additional nutrient reduction required to attain the environmental improvements. Secondly, a large interbasin water transfer (Tagus Segura Water Transfer) diverts a half of Tagus headstream waters to the Southeast of Spain. Currently, the quantity of water to be transferred is decided on a monthly basis, depending on the stored water volumes in the major reservoirs of the Tagus headwaters (MAPAMA 2014). The water transfer limits the capacity of Tagus River to dilute the pollution produced by the Madrid region. Previous studies have sought to understand the impact of the transfer (Morales Gil et al. 2005, San Martín 2011, PellicerMartínez and Martínez-Paz 2018) on water availability in the donor's region but have not assessed its implications on water quality. A previous water quality study in the region of Madrid (Cubillo et al. 1992) has been outdated because of wastewater treatment infrastructure upgrades. It does not include the effect of the interbasin water transfer and does not issue policy recommendations on the required quality of WWTP effluents. A more recent study (Paredes et al. 2010) prescribes maximum pollutant concentration for the effluents but is restricted geographically to a subregion (the Manzanares River).

2. Materials

2.1 Study area

The study area includes several river stretches of the Middle Tagus Basin, including the lower course of the Jarama and Henares rivers, the Manzanares River from upstream the city of Madrid and the Tagus River from its confluence with the Jarama at Aranjuez (Fig. 1, T.0) to the city of Toledo (T.70).

The study area has a Mediterranean climate with warm and dry summers, rated Csa in the Köppen-Geiger classification (Kottek et al. 2006). Average monthly temperatures range from 2.7 to 32.1ºC (AEMET 2018). Rainfall distribution is highly influenced by the presence of the Central System Mountain Range to the north of the study area (Durán Montejano 2016). The orographic effect implies that yearly precipitation varies from 1500 mm/year in the northern highlands, to below 400 mm/year in the midlands between Aranjuez and Toledo. Most water runoff is therefore generated in the headwaters. Precipitation shows also a strong seasonality, with wet winters (60 mm/month on average in the city of Madrid) and dry summers (10 mm/month). This seasonality has a major impact on the quantity and quality of circulating surface waters, and on agricultural irrigation practices.

Madrid urban area hosts more than 5.1 million people (INE 2018). Urban water demand amounts to 32 hm^3/month (CHT 2015a) and is largely met by reservoirs in the Upper Jarama River (upstream of the area of study). Fifteen major WWTPs treat and discharge wastewater produced by domestic and industrial uses (Table 1). Total authorized discharge of WWTPs is 37 hm^3/month, although normal operation flow is below this volume (CHT 2015b). Water withdrawal for irrigation amounts to 16 hm^3/year (CHT 2015a).

Groundwater and surface water bodies receive the surplus of nutrients of the fertilizers applied to 200,000 ha of agricultural land in the region (DGA 2017).

The average flow of Jarama River in its lower stretch ranges between 20 hm^3/month in summer and 122 hm^3/month in winter (MAPAMA 2018), and average flow in the Middle Tagus River ranges between 51 and 159 hm^3/month.

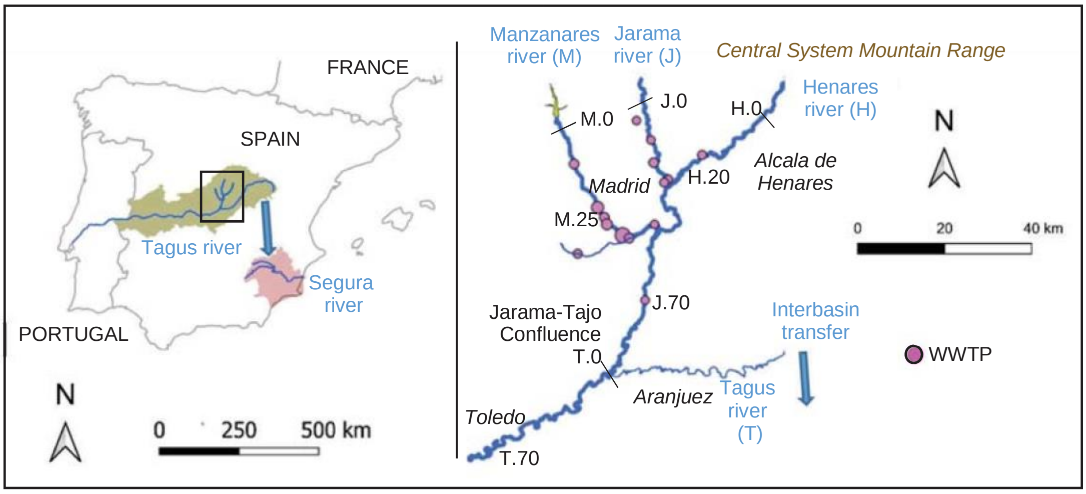

Figure 1. Area of study. Arrow shows Tagus Segura water transfer. Positions (e.g. M.0) are labelled as "river code.kilometric distance from reference starting point".

Table 1. Position (river code.distance from reference point) and average discharge of major WWTPs.

Position	WWTP	Discharging to	Average discharge (hm^3/month)
H.17	Alcala Este	Henares	0.3
H.21	Alcala Oeste		1.3
M.12	Viveros	Manzanares	1.7
M.24	China		2.6
M.27	Gavia		2.5
M.29	Butarque		3.2
M.34	Sur		7.3
M.37	A. Culebro		2.6
M.43	Suroeste		0.7
J.17	Valdebebas	Jarama upstream Henares confluence	0.7
J.24	Rejas		1.5
J.29	Torrejon		0.7
J.30	Casa Quemada		1.2
J.40	Velilla S. Antonio	Jarama downstream Henares confluence	0.2
J.68	Soto Gutierrez		0.5
		Total	27.2

Upstream of its confluence with the Jarama River, Tagus River is subject to a major interbasin water transfer that diverts an average of 30 hm^3/month to the southeast of the Iberian Peninsula (Fig. 1).

2.2 Data collection

The data used in this study is collected from the following sources:

- River flow data from gauging stations (MAPAMA 2018): water flow data measured in 19 stations on a daily basis, available until 2015.
- Pollutant concentration from river water quality stations (CHT 2018): concentration of dissolved oxygen (DO), biological oxygen demand (BOD_5), ammonium (NH_4), nitrate (NO_3) and phosphate (PO_4) in the surface bodies of water, measured in 30 stations every 90 days, on average, from 2003 to 2017.
- WWTP effluent pollutant concentration from Tagus River Basin Authority: concentration of biological oxygen demand, ammonium, total nitrogen and total phosphorus from the WWTPs in the study area, measured every 60 days on average from 2009 to 2017.
- Digital elevation model DEM (IGN 2018) with a spatial resolution of 25 m.
- Rainfall (AEMET 2018). Daily precipitation for eight meteorological stations in the study area, series starting before year 2000.
- Nitrogen surplus: Nitrogen applied to the agricultural lands that is not taken by the crops. Yearly average from 2000 to 2013 per autonomous community (administrative region), reported by the authorities (DGA 2017), in accordance with the Directive 91/676/EEC.

In view of data availability, the 2009–2015 period is selected for being the most recent time span with a complete set of information for all the required variables.
The river flow data allows a description of the quantity of water in the system on a daily basis. The geographical distribution of the river quality stations is acceptable (with an average distance of 10 km between stations), but the number of observations per year does not allow a detailed characterization of the temporal distribution of the pollutant concentration. Only the annual average for agricultural pressures, six observations per year for WWTP effluents and four observations per year for river quality are available. This does not allow the definition of a continuous series of events where a particular observation of the concentration of pollutants in the river can be correlated to a particular observation of the pressures.

2.3 WWTP and diffuse pollution pressures

WWTP effluents are subject to Spanish regulations derived by the Urban Waste Water Treatment Directive (UWWTD) 91/ 271/EEC (Council of the European Communities 1991). In particular, effluent from WWTPs treating wastewater from over 100,000 equivalent inhabitants must have concentrations of total suspended solids (TSS) below 35 mg/L and a BOD_5 below 25 mg/L. Since the study area is declared as a catchment of an area sensitive to eutrophication by phosphorus, there is an additional limit of 1 mg/L for total phosphorus (Pt). By contrast, the area is not declared a catchment of an area sensitive to eutrophication by nitrogen, and there is no upper limit to nitrogenous compounds concentration by default. Figure 2 shows that virtually all WWTPs comply with these default limits (excess Pt is present in three smaller WWTPs, with a limited effect on receiving waters).

With respect to diffuse pollution coming from agricultural sources, only nitrogenous compounds are regulated. Unlike the UWWTD, Directive 91/676/CEE (concerning the protection of waters against pollution caused by nitrates from agricultural sources) does not impose upper limits to nitrate pollution. It only prescribes the monitoring of applied fertilizers, crop uptake, and nitrogen surplus. No specific calculation is required by the legislation to sort out which fraction of the nitrogen surplus is washed out in superficial runoff, and which fraction is leached to the aquifer. According to official reports (DGA 2017), total nitrogen surplus in the study area ranges between 8 and 28 kg/ha per year for the period 2009–2013. A fraction of this surplus will reach the surface waters through runoff. In the case of phosphorus, there is no equivalent to the agricultural nitrates directive to quantify the surplus generated by agriculture.

2.4 River water quality objectives and current situation

The Water Framework Directive (WFD) 2000/60/EC (European Parliament and Council 2000) describes the conditions to be fulfilled by surface water bodies to guarantee the sustainability of their ecosystems, i.e. to reach a good status. Figure 3 shows the observed (average +/− standard deviation) concentration of nutrients compared to the maximum limit established according to the WFD for Jarama River and Tagus River between Aranjuez and Toledo. Despite overall compliance of WWTP effluents to standard limits, there are pollutant concentrations in the receiving waters that are above the thresholds compatible with a good status. This points to the need for more stringent constraints on the polluting pressures.

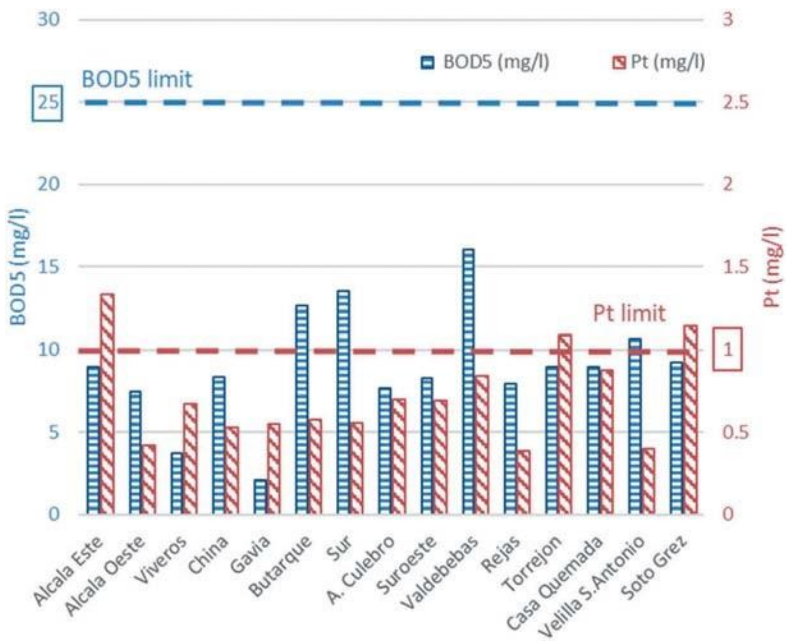

Figure 2. WWTP effluent concentrations (2009–2015 average). BOD5: biological oxygen demand, Pt: total phosphorus.

Nitrate concentration (Fig. 3) grows steadily along the Jarama River as it receives the WWTP effluents, diffuse pollution and the waters of the Henares and Manzanares rivers, then it decreases at the confluence with Tagus headwaters. The average concentration remains under the 25 mg/L limit for all the surface waters except for a minor noncompliance at the lower Jarama.

Ammonium concentration along the Jarama River is prone to substantial changes depending on the nature of the point loads. From a low concentration in the headwaters, effluents from non-nitrifying WWTPs in the upper middle Jarama force an upwards spike. Relatively low concentrations from Henares River contribute to water down these high values (at J.33 position, i.e. Jarama River, 33 km downstream of the reference point), but high ammonium loads from WWTPs in Manzanares River drive up the values at its confluence with the Jarama (J.53). Ammonium and phosphate concentrations remain above the limits for most of the river stretches in the study area.

3. Methodology

A water quality model is built to simulate the concentration of pollutants in the study area. Details are given in the Supplementary material. The model boundaries include the Jarama, Henares, Manzanares and Tagus river stretches that support the urban pressures. The upper border of the model is pushed upstream to a point where the rivers flow in near to pristine water quality conditions before major pressures are applied. The hydrological model takes account of average contributions from river headwaters measured in gauging stations at the upstream frontier, as well as local diversions, WWTP effluent discharges and contribution from soil runoff.

The water quality model is based on the RREA (Spanish acronym for "Rapid Response to the Ambient State") model approach (Paredes-Arquiola 2018) for transport and fate of pollutants.

Point loads are introduced in the model at the authorized discharge locations of WWTPs. The concentration of pollutants in WWTP effluent is calculated from measurements at plant discharge.

For each river stretch, the corresponding sub-basin – or Exclusive Contribution Area (De Oliveira et al. 2016) – is calculated from the DEM using QGIS software (QGIS Development Team 2018). Since the diffuse load (nitrates and phosphates) for each sub-basin is unknown, the value introduced is calculated in the calibration stage correlating simulated and observed values. Only anionic nitrate mobility is modelled, since cationic ammonium is considered to be more electrically bonded to soil particles (García-Serrano et al. 2009). Since the total nitrogen surplus is known from European Union reports (DGA 2017), we will assume in our study that a particular fraction of this surplus reaches the surface water. In the case of phosphorus, we will assume that anionic phosphates have the highest mobility in the soil, and that a particular load in kg/ha per year is applied in each region. This load is quantified in the calibration paragraph. The nitrogen and phosphate surpluses will be quantified in the calibration section.

The reaction coefficient values (nitrification, denitrification, phosphate decay and reaeration) are also calibrated comparing simulated and observed values.

3.1 Calibration

Model calibration requires the definition of a goal function sensitive to performance (i.e. it penalizes simulated values different from observed values). Then, model coefficients can be modulated to minimize (or maximize) this goal function. The available observations collected by Tagus River Basin Authority (2009–2015) are used for the calibration.

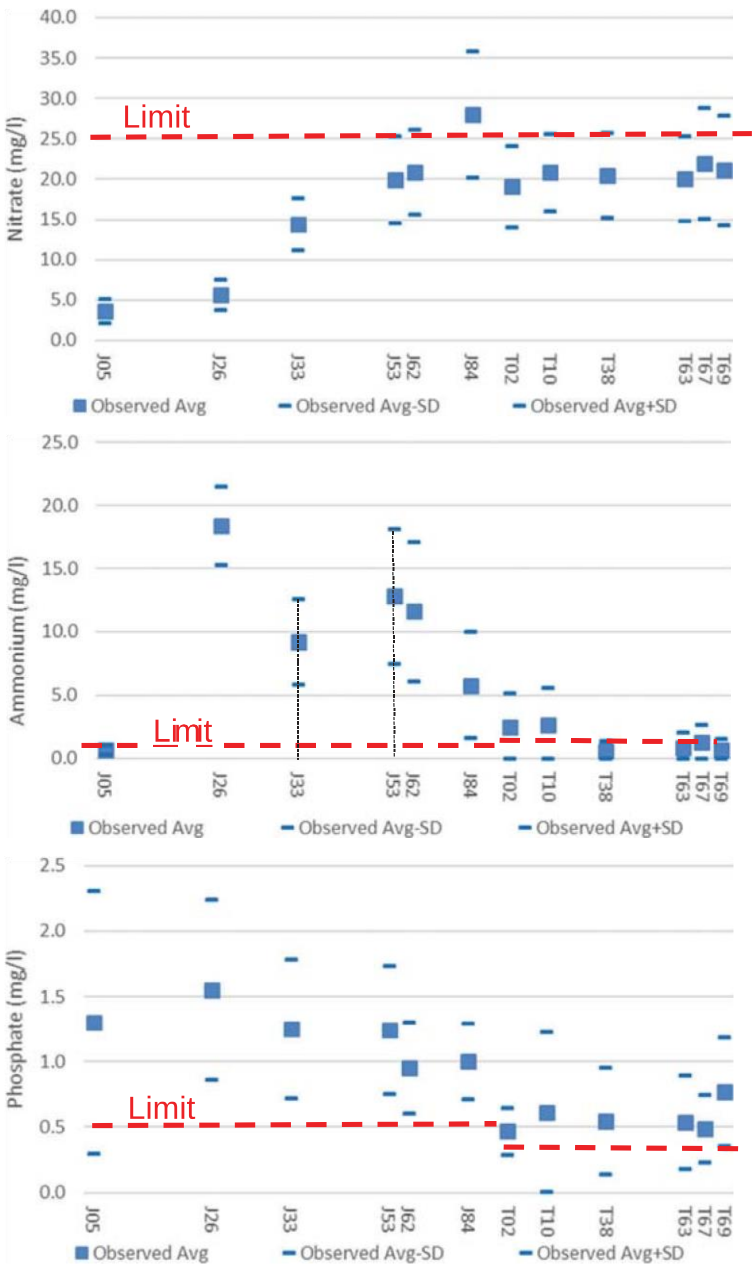

Figure 3. Observed nutrient concentration and limits in legislation along the Jarama (J.05–J.84) and Tagus (T.00–T,70) rivers.

Several approaches are used in the literature to address this task. The coefficient of determination (R^2: the square of Pearson product-moment correlation coefficient) is widely used as a goal function (Donigian 2002), although its flaws are widely acknowledged. Since it describes the degree of collinearity between variables, it is insensitive to linear transformations (translation and proportionality) in the simulated values (Legates and McCabe 1999). At the same time, the square factor makes it oversensitive to outliers, tending to bias the result into extreme events.

Some authors cope with insensitiveness to translation by combining the coefficient of determination with the percent bias (PBIAS) factor (Fonseca et al. 2014, Xue et al. 2015) but insensitivity to proportionality on R^2 remains unchecked.

The Nash-Sutcliffe efficiency criterion (Nash and Sutcliffe 1970) solves the insensitiveness effect and is extensively used in water flow calibrations. It is rarely found in water quality studies due to less extensive data collection and lower accuracy of prediction.

Apart from the above considerations, all these factors are based on a pair-wise comparison: for a given point of the river, we need the observed and simulated value of pollution concentration for the same set of dates. This makes sense for relatively inexpensive observations of precipitation and river flows, which can be easily automated. Water quality observations, however, often require on-purpose field visits for sample collection and laboratory tests. As a result, available water quality data is generally sparser, and studies must adapt to this scarcity (Zou et al. 2014, Elshemy et al. 2016).

Model calibration using solely the PBIAS coefficient does not exploit available information on the variability of observations.

The coefficient is insensitive to the difference between compact, repetitive observations and highly variable and less reliable measurements.

This paper proposes a coefficient to reproduce the observed data as closely as possible. It takes into account the difference between observed and simulated values and the variability of the observed data. The conditions set for the coefficient are:

- It should minimize the distance between observed means and simulated values;
- The effort required to adjust simulated values in order to replicate the observed means should grow progressively with the distance between those values;
- More effort should be directed to replicate observed values with low variability than highly variable observations; and
- It should be simple and easy to calculate with available data.

The proposed formula is:

$$\mathrm{GF} = \frac{[\mathrm{mean(obs)} - \mathrm{sim}]^2}{\mathrm{SD(obs)}} \qquad (1)$$

where obs represents the observed concentrations (measured data); sim refers to the simulated concentrations (model output); and SD is the standard deviation. Equation (1) fulfills all the requirements simultaneously. It is the simplest function that grows progressively with the difference between simulated and average observed values; and for a given difference, the goal function presents higher values for smaller observed variability.

The strong seasonality of precipitation in our study area affects the time distribution of runoff and river flow. Monthly river flow distribution in our study area (Fig. 4) shows that the time period that allows concentration comparisons corresponds to the dry season in summer (June–September). WWTP effluent and river quality measurements are observed at different dates and cannot be correlated in rainy months with highly variable flow. During the rainy months (October– May), the standard deviation of flow is 1.2 times the average flow, meaning that the observation in a particular day is a poor predictor for the observation in another random day. In contrast, relatively constant flows during the dry months (standard deviation of flow is 0.4 times the average flow) allow for comparison of average values.

The dry summer season also corresponds to the critical load case. Nearly constant pollutant load combined with low summer flows (due to lower precipitation and higher agricultural abstractions) bring about maximum pollutant concentration.

The available time series is not considered long enough to allow for a temporal sample split for calibration and validation. In our 7-year data sample, putting aside one-third of the data series for validation (Abbaspour et al. 2015) would depopulate an already exiguous calibration period and provide an insufficient calibration period. Previous literature (Daggupati et al. 2015) admits that comparison data are not always available for robust model calibration and validation, requiring additional analysis of model diagnosis to supplement validation and improve confidence in model performance. The implications will be examined in the discussion section.

4. Results

Once the model is set, the coefficients are calibrated to minimize the goal function described in the methodology.

4.1 Water quality model calibration

The model calibrates the applied diffuse load for each area that best fits the observed data. The results show up to 4.8 kg/ha per year loads for nitrate and up to 3.6 kg/ha per year for phosphate, which is consistent with the values in previous literature (Pieterse et al. 2003, Elrashidi et al. 2004, Grizzetti et al. 2008, Yang and Wang 2010). Figure 5 shows the spatial distribution of the calibrated loads.

The results suggest a more intense nitrogen fertilization in the Henares River and the Upper Middle Jarama, while phosphorus diffuse pollution appears to be more evenly spread.

Evolution coefficients for the chemical reactions in the river are also assessed. Figure 6 shows the comparison between observed and simulated values.

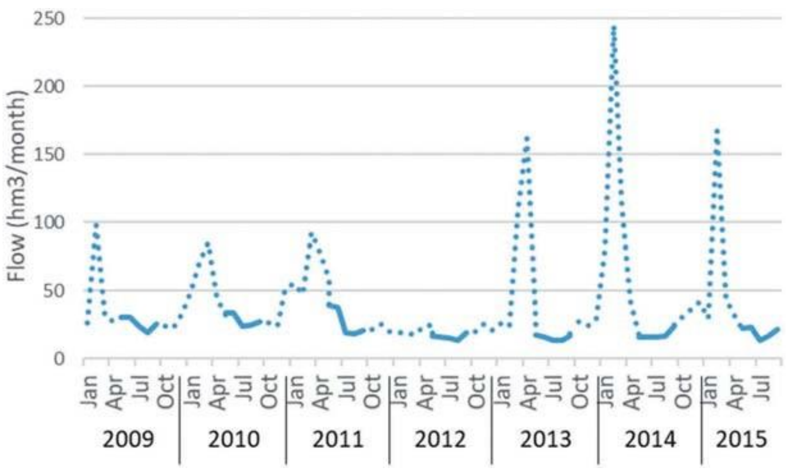

Figure 4. Monthly river flow at gauging station J.33 (Jarama River, 33 km downstream of reference point). Continuous line: June–September (low-flow variability), dotted line: October–May (high-flow variability).

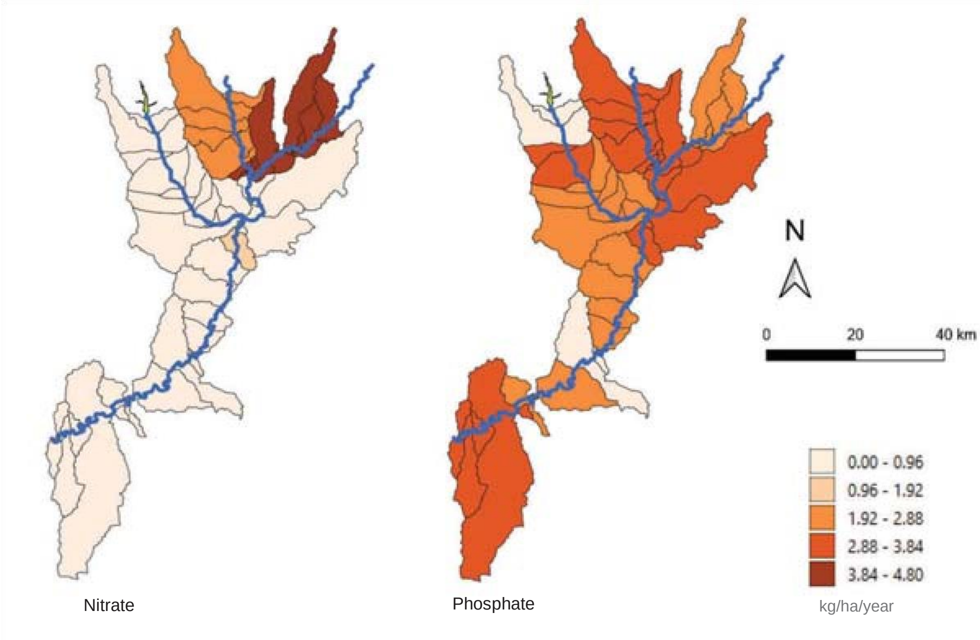

Figure 5. Nitrate and phosphate calibrated diffuse load.

According to Moriasi et al. (2007) for monthly nitrogen and phosphorus predictions, a PBIAS value below 25% is considered "very good"; below 40% is considered "good"; and below 70% is "satisfactory". Table 2 lists the average PBIAS value for each compound, showing a very good fit for phosphates and nitrates, and an acceptable fit for ammonium. In the absence of validation, Daggupati et al. (2015) recommend to supplement the analysis with a graphical and statistical comparison of model responses at multiple locations. Figure 6 shows that the simulated values follow adequately the observed values when their statistical variation is taken into account. Further confidence on model performance can be gained with scatterplots (Fig. 7), as recommended by Moriasi et al. (2015) for short periods and coarse temporal resolution of available data.

4.2 Relative weight of applied pressures

With this information, we can now calculate the relative weight of each pressure on the receiving waters (Fig. 8). The results show that in the Middle Tagus 68% of the nitrates correspond to direct nitrate discharge from WWTPs, 31% correspond to nitrified ammonium from WWTPs and the remaining 1% corresponds to nitrate from diffuse pollution. In the case of phosphate, 84% corresponds to WWTP effluent and the remaining 16% to diffuse pollution.

4.3 Scenarios

We now focus on policy actions needed to achieve a good status in the rivers of the study area.

Since WWTPs are the major contributors to physico chemical pollution, policy intervention should focus on the adaptation of discharge permits. Previous literature on wastewater management optimization (Wang and Jamieson 2002, Zeferino et al. 2017) focuses on site selection and load allocation for new infrastructure. In our case, the site and volume allocation of existing WWTPs will be respected to avoid changes in sewage conduction network. Using the calibrated model, pollutant concentration loads are reduced until the receiving waters achieve the required physicochemical standards. The resulting limits are listed in Table 3.

A second scenario is built to quantify the effect of the Tagus Segura water transfer on the quality of water with the current WWTP effluent properties. More water diversion from Tagus headwaters (Fig. 1) means that less water is available in the Tagus River to dilute the polluted flows from the Jarama River, resulting in a worse water quality between Aranjuez and Toledo.

Figure 9 shows that, for an average month, nitrate concentration in the Tagus River between those two cities respects the 25 mg/L limit only when the transferred volume is below 37 hm^3/month. Under current circumstances, the ammonium concentration in the same stretch of the Tagus River is above the allowed limit (1 mg/L) for any volume of transferred water.

In the case of the phosphate, Fig. 10 shows that the good physicochemical status (when concentration is below 0.4 mg/L of phosphate) is not attained even for small volumes of water transfer.

Therefore, even in the absence of water transfer, a corrective action is needed in the WWTPs of the Manzanares and Jarama basin in order to achieve a good status for the Tagus River after the confluence with the Jarama River.

Given the large investments associated with the upgrade of the WWTPs and the practical unfeasibility of building all the required infrastructure at the same time, a third scenario is built to define the optimal sequencing for these interventions. In the case of European legislation, preamble 29 of the WFD accepts a “phase implementation of the programme of measures in order to spread the costs of implementation”.

In this phased implementation we propose to focus each time on the river stretch that is the furthest from the physicochemical conditions associated to the good status. That is, to bridge the biggest breach of compliance at each phase. Figure 11 shows the maximum concentration of ammonium for each river stretch. At the initial (current) situation, maximum ammonium concentration (20 mg/L) occurs in the Manzanares River. Initial corrective action is therefore directed to the WWTP discharging the maximum quantity of ammonium in this river, namely Sur WWTP (see Table 1). Once that WWTP is upgraded in line with Table 3 requirements, maximum ammonium concentration (14.5 mg/L) still occurs in the Manzanares River, pointing to the next

WWTP in the same river stretch. After the second upgrade, ammonium concentration at Manzanares River drops below 9 mg/L and Jarama River upstream of Henares confluence becomes the critical river stretch with a concentration of 14 mg/L of ammonium.

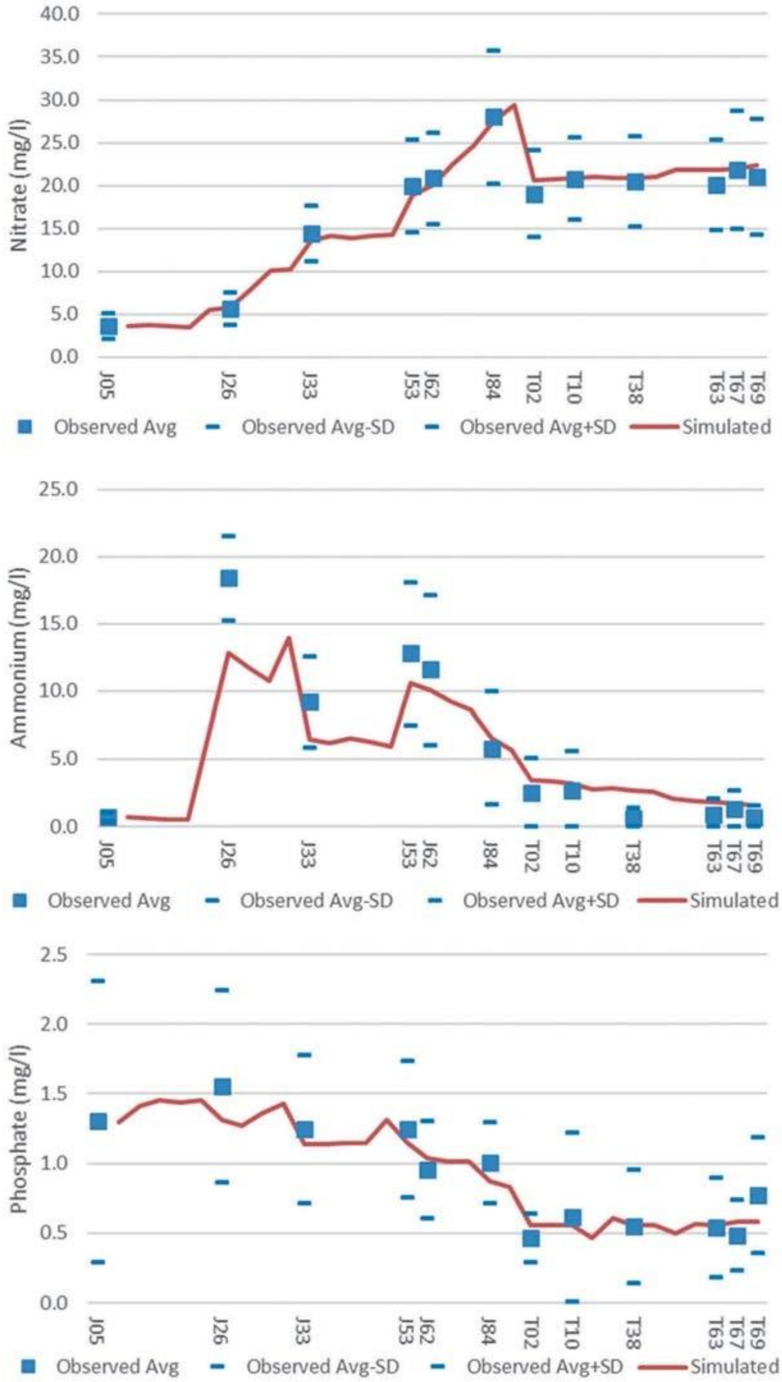

Figure 6. Simulated vs. observed nutrient values along the Jarama (J.05–J.84) and Tagus (T.00–T.69) rivers.

Table 2. Average percentage bias of modelled pollutants.

NO_3	NH_4	PO_4
5.6	42.1	13.1

Corrective action therefore addresses the largest pressure on the river stretch, namely effluents from the Rejas WWTP (Table 1). The sequencing of the upgrade of wastewater treatment infrastructure is therefore established until all the receiving surface water bodies reach physicochemical conditions compatible with good status, or until a corrective action would imply disproportionate costs as described in Article 4 of the WFD.

As noted earlier, even in the absence of water transfers, ammonium concentration at Tagus River between Aranjuez and Toledo remains above the 1 mg/L limit, due to the poor water quality of Jarama River flows. After the upgrade of five WWTPs discharging to Manzanares and Jarama Rivers (Fig. 11), ammonium concentration in the Tagus between Aranjuez and Toledo falls below the 1 mg/L limit. A fourth scenario is built to investigate which is a safe volume of water transfer that can be sustained during the process of WWTP upgrading.

For “safe” we mean that it is compatible with the achievement of physicochemical conditions associated with a good status of Tagus River between Aranjuez and Toledo. This is the stretch of our study area that is affected by the water transfer. Figure 12 shows that no amount of water transfer would be consistent with the good status before the upgrading of the first four WWTPs.

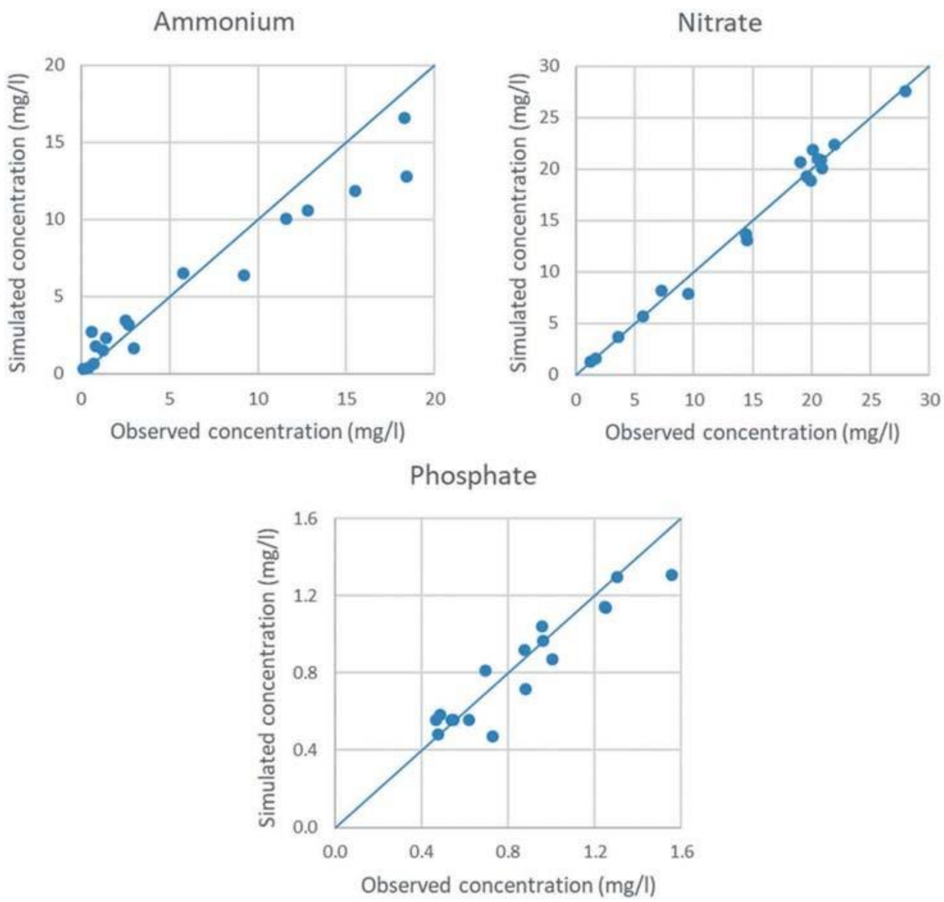

Figure 7. Scatterplots of observed and simulated concentrations of ammonium, nitrate and phosphate in the study area.

This scenario shows the intimate relation between the transferred volumes and the upgrading of WWTPs in the Madrid metropolitan region in order to achieve the good status of the receiving waters. Any decision regarding monthly water transfer volumes should guaranteethat theresulting qualityin Tagus River downstream of the confluence with Jarama is not jeopardized.

5. Discussion

The methodology chosen (a calibrated water quality stationary model upon which change scenarios are built) is deemed appropriate since it can exploit the available data and provide a better understanding of the processes involved.

Special care has been taken to choose the upper and lower ranges of the parameters to ensure that they are consistent with previous literature (Paredes et al. 2013) and representative of site conditions (Daggupati et al. 2015). Combining expert judgment of acceptable ranges with the empirical calibration allows to exploit all the available information, and in our study area has provided a better goodness of fit than the alternative option of assigning the parameter values in a "physical" manner (Thirel et al. 2015).

The implications of not performing a validation of the calibrated values of the parameters can be explored further. The absence of validation entails a loss of information on the predictive capacity of the model. Nonetheless, with such a short sample, a temporal split of data would have added uncertainty to the simulated values (due to a shorter calibration period) and the validation period would be so short that the results would not be statistically significant.

The fact that the four river water quality and six WWTP effluent observations per year are measured in different days implies that the causality link in the model can only be established for long-term averages. This difference in the sampling process responds to the fact that different departments of the Tagus River Basin Authority collect the data for different purposes (compliance with WFD prescriptions on receiving waters, and with wastewater legislation, respectively), not taking into consideration the possible use of data for modelling. A potential amelioration would be the coordination of sampling campaigns to allow for a pairwise collection of pressures and water status data.

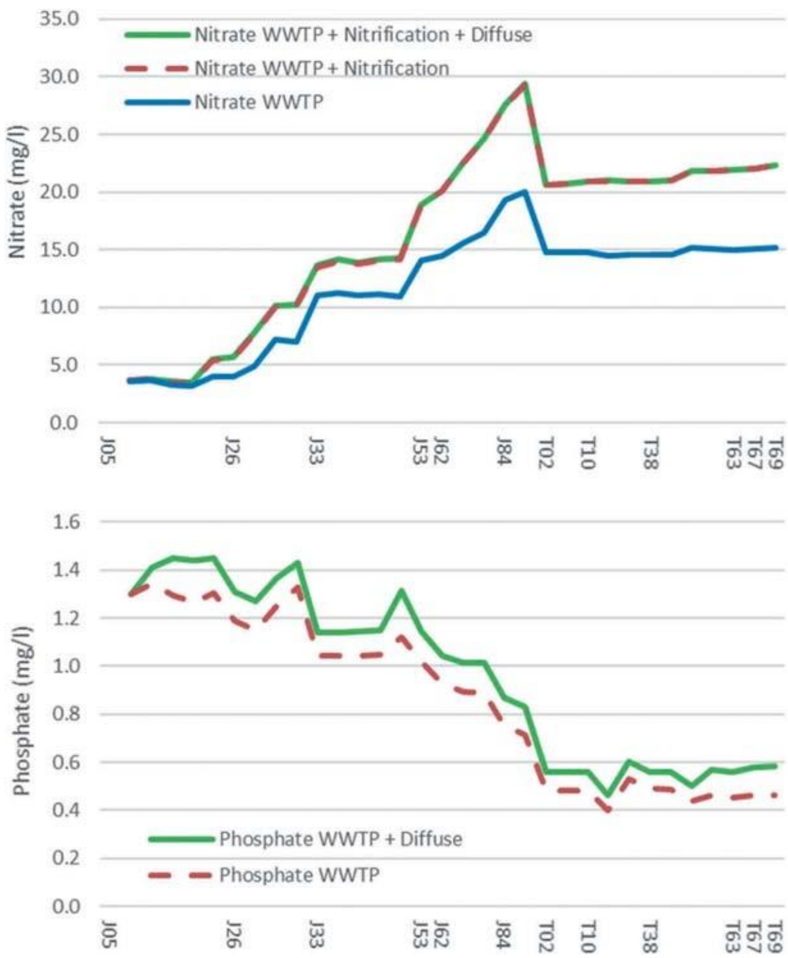

Figure 8. Relative weight of pressures on receiving waters, along the Jarama (J.05–J.84) and Tagus (T.00–T.69) rivers.

Table 3. Proposed concentration limits for WWTP effluent permits.

WWTP discharging to	NH_4 (mg/L)	NO_3 (mg/L)	PO_4 (mg/L)
Henares	4.00	60	0.65
Manzanares	0.65	30	0.55
Jarama upstream of Henares confluence	1.00	50	0.55
Jarama downstream of Henares confluence	8.00	60	1.00

In a study area where some stations consistently produce observations with lower variability than others, the proposed goal function (1) will force model simulated values to better replicate the more reliable stations. Conversely, if all observation stations produce measurements with similar variability then there is no advantage of applying the proposed method and other goal functions (PBIAS, R^2) would be more appropriate.

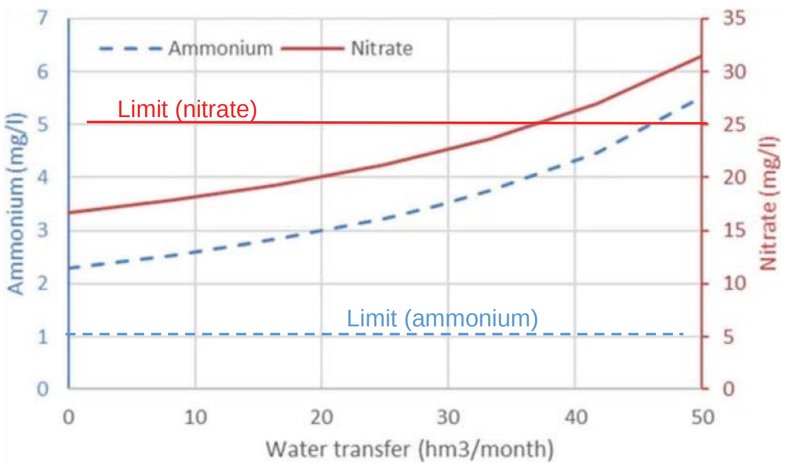

Figure 9. Simulated effect of Tagus Segura water transfer on nitrate and ammonium concentrations in the Tagus River between Aranjuez and Toledo (T.0–T.64).

Some additional considerations apply to the results of the study. Under the assumption of stationarity, only the steady state is studied; dynamic effects and particular events such as WWTP overflows due to heavy rains are not covered. A further assumption restricts the study to the critical load case of low-flow summer months. This is driven by the lack of data for the highly variable flow of rainy months. If annual averages were to be considered for the established legal standards, concentration limits at plant effluent could be relaxed. On the other hand, only physicochemical elements (i.e. the pollutants limited in WWTP discharge legislation) are studied.

The usual caveats apply to the results due to the high degree of collinearity between the calibrated factors, making aggregate results relatively more reliable. This applies particularly to the diffuse pollution, where total effect results are more solid than the particular geographical distribution.

Although PBIAS was not used for model calibration, it is still a good indicator of goodness of fit since reference thresholds are available in the literature (Moriasi et al. 2007).

6. Conclusions

The paper illustrates an approach for formulating policy recommendations to recover river water quality in a context of scarce observational data. A calibration function for the water quality model is proposed to exploit the statistical properties of the available data. This affords satisfactory calibration results even without a set of event-to-event, pressure-effect data, which would be necessary for a calibration based on R^2 or Nash-Sutcliffe criteria. As a result, the model replicates the water quality behaviour in the study area with an acceptable degree of certainty.

The approach is applied to the Middle Tagus Basin, where the model is able to quantify the relative weight of the existing pressures on surface water quality, and the policy actions

Figure 10. Simulated effect of Tagus Segura water transfer on phosphate concentration in the Tagus River between Aranjuez and Toledo (T.0–T.64).

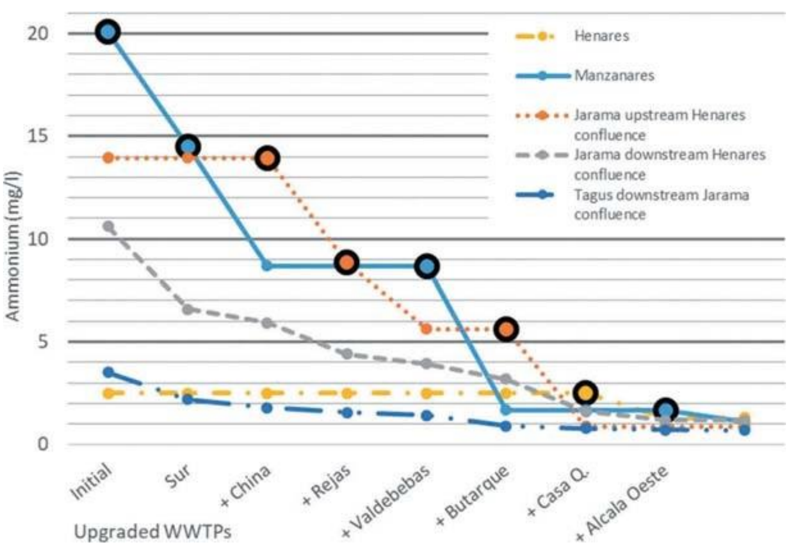

Figure 11. Simulated maximum ammonium concentration per river stretch. The x-axis indicates the initial (current) situation and the upgrade of each WWTP. For Tagus,

current levels of transferred volumes are assumed. Black circles represent the most critical position at each stage (requiring priority of intervention).

required to achieve a good physicochemical status of surface more than 80% of phosphorus pollution. More stringent conwater bodies. Results show that contaminant loads from centration limits should be set for WWTP effluents. The model WWTPs represent more than 95% of nitrogen pollution and determines that ammonium concentration must be below 0.65 mg/L for WWTPs discharging to Manzanares and below 1 mg/L for WWTPs discharging to Jarama upstream of the confluence with Henares.

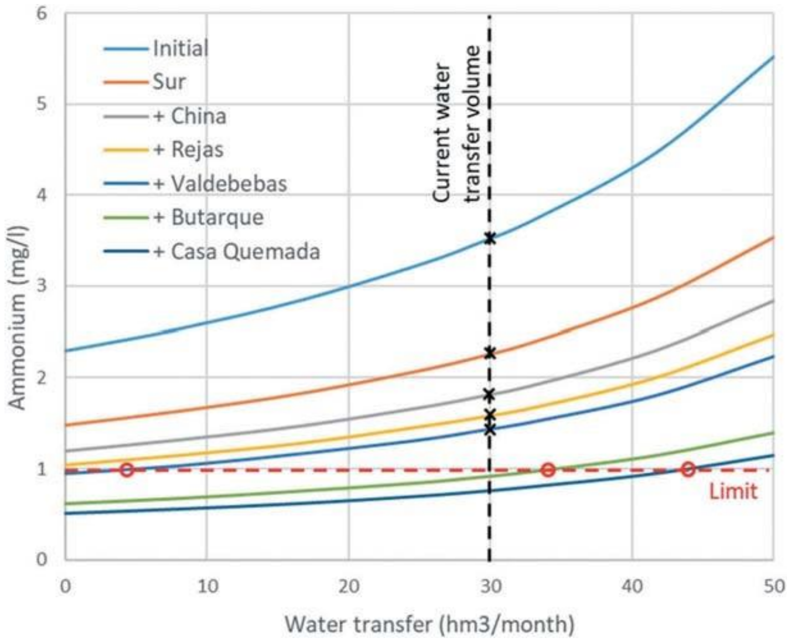

Figure 12. Expected ammonium concentration for Tagus waters between Aranjuez and Toledo for different water transfer volumes and WWTP upgrade scenarios. Crosses represent the ammonium concentration for each scenario if current transfer volume is kept, and circles represent maximum transfer volume that is consistent with the allowed concentration of ammonium. No water transfer would be consistent for the first four stages.

Due to the magnitude and cost of the intervention, this process should be phased. At each step, the model identifies the river stretch with the largest breach of the pollutant concentration limits and supports the definition of the optimal sequencing of the upgrade of five WWTPs in the study area.

Additionally, the model quantifies the effect of the Tagus Segura transfer in terms of water quality. The expected concentration of physicochemical pollutants is calculated for different transfer volume scenarios. An important result is that with the current sewage treatment infrastructure, Tagus River waters between Aranjuez and Toledo (downstream of the Region of Madrid) cannot attain a good status even in the absence of abstractions in the headwaters for inter-basin transfers.

After the upgrade of WWTP infrastructure, the model is able to inform water authorities on the volume of water that can be transferred without jeopardizing the physicochemical conditions in the Middle Tagus River. Further investigation should focus on the quantification of the costs of the required intervention, and the assessment of their affordability and proportionality with respect to the expected environmental benefits.

Disclosure statement

No potential conflict of interest was reported by the authors.

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7. Las exenciones a los objetivos medioambientales por costes desproporcionados: metodología aplicada al Tajo Medio

7.1. Introducción

En el capítulo anterior hemos visto cómo la consecución del buen estado de las aguas superficiales solamente es posible implementando medidas de mejora de la infraestructura de depuración que requieren grandes inversiones. En el apartado 4.4 se ha mencionado cómo la legislación establece que en ciertos casos se podrían flexibilizar los objetivos medioambientales para evitar incurrir en costes desproporcionados.

En el caso del Tajo Medio, en el segundo ciclo de planificación (MAPAMA 2016) se establecieron exenciones a los objetivos medioambientales de varias masas de agua (Tabla 2) con una justificación poco desarrollada. En los documentos previos al tercer ciclo de planificación (CHT 2018c) se apela al concepto de desproporcionalidad y se estiman los costes de las medidas necesarias, sin especificar los métodos de justificación de dichas exenciones. Por tanto, la investigación presentada en este capítulo desarrolla una metodología que pretende contribuir a la determinación de los casos donde sería justificable una flexibilización de los objetivos medioambientales.

7.2. Metodología

La Directiva Marco del Agua y el Documento Guía elaborado por la Comisión Europea sobre economía medioambiental (European Commission 2003c) identifican el Análisis de Costes y Beneficios (CBA en sus siglas en inglés) como la metodología preferente de justificación de los costes desproporcionados de la consecución del buen estado de una masa de agua. Desde la aprobación de la DMA se han propuesto varios métodos prácticos, que se han enfrentado con el mismo obstáculo: mientras los costes de las medidas pueden cuantificarse en unidades monetarias por año, existen grandes dificultades en expresar en las mismas unidades el beneficio obtenido por dichas medidas. La falta de métodos prácticos para monetarizar los beneficios ha impedido la generalización de justificaciones de exenciones y su comparabilidad. Para ello, se propone un método basado en los siguientes pasos:

- Una vez identificadas las presiones causantes del estado peor que bueno (ver capítulo anterior), se identifican las medidas que por un criterio de eficiencia serían capaces de contribuir a la consecución del buen estado con un coste mínimo (C en la Figura 15). Dicho coste es expresado (en unidades monetarias anuales) como la suma del coste anual de operación y mantenimiento (OPEX) más el equivalente anual de la inversión inicial (CAPEX), calculado mediante fórmulas financieras (European Commission 2008) teniendo en cuenta la vida útil de la infraestructura.

Se necesita una caracterización del coste real de las medidas, a partir de la experiencia de los costes de medidas similares anteriores.

- El beneficio cualitativo (qB) se cuantifica con una media ponderada de la mejora de los indicadores de calidad de las masas de agua de la zona de estudio (NOD). Los factores de ponderación son el peso relativo que se le otorga a cada masa de agua (WB), lo que posibilita dar mayor peso a masas de mayor volumen o en situación de protección; y el peso relativo de cada tipo de indicador (QE), permitiendo dar mayor relevancia a los indicadores más importantes (como puedan ser los biológicos), o a aquellos cuya evolución tras las medidas se conozca con mayor certeza (como puedan ser los elementos físico-químicos). La mejora medioambiental se mide por tanto con los mismos indicadores descritos por la DMA para caracterizar el estado de cada masa de agua (apartado 4.2).
- El beneficio se monetariza mediante un factor de capacidad presupuestaria (WSPE), que tiene en cuenta tanto el Producto Interior Bruto (GDPsa) del área causante de las presiones (bajo el concepto de quien contamina paga), como el porcentaje medio de gasto medioambiental (PEE) a nivel europeo.
- Se crea un Factor de Desproporcionalidad (DF), que compara el coste de cada medida con el factor de beneficio, compuesto por la mejora medioambiental que se espera conseguir y la capacidad presupuestaria de la región responsable de la presión.

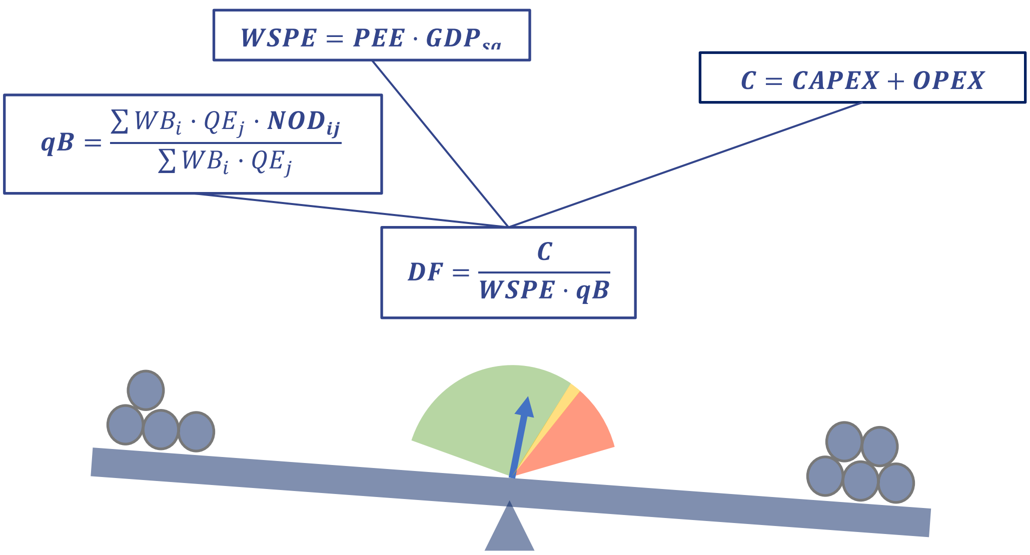

Figura 15. Factor de Desproporcionalidad

La metodología permite establecer una jerarquía entre las medidas propuestas para las masas de agua en estado peor que bueno que ayude a priorizar las actuaciones, y a justificar las exenciones en los casos en que alcanzar el buen estado suponga un coste desproporcionado. El umbral de desproporcionalidad, es decir el valor del Factor de Desproporcionalidad a partir del cual se consideran justificadas las exenciones, no debería fijarse arbitrariamente en cada demarcación, sino que debería consensuarse a nivel europeo para garantizar que los esfuerzos para alcanzar los objetivos de la DMA sean similares en los distintos Estados Miembros de la UE.

7.3. Resultados y discusión

En su aplicación al Tajo Medio, el método propuesto asigna un valor cuantitativo al Factor de Desproporcionalidad, de manera que se puedan comparar las medidas propuestas (según sus costes y beneficios) con las de otras demarcaciones.

Si se considera que el coste total es inasumible y que solamente se pueden efectuar una parte de las medidas propuestas, el método identifica las medidas con mayor beneficio por coste.

7.4. Conclusiones

El método propuesto supone una caracterización sistemática de los costes y beneficios de las medidas estudiadas. Al organizar toda la información en un solo índice de desproporcionalidad, el método ayuda a visualizar el esfuerzo y beneficio requerido en cada contexto y racionalizar la distribución de recursos.

En todo caso, la metodología propuesta no puede fijar el valor del umbral a partir del cual los costes se consideran desproporcionados. Esta debe ser una decisión consensuada entre todos los Estados Miembros de la Unión Europea, que garantice una distribución equitativa de esfuerzos para alcanzar los objetivos medioambientales. Para contribuir a la definición del umbral, en el siguiente capítulo se presenta una comparativa de la política de exenciones de seis demarcaciones europeas, donde se confrontan los niveles de impacto de las presiones a partir del cual se empiezan a flexibilizar los objetivos medioambientales.

7.5. Artículo

A continuación, se reproduce, con el permiso de los coautores, el contenido del artículo:

Antonio Bolinches, Lucia De Stefano and Javier Paredes-Arquiola (2020) Too expensive to be worth it? A methodology to identify disproportionate costs of environmental measures as applied to the Middle Tagus River, Spain, Journal of Environmental Planning and Management, DOI: 10.1080/09640568.2020.1726731

Too expensive to be worth it? A methodology to identify disproportionate costs of environmental measures as applied to the Middle Tagus River, Spain

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Abstract

The European Union (EU) Water Framework Directive (WFD) established in 2000 that EU Member States should achieve good status for all their water bodies by 2027 at the latest. The competent authorities are obliged to commit the necessary resources to achieve this goal. In water bodies where the costs are deemed disproportionate, the Directive foresees the definition of exemptions. Two decades after approval of the WFD, however, there is no common method across the EU to evaluate the disproportionality of costs and define the associated exemptions. We propose a methodology based on WFD indicators of water body status and economic variables that are common to all the EU countries. The method uses data that is already available in Eurostat and European Environment Agency databases, thus minimizing data collection costs. The method is applied to the Middle Tagus (Spain), where currently there are several water bodies with declared exemptions for disproportionate costs.

Keywords: Water Framework Directive; disproportionate costs; less stringent objectives; Tagus

1. Introduction

Achieving environmental improvements at times can imply high costs in terms of investments, or opportunity costs for economic activities that would need to change substantially to be environmentally sustainable. When the required actions are particularly expensive, policy makers are faced with the question “is it worth it?”. Answering that question requires factoring in many aspects, including legal, political, economic, and technical issues, and practitioners need tools and data to inform their decisions.

The European Union (EU) Water Framework Directive (WFD) (European Parliament and Council 2000) represents an interesting case where the legislation explicitly acknowledges the dilemma of disproportionate costs of the measures required to achieve given environmental objectives. Corrective measures should be defined following cost-effectiveness criteria (European Commission 2003), but in some situations even the most cost-effective measure may prove extremely onerous. For this purpose, the WFD justifies exemptions to its environmental objectives. These can be applied as a time extension when “all the necessary improvements in the status of water bodies cannot reasonably be achieved within the timescales” (Article 4.4), or as a relaxation of the environmental objective (“Member States may aim to achieve less stringent environmental objectives (…) if the achievement of these objectives would be infeasible or disproportionately expensive”; Article 4.5).

According to the most recent River Basin Management Plans (2016–2021), exemptions based on disproportionate costs have been claimed for 2% of groundwater bodies across the EU. In the case of surface water bodies, cost disproportionality supports exemptions in 8% of the chemical status objectives and 22% of ecological status objectives (European Commission 2019). As a result, more than nine thousand water bodies in the EU are affected by exemptions justified with the disproportionate cost clause.

EU Member States may face economic penalties in the future if they do not comply with the established environmental objectives, or if they do not clearly justify exemptions. On one side the European Commission (EC), who oversees the adequate implementation of the Directive, needs clear criteria to assess whether Member States are misusing exemptions. On the other side, the application of uneven methods across and within Member States could create unfair situations. As a matter of fact, since the approval of the WFD in 2000, the EC and Member States have worked on a Common Implementation Strategy (CIS) to create common technical approaches on several aspects of the Directive.

In this context, the CIS has provided Member States with guidelines about economic aspects of the WFD (European Commission 2003) and the application of exemptions to environmental objectives (European Commission 2009). These documents recommend the application of Cost and Benefit Analysis (CBA) approaches in order to identify the disproportionality in the cost of measures, but do not provide a clear roadmap for their implementation. One reason for this is the lack of clarity over which measure needs to be taken in order to improve the water status and what would be the respective cost of the measure. This has led to misinterpretations and implementation problems (Berbel and Exposito 2018), and to inconsistent ways of applying disproportionality across the EU (Martin-Ortega et al. 2014). Disproportionality is often supported with poor justifications and insufficient information on underlying criteria and methods (Boeuf, Fritsch, and Martin-Ortega 2016). The misuse of exemptions and uncertainty about their correct application have led European Water Directors to consider changes in the legislation, such as broadening the application of time extensions beyond 2027 or extending the scope of the reference natural conditions (European Water Directors 2019).

1.1. How to assess whether it is worth it?

Existing literature on evaluation of the economic viability of environmental measures can be grouped into three strands (Machac and Brabec 2018; Gorlach and Pielen 2007): Monetized Benefits CBA, Qualitative Benefits CBA and Cost/Affordability.

All these approaches start from the estimation of the cost of a given intervention, which will then be compared with a benefit estimation term in order to decide whether or not the expenditure is worth it. The CBA approach compares the estimated cost of a given set of measures with their expected benefit. Monetized benefit CBA quantifies the economic welfare that the society gains by solving the environmental problem (Hernandez-Sancho, Molinos-Senante, and Sala-Garrido 2010; Molinos-Senante, Hernandez-Sancho, and Sala-Garrido 2010; Martin-Ortega et al. 2014; Jensen et al. 2013; Galioto et al. 2013). It is deemed to be the only criterion that ensures that the chosen solution is socially desirable (Gorlach and Pielen 2007). However, most of the environmental benefits are not traded in existing markets and it is therefore difficult to attribute a monetary value to many environmental and social benefits of the measures (European Commission 2009). Two main approaches have been developed to quantify this value. The Revealed Preference methods (TEEB 2010; Martin-Ortega et al. 2014) study the evolution of markets linked to the area under study, while the Stated Preference methods (Brouwer 2008) are based on surveys on the willingness to pay of the population.

Irrespective of the method used, the net present value for each cost and benefit has to be calculated. This is done through the application of a discount rate that reflects the social preference for present, rather than future, consumption. Additional assumptions on the value of the discount rate for future costs and benefits are needed, since there are ethical considerations to attribute lower importance to costs and benefits for future generations in an environmental context (Martin-Ortega et al. 2014). Some authors (Almansa and Martınez-Paz 2011; Roumboutsos 2010) suggest that future environmental benefits should be subject to lower discount rates. It is subsequently apparent that a formal Cost and Benefit Analysis is not straightforward to conduct because it requires a large quantity of assumptions and information (Jensen et al. 2013).

Because of the inherent difficulties of the CBA approach, Water Authorities in Spain (CHG, 2015), France, Sweden, and the UK (Thaler et al. 2014) allow substituting the calculation of monetized benefits with the definition of a qualitative scale (Qualitative benefits CBA). A batch of indicators is created to assess the expected impact of the measure in terms of, amelioration of water quality for abstraction, reduction of risk to drought and flooding; economic activities linked to the water use and recreational opportunities. Each indicator is rated by the technicians conducting the analysis. They are aggregated to form the benefit rating, which will be compared to the costs. The method presents two limitations; first, subjectivity in the selection of indicators (i.e., choosing which effects are considered), and second, the subjectivity of the rating.

Affordability methods compare the estimated costs with some metrics of the capacity of society to cover those costs. A widely used metric is household income (Courtecuisse 2005). In relation to water management measures, for instance, the Organization for Economic Co-operation and Development (OECD 2009), suggests that water supply and sanitation bills for a household should not represent more than 5% of its disposable income. Another metric of affordability relates to the current level of expenditure to address a given problem (Laurans 2006). For instance, the Average Cost or Old Leipzig Approach (Ammermuller et al. 2011; Klauer, Schiller, and Sigel 2017) confronts the cost of achieving the good status of a water body (normalized with the catchment area of the water body) with a threshold calculated as a function of average normalized costs in all water bodies in the same administrative region. An evolution of this method is the Benchmark or New Leipzig Approach (NLA) (Klauer, Sigel, and Schiller 2016; Klauer, Schiller, and Sigel 2017; Machac and Brabec 2018), which, instead, compares the costs of achieving good status with the average water quality expenditure before the implementation of the WFD, reducing the data requirements with respect to the Average Cost approach. The NLA method will be presented in detail in the Methodology section. The EU Water Directors agreed to accept both CBA and affordability arguments to justify exemptions, although in the case of less stringent objectives the suitability of the affordability principle was more disputed (European Commission 2009).

The cost disproportionality clause is applied differently across countries and river basin districts in the EU (Martin-Ortega et al. 2014), highlighting the need for harmonization of methods (Berbel and Exposito 2018). In the scholarly literature, there are few studies that provide explicit recommendations to the water authorities. Additionally, most of the available studies do not consider interdependencies between upstream and downstream water bodies (Galioto et al. 2013). A balance is needed between sophistication and representativeness. The more sophisticated the method, the better it can reflect the local conditions of a given water body or river basin district. At the same time, this implies additional assumptions that may add a layer of subjectivity. This limits the transferability of the same method to other areas and the comparability of results across regions.

Albeit admitting the difficulty of quantifying the benefits, many studies undertake complex analysis to include as many externalities as possible. This may divert attention from the fact that the main benefit of the measures is to improve the status of the water bodies, as requested by WFD Article 11. Other benefits cascading from the achievement of the environmental objectives (e.g., increased tourist activity associated with freshwater ecosystems) may be included in the evaluation as an additional asset. The analysis should be as simple as possible, then as detailed as necessary (European Commission 2009). Any additional benefits to the community can be taken into account after the disproportionality study, since in the end it is a political decision informed by economic information (European Commission 2003).

In this paper, we start from an existing methodology, the New Leipzig Approach (Klauer, Sigel, and Schiller 2016; Klauer, Schiller, and Sigel 2017), to propose an improved approach that facilitates comparability across cases and minimizes the costs associated with the collection of relevant data, as required by WFD Annex III (European Parliament and Council 2000). Moreover, we apply the original NLA to our case study to discuss its results in relation to other case studies in the RhinelandPalatinate (Klauer, Sigel, and

Schiller 2016), another federal state in Germany (Klauer, Schiller, and Sigel 2017), and the Czech Republic (Machac and Brabec 2018).

Particular attention has been paid to ensure that the method is based on variables from Eurostat and European Environment Agency databases that are available for all the countries in the EU. This responds to a prominent concern of the Common Implementation Strategy for the WFD (European Commission 2009) to provide a level playing field for decision making and compliance assessment.

Our approach is applied to assess the disproportionality of costs in the Middle Tagus Basin, central Spain. This region is subject to high urban pressure combined with low natural river flows, which has driven the Tagus River Basin Authority (RBA) to declare less stringent objectives for several surface water bodies in the 2015–2021 River Basin Management Plan (RBMP).

2. Study area

Our study area is located in the Middle Tagus Basin (Figure 1). Rivers and streams in this region (Jarama, Manzanares, and Tagus Rivers) bear the pressure exerted by the 4.7 million inhabitants of the Madrid metropolitan area (INE 2018a). Despite continuous efforts in wastewater treatment, the waters receiving treated effluents still have high concentrations of physico-chemical pollutants (CHT 2018b).

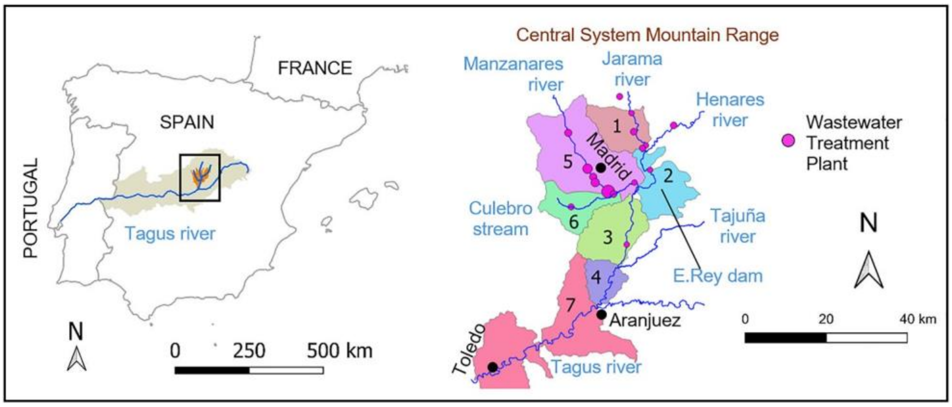

Figure 1.Study area. Numerical codes defined in Table 1.

Consequently, significant additional investments in infrastructure are needed to achieve good status for the receiving waters. Claiming disproportionate costs, Tagus RBA has defined less stringent environmental objectives for six water bodies in the region.

The diagnosis by the Tagus RBA concludes that the pressures that hamper the achievement of good status are mainly from urban point sources. Therefore, the cost of good status was calculated as the cost of additional removal of physico-chemical pollutants discharged by Wastewater Treatment Plants (WWTP). The Tagus RBA declared a potential cost of 113 M€/year (CHT 2018a) to achieve good status in the water bodies with less stringent objectives, considering a lifetime of 15years for the required infrastructure. This quantity is expressed in euros for the year 2006 and is exempt from taxes and subsidies (CEDEX 2012). We present an independent calculation of this cost in the Supporting Material 1.

Table 1 shows the environmental objectives of physico-chemical pollutants for each water body, as established in the Spanish legislation (MAPAMA 2015) and the less stringent objectives declared in the 2016–2021 RBMP (MAPAMA 2016). The Table also shows the average concentration measured in those river stretches during the 2009–2015 period (CHT 2018b). Environmental and less stringent objectives are met for nitrate concentrations for all except one water body (Culebro stream). Phosphate concentrations comply with the less stringent objectives, but are above values compatible with good status. Ammonium concentrations are above the less stringent objectives in four surface water bodies. The Tagus River, from the Jarama junction to the city of Toledo, has no significant pressures and no less stringent objectives declared. It is, nevertheless, impacted by pollution upstream, as shown by the high ammonium concentrations observed, which are fivefold the concentrations compatible with good status. WWTPs in the Madrid metropolitan area affect the surface water quality from the middle Jarama River to the city of Toledo (Bolinches, De Stefano, and Paredes-Arquiola 2020), as indicators downstream of Toledo show ammonium and phosphate concentrations that comply with good status requirements for physico-chemical elements. The interdependency of water quality in those water bodies (Galioto et al. 2013) has been the criterion used to group them in our analysis of cost disproportionality rather than studying them individually.

Table 1. In-stream concentration associated with environmental objectives (EO), less stringent objectives (LS), and 2009–2015 observed averages (OA) (CHT 2018b).

Concentration (mg/l)	Ammonium			Nitrate			Phosphate		
Water body	EO	LS	OA	EO	LS	OA	EO	LS	OA
1 Jarama from Vbebas to Henares	0.6	10	15	25	25	6	0.5	3.1	1.2
2 Jarama from Henares to E. Rey	0.6	8	8	25	25	14	0.5	3.1	1.2
3 Jarama from E. Rey to Tajuña	0.6	10	13	25	25	17	0.5	3.1	0.9
4 Jarama from Tajuña to Tajo	0.6	10	9	25	25	22	0.4	3.1	0.9
5 Manzanares through Madrid	0.6	10	25	25	25	16	0.5	3.1	1.0
6 Culebro stream	0.6	2	3	25	25	47	0.4	3.1	1.3
7 Tagus from Jarama to Toledo	1.0	–	5	25	–	24	0.4	–	0.6

Note: Phosphate LS objectives are declared as Total Phosphorous concentration, here expressed as phosphate equivalent for the sake of comparison with EO. Water body numbering refers to Figure 1.

3.Methodology and data

The method used in this study is based on the New Leipzig Approach (Klauer, Sigel, and Schiller 2016; Klauer, Schiller, and Sigel 2017). The NLA assesses disproportionality through a series of steps: past average public expenditure for water protection is calculated for the country where the study area is located, and then divided by the country area to produce a reference expenditure in euros per year per km^2. The resulting value is multiplied by the surface of the catchment of the water body under study to calculate the specific reference expenditure. An effort factor is defined as a linear increasing function of two variables: first the difference between the current and the good status of the water body (measured using four biological indicators and the EU Environmental Quality Norm) and, second, an additional benefit factor that takes account of amelioration of terrestrial ecology, freshwater, flood and soil protection and tourism. A bigger effort factor reflects a higher expected amelioration of water bodies after the application of the measures. Multiplying the effort factor with the water body reference expenditure produces the water body specific disproportionality threshold for costs. The factors are designed to allow for a maximum additional expenditure of 50% with respect to previous investments. If the cost of the measures under study was above this threshold, the intervention would be deemed disproportionate. A set of modifications is proposed in this paper in order to simplify the application of the NLA through the exploitation of existing databases, and to improve the comparability of results across different countries and river basin districts.

In order to correctly allocate larger economic resources to cases with more important compliance breaches, the costs of achieving good status in each water body (usually stated in the Programme of Measures) are identified. Then, a term that measures the environmental amelioration (qualitative benefit, as an objective distance to good status) is calculated. Finally, this benefit term is monetized through a reference expenditure term to facilitate comparison with costs and assess the proportionality of the measures. A detailed comparison between the proposed method and the NLA is presented in the discussion section.

In the next sections, we will describe the data sources needed to apply the methodology, the quantification of the gap between current and good status of the water bodies under study, the estimation of the economic capacity of the region that would finance the measures and the calculation of a factor to discern the cases with better arguments to justify an exemption due to cost disproportionality.

3.1. Normalized Objective Distance to good status

The benefit of the measure, that is, the compliance breach that is bridged in the study area is calculated by aggregation. First, an objective distance parameter is calculated for each water quality indicator (as defined by the WFD) in each water body in the study area. The WFD indicators describe the status of the surface water bodies and measure chemical, biological, hydromorphological, and physico-chemical elements. To ensure comparability and homogeneity of data across the EU, we use the values of indicators reported by Member States to the EC and the European Environment Agency (EEA 2019b).

Since each indicator (IND) is measured in the EEA database (EEA 2019b) with different scales (1–5 for biological and hydromorphological, 1–3 for physico-chemical, and 2–3 for chemical indicators), the objective distance parameter is normalized. The Normalized Objective Distance (NOD) is calculated for each water body (i) and each quality element (j) dividing the status change (from current to good) by the maximum gap of the indicator scale:

$$NOD_{ij} = \frac{\left(IND_{ij,current} - IND_{ij,good}\right)}{IND_{maxgap}} \quad (1)$$

Where NODij ∈ [0,1] and is dimensionless. It is assumed that the measures are sufficient to reach a good water status. Table 2 shows the available status codes for each indicator type, and the corresponding normalized objective distance. For example, a water body i with a moderate status of a biological quality element j would have a current status code of $IND_{ij,current} = 3$ and a good status code of $IND_{ij,good} = 2$. Considering a maximum gap of $IND_{maxgap} = 5 - 1 = 4$ for biological indicators, the objective distance to good status would be $NOD_{ij} = \frac{3-2}{4} = 0.25$. Status codes and NODs for all indicator types are presented in Table 2.

In the case that a measure affects more than one water body, the values of all the NOD parameters of the water bodies in the study area are aggregated to obtain a global value. The consolidation of a group of water bodies in one single analysis must be duly justified. The expected qualitative Benefit (qB) of the measure is calculated as the weighted average NODs, taking into account the relative importance of the water bodies and the indicator types. It represents the magnitude of the problem to be solved with the corrective measure.

$$qB = \frac{\sum_{i,j} WB_i \cdot QE_j \cdot NOD_{ij}}{\sum_{i,j} WB_i \cdot QE_j} \quad (2)$$

qB Expected qualitative Benefit of the measure for the study area,
WB_i Relative weight of Water Body i, and
QE_j Relative weight of Quality Element j.

Where all variables

$$(qB\ ,\ WB_i\ ,\ QE_j\ ) \in [0,1]$$

and are dimensionless.

Table 2. WFD status codes (European Water Directors 2016) and Normalized Objective Distance to good status.

Indicator IND	Status code					Max. gap	Normalized Objective Distance from current to good category (gap/max.gap)				
WFD report, biological and hydromorph.	1	2	3	4	5	4	–	0	0.25	0.50	0.75
	High	Good	Moderate	Poor	Bad		High	Good	Moderate	Poor	Bad
WFD report, physico-chemical	1	2		3		2	–	0		0.50	
	High	Good		Worse than good			High	Good		Worse than good	
WFD report, chemical	2			3		1	0			1	
	Good			Poor			Good			Poor	

The water body factor (WB_i) takes into consideration the different relative importance that decision makers give to each water body, based on its size and ecological value. Size can be expressed in terms of length and average flow (natural or altered) for river water bodies, and in terms of average stored volume and areal extent for lakes and groundwater bodies. Ecological value can take account of special protection status listed in WFD Annex IV (European Parliament and Council 2000); i.e., water intended for human consumption, presence of economically significant aquatic species, recreational waters, nutrient-sensitive areas, and habitat protection, including Natura 2000 sites. Other protection figures can be included in the analysis. So, for the different types t of protection figures:

$$WB_i = \sum_t WB_{i,t} \tag{3}$$

The relative weight of the Quality Element (QE_j) allows different significance for each indicator type (chemical, biological, hydromorphological, and physico-chemical). In this case, besides the relative importance, decision makers could also take into account the degree of certainty of the expected status of the water bodies after the implementation of the measures under study. For each of these factors r:

$$QE_j = \sum_r QE_{j,r} \tag{4}$$

The relative weight of each of these factors should be a consensual decision at European scale to level the playing field advocated for EU Member States (European Commission 2009).

3.2.Reference expenditure

A comparison between costs and the qB term corresponds to a Qualitative Cost and Benefit Analysis. In this section, we propose an additional step to include the economic capacity of the region causing the pressure.

Firstly, a reference value for past budgetary capacity and political will is calculated. The environmental protection expenditure per country is calculated and expressed as a percentage of its national GDP. The values per country are readily available at the statistical office of the EU (Eurostat 2019). The relative expenditures for each Member State are averaged to calculate past relative expenditure at a European level.

The resulting percentage is then applied to the GDP for the study area. This will be a proxy for the local capacity to finance the corrective effort. The reference expenditure is therefore calculated as:

$$WSPE = PEE \cdot GDPsa \qquad (5)$$

WSPE Water body-specific reference past expenditure (e/year)
PEE Past European-level relative environmental expenditure (% of GDP)
GDPsa Gross Domestic Product of the study area (e/year)

This parameter formulates a central assumption of the methodology. PEE parameter reflects the average budgetary effort at European level to ameliorate the environment. Using it as a reference yardstick to assess the measures under study guarantees both proportionality (that future expenditure is comparable to past budgetary compromises) and homogeneity (that all Member States apply similar efforts to achieve their environmental objectives). Since PEE is presented as a percentage of GDP, multiplying it by the domestic product for the study area (GDPsa) results in a reference expenditure in euros per year for our area of interest (further implications of this formulation, along with a comparison with the original NLA approach are presented in the discussion chapter).

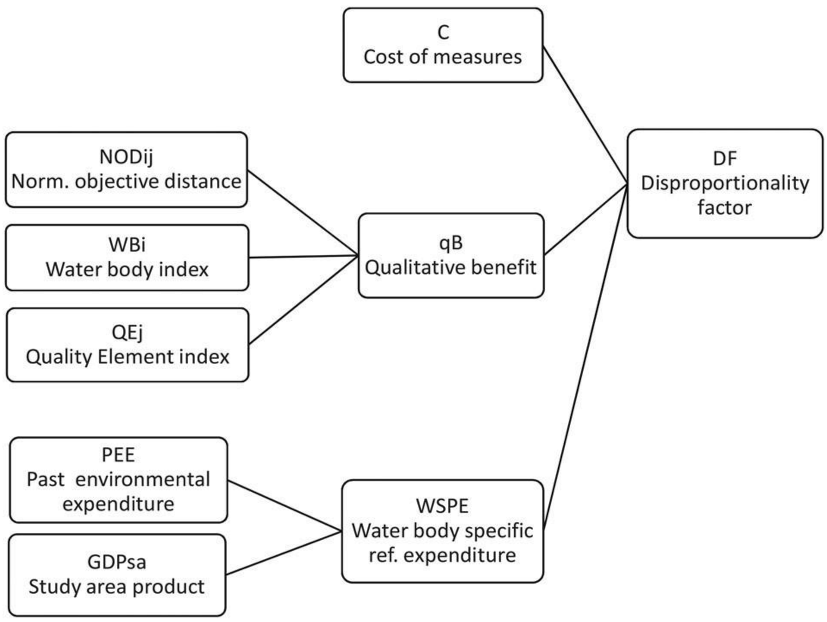

Figure 2.Calculation steps for the disproportionality factor.

3.3.Disproportionality factor

Once the costs, qualitative benefits and economic capacity have been evaluated, the information can be combined to form an index that straightforwardly assesses whether or not disproportionality may be claimed. The Disproportionality Factor is then defined as:

$$DF = \frac{C}{qB \cdot WSPE} \qquad (6)$$

DF Disproportionality Factor (dimensionless)

C Cost of measures (€/year)

qB Expected qualitative Benefit (dimensionless)

WSPE Water body-specific reference past expenditure (€/year)

The cost of the measures (C) should take account of the initial capital expenditure (CAPEX) and the operating expenses (OPEX) for the measures, both expressed in equivalent euros per year. The annual equivalent cost (European Commission 2003; Jacobsen 2005) of the initial expenditure should be calculated using a realistic discount rate (European Commission 2008, Annex B). Expected Qualitative Benefit qB and Water body-specific reference past expenditure WSPE are calculated using Equations (2) and (5), respectively.

The Disproportionality Factor (Figure 2) is built as a scalar that grows linearly with the cost of the measures. On the contrary, it will decrease with a greater enhancement of the environmental status of the study area (measured through the qB factor) as a result of the implementation of the measures. The DF factor will also diminish with the domestic product for the study area, since higher economic capabilities hinder the exemption claims on financial terms.

A measure that implies relatively high costs, but would imply that the water body evolves from a poor to a good status for all its indicators (high qB score) and is financed by an extremely wealthy region (high WSPE capacity), would yield a relatively low DF, making it difficult to claim disproportionality. On the contrary, if a measure with the same cost in a similar area would only imply an evolution from moderate to good status for a few quality indicators (low QB score), its DF would be ostensibly higher, endorsing the claim for cost disproportionality. The proposed method strives to compile all the significant information in the most concise form.

With this definition, the Disproportionality Factor represents a one-dimensional projection of a cloud of information of measured costs and expected benefits, account taken of the budgetary capacity of each area. Thus, the Disproportionality Factor is calculated for the study area as a whole. Water authorities at a European level should agree upon the guidelines to define a threshold value for the Disproportionality Factor, above which exemptions could be justified. As advanced by Berbel and Exposito (2018), this consensus is required to guarantee that the economic tools are operational and useful to draw conclusions.

3.4.Data

The data used in the proposed methodology is extracted from the following databases:

- Current water body status: the status (calculated as the worst value out of the chemical and ecological status) is described through the values of several indicators (WFD Article 2 and Annex V). EU Member States declare the current values for each of these indicators and the data is compiled and published by the EEA Water Information System for Europe (WISE) database (EEA 2019b).

- Average past water protection expenditure: calculated and expressed as a percentage of Gross Domestic Product (GDP). The values per country are available at the statistical office of the EU (Eurostat 2019) under Eurostat code env_ac_exp2 ("Environmental protection expenditure in Europe as % of GDP"). The implications of using this indicator are examined in the discussion section.
- Economic capacity of the study area: measured as the local GDP. The values per region are available under Eurostat code nama_10r_2gdp (GDP for NUTS 2 regions) (Eurostat 2019).

4.Results

The proposed methodology is applied to our study area of the Middle Tagus, where an upgrade of the existing WWTPs is being considered in order to ameliorate the status of the receiving waters.

4.1.Objective distance in the study area

The objective distance is calculated using the values of the current categories of the water bodies for 2016 (Table 3) published in the WISE WFD database (EEA 2019b). Assuming that after the implementation of the planned measures, all the ecological indicators reach the values associated with good status, the Normalized Objective Distance is calculated for each indicator (Equation 1). For this case study, we will assume that the indicators of chemical status remain unchanged and are not taken into account. We will further assume unit weighting factors for water bodies and quality indicators ($WB_i = 1$, $QE_j = 1$ $\forall$ i,j): The implications of this assumption will be considered in the discussion section. The resulting expected qualitative Benefit is then calculated (Equation (2)) as qB = 0.21.

Table 3. Current value of water quality indicators (WISE WFD database) and Normalized Objective Distance to good status (based on Table 2).

	Current status									Normalized Objective Distance								
	Biological		HM	Physico–Chemical						Biological		HM	Physico–Chemical					
Water body	QE1-2-4	QE1-3	QE2-3	QE3-1-3	QE3-1-4	QE3-1-5	QE3-1-6-1	QE3-1-6-2	QE3-3	QE1-2-4	QE1-3	QE2-3	QE3-1-3	QE3-1-4	QE3-1-5	QE3-1-6-1	QE3-1-6-2	QE3-3
1 Jarama from Vbebas to Henares	4	4	2	1	1	1	3	3	–	0.5	0.5	0	0	0	0	0.5	0.5	–
2 Jarama from Henares to E. Rey	4	5	2	1	1	1	3	3	–	0.5	0.75	0	0	0	0	0.5	0.5	–
3 Jarama from E. Rey to Tajuña	4	4	2	1	1	1	3	3	–	0.5	0.5	0	0	0	0	0.5	0.5	–
4 Jarama from Tajuña to Tajo	3	3	2	1	1	1	3	3	–	0.25	0.25	0	0	0	0	0.5	0.5	–
5 Manzanares through Madrid	4	4	2	1	1	1	3	1	–	0.5	0.5	0	0	0	0	0.5	0	–
6 Culebro stream	4	4	2	1	1	1	3	1	1	0.5	0.5	0	0	0	0	0.5	0	0
7 Tagus from Jarama to Toledo	2	3	2	1	–	1	3	1	3	0	0.25	0	0	–	0	0.5	0	0.5

Notes: HM - Hydromorphological.
QE1-2-4 - Phytobenthos.
QE1-3 - Benthic invertebrates.
QE2-3 - Morphological conditions.
QE3-1-3 - Oxygenation conditions.
QE3-1-4 - Salinity conditions.
QE3-1-5 - Acidification status.
QE3-1-6-1 - Nitrogen conditions.
QE3-1-6-2 - Phosphorus conditions.
QE3-3 - River Basin Specific Pollutants.

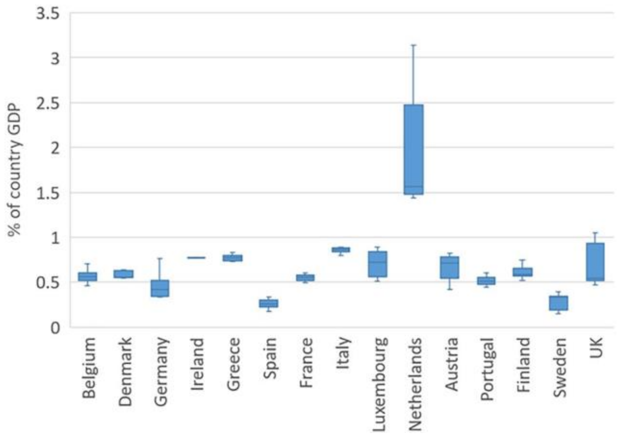

Figure 3. Box and whiskers plot of annual expenditure on environmental protection (1995–2013).

An important aspect of the qualitative Benefit calculation is the level of certainty of the expected evolution of the indicators after the measures have been implemented, and its influence on the chosen QEj factors. An example of the sensitivity of the results to these effects is presented in subsection 5.1.

4.2.Reference expenditure in the study area

Figure 3 shows the average environmental protection expenditure of the first 15 countries to join the EU (for which the most complete data is available), under Eurostat code env_ac_exp2 (Environmental protection expenditure in Europe as % of GDP, Eurostat 2019). The distribution is calculated for the available data for the period 1995–2013. The average effort results in PEE = 0.55% of the GDP.

Current (2018) GDP/person in the Madrid region (Eurostat for NUTS 2 regions, code nama_10r_2gdp) is 33,800 €/(inhab·year), which for 4.7 million inhabitants in the study area gives a Gross Domestic Product (GDPsa) of 158,860 M€/year.

The resulting reference Product-Based Expenditure WSPE (Equation (5)) is 874 M€/year, i.e., 0.55% of the GDPsa.

4.3. Disproportionality factor in the study area

The estimated cost of corrective measures in our study area (i.e., the upgrade of WWTPs) is estimated (see Supporting material 1) to be 125.7 M€/year. Under the assumption that all indicators reach good status, Equation (6) would yield the following disproportionality factor:

$$DF = \frac{125.7\ \text{M€/year}}{0.21 \cdot 874\ \text{M€/year}} = 0.68$$

Whether this value is above or below the established threshold would justify the application of exemptions. Given the public good nature of the WFD objectives, and in the interest of future generations, decision makers may be willing to accept higher values for the cost-benefit ratio (Pulido-Velazquez et al. 2009; Brouwer 2008). The criteria to establish this threshold remains a political decision that should be discussed and agreed upon at European level.

5. Discussion

The disproportionality of the cost of the measures in our case study depends on the weight of the factors and on the assumed amelioration of the indicators after implementation of the planned measures.

A necessary condition for the selection of the variables used in the evaluation of disproportionality is that they should be easily accessible and directly comparable. This condition increases transparency and contributes to fair comparisons across water bodies. When possible, costs should be extracted from the Programme of Measures of the official RBMP. When unavailable, standard approaches for cost assessment (European Commission 2010) should be used. In our case study, we used the methodology proposed by Tagus RBA (CHT 2018a), although the actual cost could be higher if factors such as land cost are considered. For past expenditure, national statistics are not recommended due to different metrics and the language barrier, which may complicate across-country comparisons. Instead, environmental expenditure reported in Eurostat provide a homogeneous metric. The homogeneity may overweight the sacrifice of more specific national statistics (as used by Klauer, Sigel, and Schiller 2016), although this remains an open discussion. The same consideration can be applied to the current status of the water bodies, where readily available data from the EEA has been preferred to national data for the sake of homogeneity.

The GDP-based approach reflects the capacity of a region to cope with the costs of the measures. GDP is an indicator for public sector budgetary capacity, industrial activity and of household income. These are the actors that are likely to cover the costs of the measures. In light of the Polluter-Pays Principle (Article 9 of WFD), the method allows us to confront the economic capacity of a human activity with the cost of the measures needed to address the associated pollution. If the relative effort is small, then the polluting activity should finance the corrective measures. When the effort is high, then a policy decision should determine whether the activity is to be maintained and the financing effort supported by inter-territorial transfers, or whether environmental exemptions should be applied. In the case of inter-territorial transfers, costs may be considered disproportionate at the level of an individual region, but economically beneficial at a higher level (Pulido-Velazquez et al. 2009; Brouwer 2008). This is particularly relevant when the benefits of an activity are registered far from the area that is affected by the environmental pressure. The alternative to these options would mean that the polluting activity is not sustainable.

5.1. Sensitivity to quality element weighting factors

Further considerations could be added in the averaging of objective distances to reach the effort factor. While the biological indicators could be deemed relatively more important, the degree of certainty of the response of physico-chemical indicators to measures is certainly higher. The evolution of concentration of physico-chemical indicators can be simulated with water quality models (Chapra 2008; Thomann and Mueller 1987). These models include transport and diffusion mechanisms and allow for modeling of chemical reactions of the pollutants of interest for WFD (oxidation of organic matter, nitrification of ammonium, denitrification of nitrate). This is an additional advantage, taking into account the uncertainty about natural processes in water bodies and about how anthropic pressures and corrective measures influence the status of water bodies (European Water Directors 2019). For biological indicators, there is limited evidence that after the application of corrective measures, these indicators will achieve values compatible with good ecological status (Palmer, Menninger, and Bernhardt 2010; Jahnig et al. 2010; Hering et al. 2010; Feld, Segurado, and Gutierrez-Canovas 2016; Segurado et al. 2018).

In Table 3, we assumed that after implementing the WWTP upgrade, all the ecological indicators will achieve good status with a high degree of certainty (included in the factor $QE_j = 1$). On the contrary, we could assume that for the same set of measures there is no evolution of biological indicators (NOD = 0 for QE1-2-4 and QE1-3), or that there is no certainty for this evolution (QE = 0 for these indicators). The qualitative Benefit would become qB = 0.11.

In this case, the disproportionality factor would be:

$$DF = \frac{125.7\ \text{M€/year}}{0.11 \cdot 874\ \text{M€/year}} = 1.31$$

This DF is considerably higher than the value presented in the results section and could change the decision on cost disproportionality.

5.2. Sensitivity to water body weighting factors

An example of the importance of the weighting factors is shown in this section. Equation (3) shows how each water body may have a different relative weight according to the ecological protection types defined in the WFD. Taking the Natura 2000 protection figure as an example, Figure 4 shows the Special Protection Areas (SPA) and Special Conservation Interest (SCI) Zones in our study area (EEA 2019a).

If the water body weighting factor types were chosen to be proportional to the length in kilometers in each special protection area, the resulting weight factor for each water body would be as shown in Table 4.

Equation (4) shows the relative weight of each indicator according to importance and reliability. If it was decided that the reliability of predictions of evolution for chemical, biological, and hydromorphological indicators is too low, and that from physico-chemical indicators the most important are firstly dissolved oxygen and secondly nutrient concentration, the quality indicators and weight factor matrix would be as shown in Table 4.

Applying Equations (1) and (2) would result in a gap of qB = 0.19, and the disproportionality factor would be DF = 0.76. This is the value to be compared with the agreed threshold to assess disproportionality.

These two sections dealing with factor sensitivity highlight the relevance of the consensus to be sought to define the relative importance of quality indicators and water bodies. Otherwise, water policy objectives would be implemented differently in each country following subjective considerations regarding water body size or the importance of protection figures. Furthermore, different approaches to deal with the level of reliability of the pressure-impact-status predictions for each type of indicator would lead to different results, undermining a common goal of European water policy.

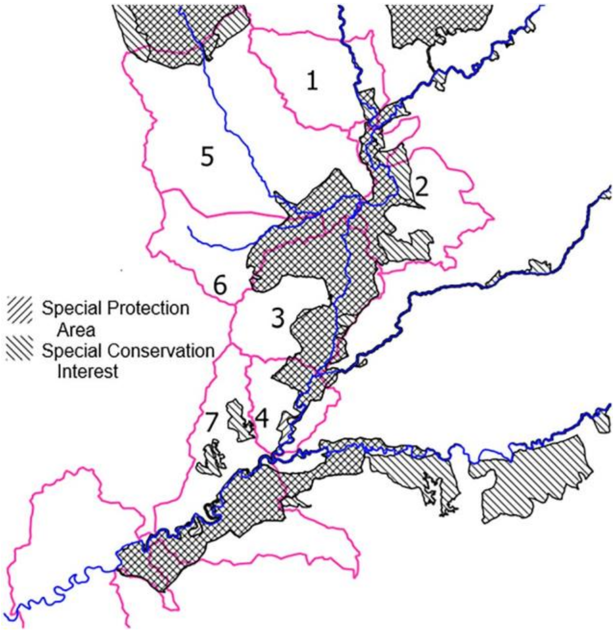

Figure 4.Natura 2000 protection status in the study area. Numerical code defined in Table 1.

5.3. Comparison with NLA approach

The proposed methodology is based on the NLA, which is modified to broaden its geographical scope, add flexibility to the relative weight of the input factors and facilitate the use of existing EU data sources. The results of the original NLA method in our study area are presented here for comparison.

Table 4. Length of water body (km) in special protection areas and weight factors.

				QE1 *QE1-2-4*	QE2 *QE1-3*	QE3 *QE2-3*	QE4 *QE3-1-3*	QE5 *QE3-1-4*	QE6 *QE3-1-5*	QE7 *QE3-1-6-1*	QE8 *QE3-1-6-2*	QE9 *QE3-3*	QE10 *Chem. inds*
Water body	$WB_{i,1}$ SPA	$WB_{i,2}$ SCI	Wbi	0	0	0	0.2	0	0	0.1	0.1	0	0
1 Jarama from Vbebas to Henares	0.07	0.16	0.23	0	0	0	0.05	0	0	0.02	0.02	0	0
2 Jarama from Henares to E. Rey	0.19	0.19	0.38	0	0	0	0.08	0	0	0.04	0.04	0	0
3 Jarama from E. Rey to Tajuña	0.22	0.22	0.45	0	0	0	0.09	0	0	0.04	0.04	0	0
4 Jarama from Tajuña to Tajo	0.17	0.11	0.28	0	0	0	0.06	0	0	0.03	0.03	0	0
5 Manzanares through Madrid	0.15	0.16	0.31	0	0	0	0.06	0	0	0.03	0.03	0	0
6 Culebro stream	0.10	0.10	0.20	0	0	0	0.04	0	0	0.02	0.02	0	0
7 Tagus from Jarama to Toledo	0.40	0.40	0.80	0	0	0	0.16	0	0	0.08	0.08	0	0

The past average public expenditure nation wise (calculated here with environmental protection values due to the lack of specific water protection values) amounts to (Eurostat 2019) 2,344 M€/year / 505,990km^2 = 4,633 €/(year km^2). Considering a catchment area of the concerned water bodies of 2,195km^2, the area-based public expenditure in the past is 4,633 €/(year km^2) · 2,195km^2 = 10.2 M€/year. The required indicators (macroinvertebrates, macrophyten, phytoplankton, fish and Environmental Quality Norms) would give an objective distance of 0.95 and the additional benefits considered in the NLA method (related to terrestrial ecology, freshwater, flood protection, soil protection, and tourism) are estimated at a value of 1, resulting in an effort factor of 0.16 according to this method. The water body-specific cost threshold would be 10.2 M€/year · 0.16 = 1.6 M€/year. Thus, according to the NLA, the cost of the measures (125.7 M€/year) would be disproportionate compared to the threshold (1.6 M€/year).

The differences between both methodologies help in understanding the disparity of the outcome:

- The proposed methodology has been tailored to capitalize on the existing databases available for all the EU Member States. On top of a long tradition of compilation of economic indicators by Eurostat, the EEA has made a considerable effort in recent years to collect and organize water quality data from all the
- Member States. While NLA relies on German databases, the proposed methodology is designed to exploit these European databases and can be directly applied to the geographical scope of the WFD.
- NLA uses country-specific past national expenditure as a reference for future capacity, which may give a comparative advantage to countries with a low level of previous expenditure. The method presented here proposes, instead, the use of Europe-wise past expenditure to guarantee an even playing field between Member States. Since, to date, there is no European database for past expenditure specific to water protection, our method is constrained to using the more general "Environmental protection" as a proxy for past efforts and the political will of the authorities. NLA benefits from the more precise "Sewage treatment" and "Water and land management" statistics, but these are only available for Germany.
- The proposed methodology expresses past average expenditure in terms of percentage of GDP (e spent in environmental protection/e of domestic product), as opposed to the NLA that uses past expenditure per surface unit (€/km^2). The unit used in the NLA (as a unit of affordability and political will) allocates the same resources to a highly populated metropolitan area and an inhabited area, provided they have the same surface area. This would deprive populated regions of corrective resources with respect to the high level of pressures that they exert on receiving waters, prompting the authors of the NLA method to advise against its application in highly populated cities. Renno and Klauer (2018) have also proposed the use of

population instead of area for normalization, in order to allocate greater resources to highly populated areas.

- The objective distance calculation in the NLA only takes account of a particular set of indicators for water body status (namely, biological and Environmental Quality Norms). The proposed method generalizes the objective distance to all the indicators required to characterize the water body, including physico-chemical, hydromorphological and chemical (WFD Annex V, and Guidance Document European Commission 2005). It also allows the application of different weighting factors to each indicator. This difference is particularly important when we consider the central assumption that the water body achieves good status after the application of the measures. The current knowledge on biological indicators does not allow us to predict how they will evolve after the application of the measures (Palmer, Menninger, and Bernhardt 2010; Jahnig et al. 2010; Hering et al. 2010; Feld, Segurado, and Gutierrez-Canovas 2016; Segurado et al. 2018). On the contrary, the evolution of physico-chemical indicators (dissolved oxygen and nutrient concentration) can be predicted to a fair extent using water quality models (Chapra 2008; Thomann and Mueller 1987). The proposed method allows us to reduce the relative weight of indicators whose evolution cannot be predicted. The proposed method is equivalent to the NLA in the particular case where all water bodies have the same weight and only biological indicators are considered, with equal relative weight.
- The proposed method does not include the Additional Benefits factor of the NLA. The calculation of this factor is prone to some amount of discretionality, as noted by Klauer, Schiller, and Sigel (2017), who present their results for the whole range of possible values of additional benefits. Our method proposes, instead, including only objective and readily available data for the calculation of
- the disproportionality factor, leaving additional benefit considerations for the following phase of critical evaluation of the results.
- The NLA method proposes an expenditure threshold of 50% of previous expenditure, while the proposed method recommends the threshold decision to be discussed and established at European level.

5.4. Comparison with affordability analysis

Further insight can be gained by comparing our results with existing affordability analysis. The use of GDP as an aggregate indicator may have some drawbacks when the lion's share of the burden is carried by households. In this case, our method can be compared with household disposable income, as proposed by the OECD (2009). Considering that the coefficient of correlation between GDP [nama_10r_2gdp] and household income [nama_10r_2hhinc] (€/inhab) is as high as 0.8 for Europe NUTS 2 regions (Eurostat 2019), the results should be similar. Further considerations would be

needed to assess how household income would be affected by policy decisions to finance corrective measures through water tariffs, or indirectly through taxes.
In our study area, the average annual income (nama_10r_2hhinc indicator by Eurostat 2019) for 2015 was 20,700 €/inhab. The current cost of water and sanitation of 2.07 €/m^3 for an average daily use of 214 liters/inhab (INE 2018b) represents 162 €/inhab per year, i.e., 0.78% of the disposable income. Without including taxation considerations, the 125.7 M€/year that the proposed measures cost would mean an additional tariff of 25 €/(inhab·year), resulting in a final bill of 187 €/(inhab·year), i.e., 0.90% of the disposable income, well below the upper threshold of 5% recommended by the OECD. The use of affordability arguments is widely accepted for the justification of deadline extensions under WFD Article 4.4, but not for the definition of less stringent objectives under WFD Article 4.5 (European Commission 2009).
Affordability and the proposed method have in common that both confront measured costs with the economic capacity of the region that would finance them. Affordability analyses, however, lack an explicit indicator to quantify the actual environmental improvements that are sought with the implementation of the measures.

6. Conclusion

The final decision to declare a disproportionate cost remains a political one. Scientific studies contribute with methods and data but cannot evaluate all the factors involved. We propose a methodology explicitly designed to facilitate input data compilation, align benefit quantification with the objectives of the Water Framework Directive, and ease comparison between case studies. A Disproportionality Factor scalar is calculated as a one-dimensional projection of the information involved. The methodology is designed focusing on the following points:

- The calculation of the study area's gap to good status is aligned with the water status indicators reported by Member States to the European Commission.
- Eurostat and European Environment Agency data is preferred to national statistics to ensure accessibility and homogeneity of the input data.
- The reference for the disproportionality threshold is calculated from panEuropean past expenditure, to ensure that the same yardstick is used for all river basin districts.
- Past expenditure is expressed as a percentage of local GDP in the study area, to take into account demographic and economic production concentrations.
- In a pressure-impact-status analysis, the method establishes a benchmark to devise whether a polluting activity generates enough GDP to finance the required mitigation measures. When the result is positive, costs are not disproportionate, and the PolluterPays Principle should be applied. On the contrary, when the Disproportionality Factor is high, the cost of the corrective measures is unattainable for the polluter. Further research could study the available options: Water Authorities could decide to accept less stringent requirements, or policy makers

may implement an inter-territorial transfer. If none of these was considered acceptable, the polluting activity should be discontinued.

In our case study, the proposed method shows that the costs of the required measures, albeit high, may not be disproportionate when compared to previous spending of EU Member States and the current capacity of the affected region.

The proposed method can provide reference information for political decisions. The kind of data required facilitates its use in the geographical scope of the Water Framework Directive. Nonetheless, the results of the method depend heavily on the assumed evolution of the environmental indicators in response to measures and on the particular values that are decided for the water body and quality element weighting factors. Future research should focus on the implementation of the method in other study areas, and to build up consensus at a European level on the values of the weighting factors that measure the gap between current and good status of water bodies.

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Disclosure statement

The authors do not have any financial interest in or benefit from the direct application of their research. There is no conflict of interest.

Supplemental data

Supplemental data for this article can be accessed at:
https://www.tandfonline.com/doi/full/10.1080/09640568.2020.1726731
The data that support the findings of this study are openly available in WISE-WFD database at https://www.eea.europa.eu/data-and-maps/data/wise-wfd-3, and the Eurostat database at https:// ec.europa.eu/eurostat/data/database.

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Supplementary Material 1

The estimated cost of the measures is calculated in Table 5, using the methodology proposed by Tagus RBA (CHT, 2018a).

WWTP	***Water body***	Authorised Effluent Volume (m3/day)	Infrastructure cost (Meur)	Annual Infrastructure cost (Meur/year)	Annual Operation cost (Meur/year)
Valdebebas	1 Jarama from Vbebas to Henares	32 047	9.6	0.7	2.0
Rejas	1 Jarama from Vbebas to Henares	69 705	20.9	1.5	4.3
Torrejón De Ardoz	1 Jarama from Vbebas to Henares	31 271	9.4	0.7	1.9
Casaquemada	1 Jarama from Vbebas to Henares	55 570	16.7	1.2	3.5
Velilla De San Antonio	2 Jarama from Henares to E. Rey	19 868	6.0	0.4	1.2
Soto Gutierrez	3 Jarama from E. Rey to Tajuña	23 148	6.9	0.5	1.4
Viveros	5 Manzanares through Madrid	88 927	26.7	1.9	5.5
La China	5 Manzanares through Madrid	160 241	48.1	3.4	9.9
La Gavia	5 Manzanares through Madrid	79 136	23.7	1.7	4.9
Butarque	5 Manzanares through Madrid	117 456	35.2	2.5	7.3
Sur	5 Manzanares through Madrid	293 638	88.1	6.2	18.2
Sur Oriental	5 Manzanares through Madrid	31 211	9.4	0.7	1.9
Culebro Alta	6 Culebro stream	57 389	17.2	1.2	3.6
Culebro Baja	6 Culebro stream	69 390	20.8	1.5	4.3
El Endrinal	Not in the study area	36 363	10.9	0.8	2.3
Arroyo De El Soto	Not in the study area	33 926	10.2	0.7	2.3
Arroyo La Reguera	Not in the study area	23 932	7.2	0.5	1.5
Galapagar-Tlodones	Not in the study area	15 879	4.8	0.3	1.0
Chaparral (Guadar.)	Not in the study area	15 874	4.8	0.3	1.0
La Poveda	Not in the study area	14 612	4.4	0.3	0.9
Arroyo El Plantio	Not in the study area	14 379	4.3	0.3	0.9
Guadarrama Medio	Not in the study area	14 046	4.2	0.3	0.9
Navalcarnero	Not in the study area	13 688	4.1	0.3	0.8
Tres Cantos	Not in the study area	13 519	4.1	0.3	0.8
Arroyo Valenoso	Not in the study area	11 992	3.6	0.3	0.7
Boadilla Del Monte	Not in the study area	10 564	3.2	0.2	0.7
Guatén (T. De Velasco)	Not in the study area	9 094	2.7	0.2	0.6

Table 5. Cost of WWTP upgrade

Giving a total cost (infrastructure annualized plus operational) of 113 million euros, which corresponds to the cost declared by Tagus RBA (CHT, 2018a). The part corresponding to the WWTPs in our study area adds up to 94 million euros.

The costs have been calculated using an analysis of the cost of previous interventions (CEDEX 2012). The Net Present Value of the infrastructure cost varies between 185 and 398 €/(m^3/day), and the value 300 €/(m^3/day) has been retained by Tagus RBA. The Annual Equivalent Cost of this capital expenditure is calculated according to the formula proposed by the European Commission (EC 2003).:

$$AEC = \frac{NPV * DiscountRate}{(1-(1+DiscountRate)^{-lifetime})}$$

AEC = annual equivalent cost
NPV = net present value of investment
Discount rate = chosen discount rate (the same as used to calculate the NPV)
Lifetime = lifetime of the capital equipment

Tagus RBA used a discount rate of 0.75% and a lifetime of 15 years, as suggested by (CEDEX 2012). Operation and maintenance costs vary between 0.14 and 0.20 €/m^3, and the average value of 0.17 €/m^3 is retained. When applied to our study area, this method gives an estimated cost of 94.1 M€/year for the measures, leading to good status of the receiving waters.

It must be noted that the discount rate used is particularly low, compared with the European Commission's recommendations (EC 2008) of 5%. Using the European benchmark value, the annualized estimated cost would rise to 102.7 M€/year.

Since this is expressed in 2006 euros (CEDEX 2012), the inflation adjusted value for 2018 is 125.7 M€/year (indicator prc_hicp_midx, Eurostat 2019).

8. Comparativa de casos de exenciones a los objetivos medioambientales

8.1. Introducción

La Directiva Marco del Agua establece unos objetivos ambiciosos para las aguas comunitarias, pero permite ciertas excepciones en casos particulares, como se ha expuesto en el capítulo anterior. El nivel de exigencia para establecer estas excepciones debería ser común en el ámbito geográfico de la aplicación de la DMA, pero aún no existen herramientas para garantizar dicha homogeneidad.

Para contribuir a paliar esta carencia, en este capítulo proponemos una metodología que estudia el nivel de impacto de presiones a partir del cual se han establecido exenciones en el Tajo Medio y lo compara con otras cinco demarcaciones europeas (Figura 6). Al no disponer de datos de costes de las medidas (no hay bases de datos centralizadas a nivel europeo), el estudio se centra en el beneficio potencial de las medidas. Con ello se pretende establecer si las autoridades de cuenca de las demarcaciones han aplicado criterios similares para declarar exenciones o, por el contrario, cada una ha aplicado un nivel de exigencia distinto.

Debido a la amplitud geográfica del estudio, que abarca a más de 1400 masas de agua, y a la dificultad de conseguir datos de presiones para otros contaminantes, el análisis se restringe al nitrógeno, que constituye uno de los contaminantes físico-químicos de mayor incidencia en Europa (Grizzetti et al. 2017).

8.2. Metodología

Para comparar las políticas de exenciones de las demarcaciones, primeramente se caracteriza el efecto de las presiones sobre las masas de agua mediante un modelo que cuantifica la contaminación puntual, la difusa, y el caudal circulante.

La contaminación puntual se calcula a partir de las bases de datos publicadas por la UE (European Commission 2019b) sobre habitantes equivalentes de diseño de cada depuradora, así como los procesos de depuración aplicados. Mediante estimaciones de la carga contaminante influente por habitante y el porcentaje de reducción de nitrógeno de cada proceso se cuantifica la presión ejercida a través del efluente de las depuradoras.

En el caso de la contaminación difusa, se recurre a modelos estadísticos de emisión de nitrógeno (Grizzetti and Bouraoui 2007) combinados con modelos de evolución del contaminante en la fase terrestre (Munafò et al. 2005). Dichos modelos, basados en algoritmos de Sistemas de Información Geográfica, describen el efecto de la escorrentía superficial y la distancia al cauce más cercano en el transporte y evolución del nitrógeno en su fase terrestre.

Una vez las presiones llegan a las masas de agua, su impacto real depende de la cantidad de nitrógeno respecto al caudal circulante (ya que los límites se definen en términos de concentración), así como de la evolución del contaminante en las aguas receptoras. Para lo cual se construye un modelo de transporte y degradación del nitrógeno en el cauce, basado en la metodología expuesta en el capítulo 5.

Con los datos introducidos, el modelo reproduce una concentración media de nitrógeno para cada masa de agua de las cuencas de estudio. Al ser conocidas las exenciones aplicadas a cada una de estas masas de agua (European Environment Agency 2019b), seguidamente se establece una correlación estadística entre los impactos de las presiones y la aplicación o no de exenciones. Lo cual permite comparar los niveles a partir de los cuales se declaran exenciones en cada demarcación.

8.3. Resultados y discusión

El resultado del estudio nos muestra que el nivel a partir del cual se declaran exenciones es similar en la mayoría de las demarcaciones estudiadas, y es cercano a la media (Poikane et al. 2019) del límite entre el estado bueno y el estado moderado (del orden de 2,5 mgN/l en ríos, y 1 mgN/l en lagos). Las demarcaciones más septentrionales estudiadas, sin embargo, aplican exenciones a partir de umbrales más pequeños (es decir, con mayor permisividad). El hecho de que el límite sea tan bajo implica que un gran porcentaje de las masas de agua tienen exenciones, lo cual no ayuda a alcanzar el equilibrio medioambiental requerido por la DMA.

La capacidad de mostrar cuantitativamente los niveles de impacto de las presiones existentes sobre cada masa de agua y relacionarlos con las exenciones aplicadas abre la posibilidad de establecer comparaciones cuantitativas entre demarcaciones. Esto permite arrojar luz sobre la política de declaración de exenciones en la UE y sobre las implicaciones de una aplicación desigual de esfuerzos de mitigación de presiones.

8.4. Conclusiones

En vista de los resultados podemos concluir que se necesita avanzar en la definición de límites coherentes para establecer exenciones al buen estado de las masas de agua superficiales continentales. Además, si se pretende que el número de masas de agua con exenciones represente un porcentaje mínimo del total (para no comprometer los objetivos de la DMA), se debe aplicar un umbral medio más exigente que el actual.

8.5. Artículo

A continuación se reproduce, con el permiso de los coautores, el contenido del artículo:

Antonio Bolinches, Javier Paredes-Arquiola, Alberto Garrido, Lucia De Stefano (2020) A comparative analysis of the application of water quality exemptions in the European Union: The case of nitrogen. Science of the Total Environment. DOI: 10.1016/j.scitotenv.2020.139891

A comparative analysis of the application of water quality exemptions in the European Union: the case of nitrogen

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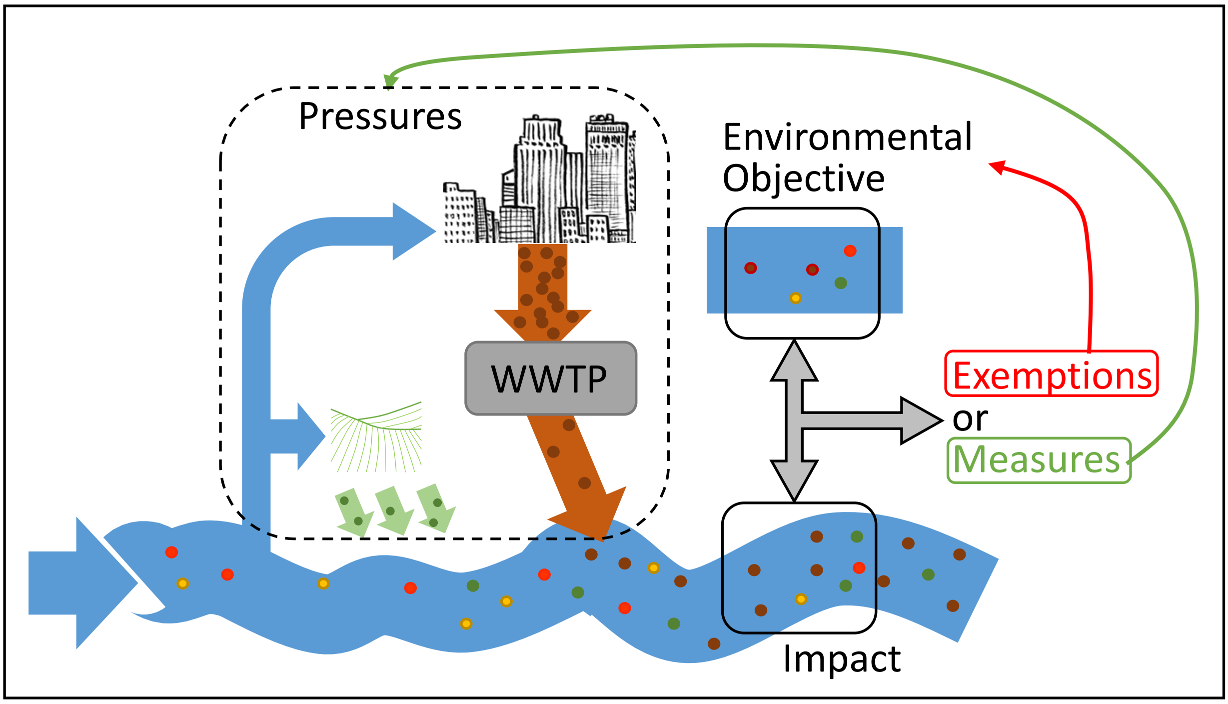
Graphical abstract



Highlights:

- Exemptions may compromise the fair distribution of efforts to achieve the good status of Europe's water bodies.

- We present a tool to compare different exemption applications to nitrogen concentration in continental surface waters.

- Diffuse and point source pressures and impacts are assessed against the application of exemptions.

- A better understanding of pressures and impacts by the Water Authorities will support the justification of exemptions and prioritization of measures.

Abstract

Freshwater ecosystems and water use may be jeopardized by the degradation of water quality. The Water Framework Directive of the European Union (EU) sets environmental objectives for water bodies but foresees the establishment of exemptions under some circumstances. The criteria used to justify these exemptions, however, are not fully developed, leaving their application open to some arbitrariness. Our study explores the relations between the magnitude of pressures affecting continental surface water bodies and the declared exemptions on the permitted concentration of nitrogen. It identifies different approaches to declare exemptions to nitrogen environmental objectives across six EU Member States and discusses the underlying criteria. A better understanding of the pressures-impact-measures/exemptions relation helps compare water policy decisions across different regions subject to the same legal obligations and set priorities for mitigation measures.

Keywords: Water Framework Directive, Exemptions, Nitrogen, Nutrients

1. Introduction

In the last century water quality in rivers has deteriorated globally (United Nations Environment Program 2015), mostly as a consequence of anthropogenic pollution (Ward and Elliot 1995) and hydromorphological alterations (Petts 1984; World Commission on Dams 2000). This applies also to the European Union (EU), where the main impact on surface waters comes from nutrient enrichment, especially nitrogen (Grizzetti et al. 2017) and the alteration of hydromorphological dynamics (EEA, 2018). Nitrogen in its uncombined form is the main constituent of Earth atmosphere with more than 75% of its mass (Haynes 2016), but is unattainable as a nutrient for most plants (Foth 1990). However, it is found in its bound form only as a trace element in the earth crust and the ocean, with an average concentration below 50 ppm (Haynes 2016). Among other reasons, the need to increase the availability of reactive bound nitrogen as a basic nutrient for plants' growth led to the synthetization of ammonia from its uncombined atmospheric form in the beginning of the twentieth century (Haber 1920). The increase in the availability of bound nitrogen is at the root of the continuous increase of agricultural production and the possibility to meet the needs of the growing world population (Smil 2000; Stewart et al. 2005). A large proportion of this nitrogen is lost to the environment in the process (UNEP and WHRC 2007), thereby increasing the nitrogen concentration in the continental waters that drain the excess not absorbed by crops after fertilization or excreted by the ever growing human population and livestock (Galloway et al. 2003; Erisman et al. 2008). Nitrogen reaches the surface continental waters in various fractions, mostly organic nitrogen from manure and human waste, ammonium and nitrate; organic nitrogen may be mineralized to ammonium, which in turn may be nitrified to nitrate with oxygen consumption (Thomann and Mueller 1987). Negatively charged nitrate is particularly mobile in soils (Haygarth and Jarvis 2002). An accumulation of these fractions may trigger an excessive growth of aquatic plants (eutrophication), which is one of the most important water quality problems in Europe (European Commission 2009b) and worldwide (United Nations Environment Program 2015). In anaerobic conditions, nitrate may denitrify to gaseous nitrogen, which is liberated to the atmosphere (Thomann and Mueller 1987).

In order to protect aquatic ecosystems and freshwater abstractions, legislation often prescribes maximum concentration of nitrogen and other pollutants in natural waters. In the case of the EU, the Water Framework Directive (WFD) sets the environmental objectives for water bodies. Reference conditions are defined, and water status categories (high, good, moderate, poor and bad) are set with respect to these references. The WFD's goal is to reach a "good" status for all waters, i.e. slight deviations from near-natural conditions resulting from anthropic activity. The status of a surface water body is determined by the poorer of its chemical status and its ecological status, which in turn depends on biological, hydromorphological and physico-chemical elements. Nitrogen is one of these physico-chemical elements contributing to the ecological status of water bodies.

The WFD asks for common definitions of the status of water in terms of quality, and the European Commission (EC) has issued several guidance documents (EC 2003e, EC 2015) in the frame of its Common Implementation Strategy to promote the intercalibration of indicators. However, this process focused on the biological elements

(Birk et al. 2013) rather than on the physico-chemical elements supporting them (Poikane et al. 2019). As a result, there is a wide range of nutrient thresholds for similar water body types; this prompted the EC to foster the definition of consistent nutrient concentration targets for all MS, to ensure a coherent and harmonious implementation of the WFD (Poikane et al. 2019). These authors found that MS defined nutrient limits using different metrics (annual mean, median, 90th percentile, etc.), nitrogen fraction measured (total nitrogen, nitrates, ammonium, etc.) and criteria (modelling, regression between nutrient and biological response, expert judgement, etc.).

Over 60% of continental surface waters present conditions below the required good status (European Environment Agency 2019c), and the impact of nitrogen pollution is one of the stressors behind this infringement. The achievement of good status has strong budgetary implications. For each water body failing to meet its environmental objectives, the WFD requires from the River Basin Authority to design the measures required to remediate the infringement and compile them in a Programme of Measures. MS have to provide the necessary financial resources for these interventions.

1.1. Exemptions to the environmental objectives

When legally-binding environmental objectives are set, there can be cases where the corrective action needed to achieve them may be technically challenging or financially overburdening. The equilibrium between higher water quality standards and economic affordability is a complex task for Water Authorities worldwide, and the adequate standard will differ between developed and developing economies (UN-Water 2015). In the EU the WFD allows for the declaration of exemptions when these can be justified, thus relieving the financial burden of the Programme of Measures. Although exemptions are not explicitly defined in the United States (US) Federal Water Pollution

Control Act (US Congress 2002), the US Environmental Protection Agency offers a broad discretion to define remedial action when increased costs would be wholly disproportionate to potential benefits (Coplan 2018). In the case of China, a substantial increase in the water quality protection levels of the Drinking Water Quality Law (Ministry of Health of the People's Republic of China 2006) let many regions struggling to meet the requirements since local governments could not afford the implied costs (UN-Water 2015).

The exemptions foreseen by the WFD may be applied in the form of time extensions to the achievement of objectives or in the definition of less stringent environmental objectives. By the time the Directive was passed, it was accepted that the use of exemptions needed to be explained further and the rules for application had to be better defined (EC 2009a). Twenty years after the WFD approval, the exact conditions to declare exemptions remain unclear and have been applied throughout the different River Basin Districts (RBDs) with poor or debatable justifications (Maia 2017). Divergence of opinion about the exemptions during the negotiation of the WFD led legislators to use a vague language in the final text (Boeuf, Fritsch, and Martin-Ortega 2016), and methodological challenges - mainly related to the monetization of net benefits (EC 2009a) - have hindered further developments. This circumstance adds a layer of arbitrariness to the permitted environmental conditions of European waters.

According to the EU Guidance Document (EC 2009a), exempting from the good status objectives should be the exception rather than the rule. But the fact is that MS have frequently invoked WFD clauses to obtain exemptions. Figure 1a shows that 59 %

of surface water bodies have some sort of exemption to their environmental objectives (European Environment Agency 2019c).

The WFD Reporting Guidance (EC 2016) establishes that MS should report the exemptions for each water body and each quality element associated to the ecological status. For physico-chemical quality elements, required entries include declarations of exemption (if any) for oxygenation, nitrogen, phosphorus and thermal conditions among others.

In the case of nitrogen limits (figures 1b and 2) MS have declared exemptions for 7% of water bodies. Nevertheless, the presence of unpopulated databases (Germany and Greece) and of countries with apparently no declared exemptions to nitrogen environmental objectives (suggesting an error in database value assignation) suggests that the actual figure may be higher. The main reasons alleged (reported as significant impacts on water bodies) are the pollution pressure in 44% of the water bodies and the hydrological or morphological alterations in 22% of the cases (European Environment Agency 2019c).

The WFD lists the conditions to be met in order to declare exemptions to the environmental objectives. In this paper, we focus on the clauses of 'disproportionate cost' and 'technical infeasibility'. The distinction between these clauses may not be neat since "in practice, the greater the effort expended in trying to overcome practical issues of a technical nature, the greater the likelihood that technically feasible ways of making the improvements will be found" (EC 2009a). This suggests that in cases where large benefits are expected, greater efforts should be made to work out a technical solution.

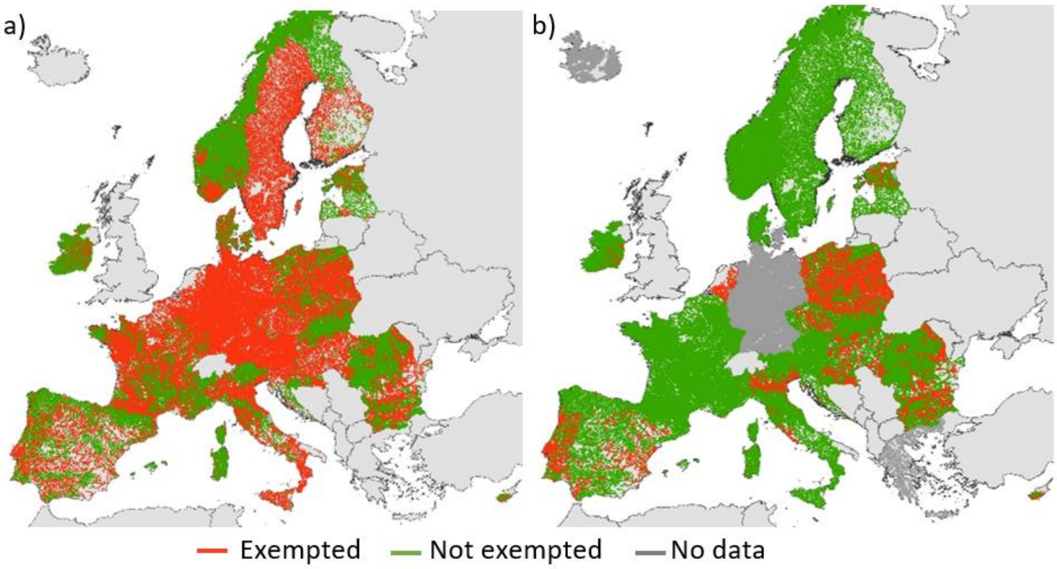


Figure 1. Geographical distribution of European continental surface water bodies with exemptions to (a) at least one environmental objective and (b) nitrogen environmental objective (EEA code QE3-1-6-1).

Several methods have been proposed in the scientific literature to justify exemptions, in particular for the cost disproportionality clause (Lago, Moran, and MacLeod 2006; Klauer, Sigel, and Schiller 2016; Bolinches, De Stefano, and Paredes-Arquiola 2020b). Some authors have undertaken the task of understanding how exemptions should be granted and compare the use of different tools for that purpose (Boeuf, Fritsch, and Martin-Ortega 2018; Klauer, Schiller, and Sigel 2017; Macháč and Brabec 2018), or investigated the regulatory efficiency of the exemption clause (Lago 2008). Although methodologies are diverse, they all share the setting of a threshold above which exemptions are deemed to be acceptable. It is desirable for this threshold to be uniform across the RBDs to guarantee a fair distribution of efforts in the EU and to ensure that the recommendations of the EC to achieve consistency to pollutant limits are not bypassed by the arbitrary application of exemptions. However, we did not find studies in the literature that confront the application of exemptions in different EU regions in order to assess the uniformity of this threshold.

The evaluation of the different pressures, their interactions and the effects on receiving waters is a difficult task, especially in the case of biological indicators (Palmer, Menninger, and Bernhardt 2010; Jähnig et al. 2010; Hering et al. 2010). However, it may be addressed with an acceptable degree of certainty for physico-chemical elements, where anthropic pressures - point loads of wastewater treatment plants (WWTP), diffuse loads emanated by different land uses - and the evolution in receiving waters can be modelled (Chapra 2008).

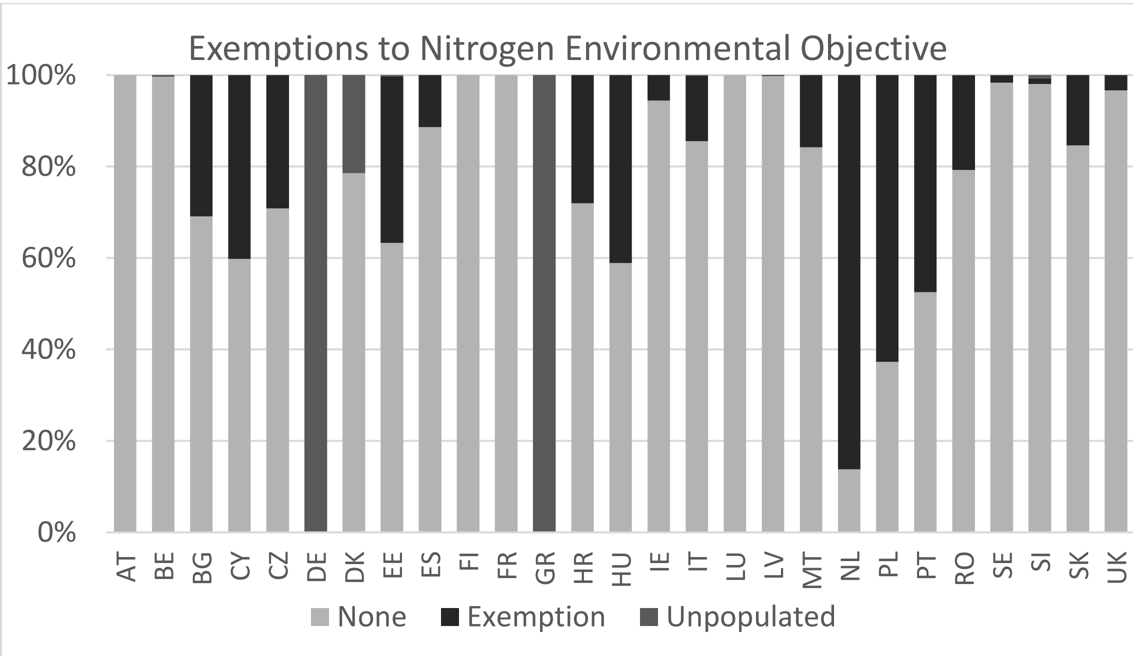


Figure 2. Percentage of exemptions to the nitrogen environmental objective (QE3-1-6-1) for continental surface water bodies in the EU.

1.2. Aims and scope of the paper

In this paper we match the level of nitrogen pollution pressures on river-type water bodies with the declaration of exemptions to nitrogen environmental objectives across MS. The goal is to check the consistency of the level of the indicator with the decision to file an exemption. The results of the analysis can be used by the RBD authority to prioritize future action on exempted water bodies or by the EC to guide inspections and ensure an adequate WFD implementation. The study seeks to contribute to the definition of the exemption threshold for the nitrogen pollution through the quantification of the expected net benefits of each mitigation measure.

For this purpose, we analyzed over 1400 water bodies in the RBDs of six different Member States of the EU (Estonia, Ireland, Czech Republic, Italy, Spain and Portugal). Each water body is characterized according to the level of nitrogen pressures derived from point and diffuse sources, and the declaration or not of exemptions to the environmental objectives of this nutrient. The impact level above which exemptions are applied is assessed for each RBD and the different approaches to this significant aspect of the WFD implementation is discussed.

2. Materials

2.1. Study area

The study area covers six RBDs in the geographical scope of the European WFD (figure 3): (1) Vouga, Mondego and Lis in Portugal, (2) the central area of Tagus basin in Spain, (3) Liffey basin in Ireland, (4) Central Apennines in Italy, (5) the Svratka and Morava basins in the Dunaj-Danube RBD in Czech Republic and (6) Laane-Eesti Vesikond in Estonia. These basins have been selected to include different Member States, climates, and geological and socioeconomic settings. The selection has been constrained by the need of a complete set of information on the declared exemptions on nitrogen objectives (indicator QE3-1-6-1 of EEA WISE database) of the surface continental water bodies. Among the RBDs with sufficient information, the catchments to be studied were selected to illustrate trends in different conditions: Atlantic climate in the Vouga-Mondego-Lis, high urban loads and water scarcity in the Middle Tagus, high natural runoff in the Liffey basin, Mediterranean climate in the Apennines, central European continental conditions in the Svratka-Morava basins and boreal climate in the Laane-Eesti.

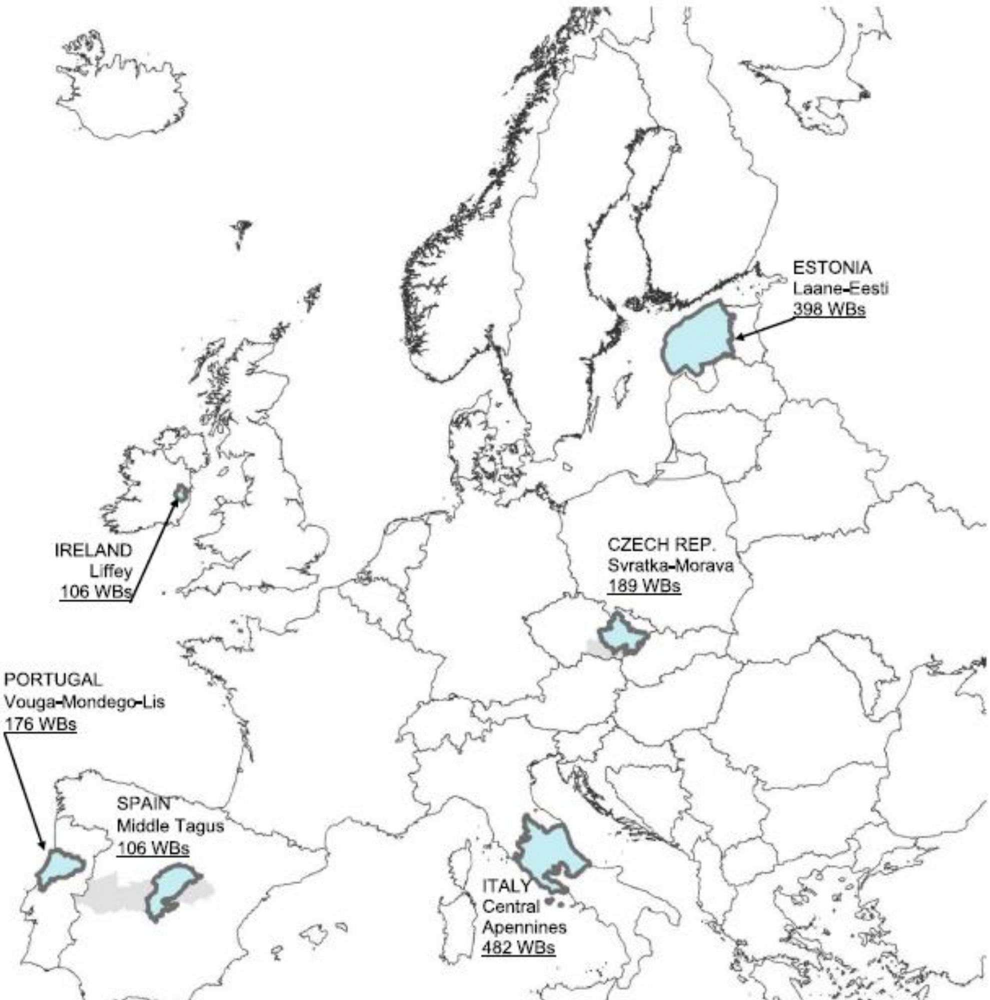


Figure 3. River Basin Districts under study. WB: water body.

Figure 4 shows the percentage of surface water bodies with exemptions to the nitrogen environmental objective declared in each catchment in our study area, with a range between 10 and 38% of exempted water bodies.

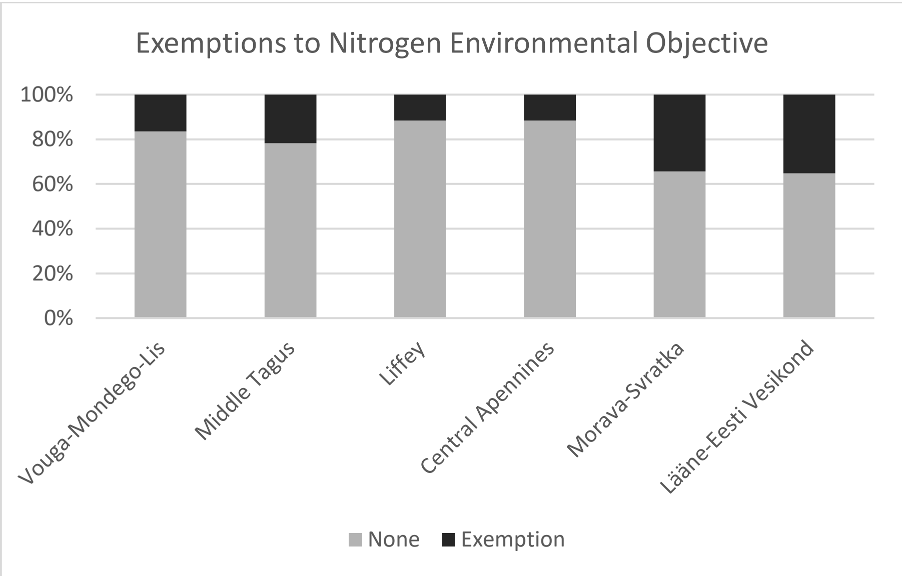


Figure 4. Percentage of surface water bodies with exemptions to the nitrogen environmental objective (QE3-1-6-1) in the study area.

2.2. Data

The data used in this study have been extracted from the following databases:

- Copernicus Digital Elevation Model (DEM), 25x25 m tile size (Copernicus Land Monitoring Service 2019). The DEM serves for the delimitation of the watershed of each water body and the length from each point of the study area to the closest water stream, following a line of maximum slope.

- Corine Land Cover, 100x100 m tile size (Copernicus Land Monitoring Service 2018). The land use raster is used in the calculation of nitrogen load emission.

- Geographical distribution of continental surface water bodies (EEA 2019d). The shapefile supports the definition of upstream/downstream relation among water bodies, and the assignation of length used in pollution degradation.

- Size and quality of treatment of existing urban waste water treatment plants (WWTP) discharging to the water bodies in the study area (EC 2019b; EEA 2019a). This database is used to quantify the point source pressures on the surface water bodies.

- Nitrogen diffuse emissions on surface waters, resulting from agriculture, atmospheric deposition and scattered dwelling (Grizzetti and Bouraoui 2007). Combined with the spatial distribution of watersheds, this map contributes to the calculation of aggregate diffuse loads on each basin.

- Gauged flows (EEA 2019c; CEDEX 2016). Since water quality limits in continental water bodies are expressed in the form of concentration of pollutants, an indication of water flow for each water body is needed to calculate the nitrogen concentration in the model.

- Average annual runoff map, with 10km tile resolution (EEA 2012a). This map enables the calculation of flow in water bodies lacking a gauging station. It also supports the calculation of the transport of diffuse loads in the land phase.

- Water body status and declared exemption databases, updated in 2016 (EEA 2019c).

The data are compiled to calculate average annual values for the pressures on the water bodies of the study area. In the areas where databases do not cover the watershed (nitrogen emissions and runoff for Laane-Eesti Vesikond), values are extrapolated from the closest area with available data.

3. Methodology

In line with the recommendations of the WFD and the EC guidelines (EC 2003b), the proposed assessment follows a four-step approach (figure 5):

S1) The driving forces (land use, agriculture activities, urban development) are described.

S2) The pressures are identified and quantified. In our case, these pressures are expressed as the mass of nitrogen per year reaching the water bodies (tN/year).

S3) The impact on each water body is assessed. Since we are concerned with the concentration of nitrogen in the receiving waters, this impact will be expressed as mass per circulating flow (tN/hm^3).

S4) The likelihood of failing to meet the objective (and eventually to declare an exemption) is evaluated.

The simulation of the impact of each pressure is undertaken through a numerical transport and fate water quality model. Nitrogen sources are identified and evaluated with a combined approach for point and diffuse sources. We model the mechanisms of evolution of nitrogen in the land phase until it reaches a water stream, and its evolution within the stream itself. Given that the environmental objectives for nitrogen pollution are expressed as a concentration, the modelled impact is combined with information on the in-stream flow.

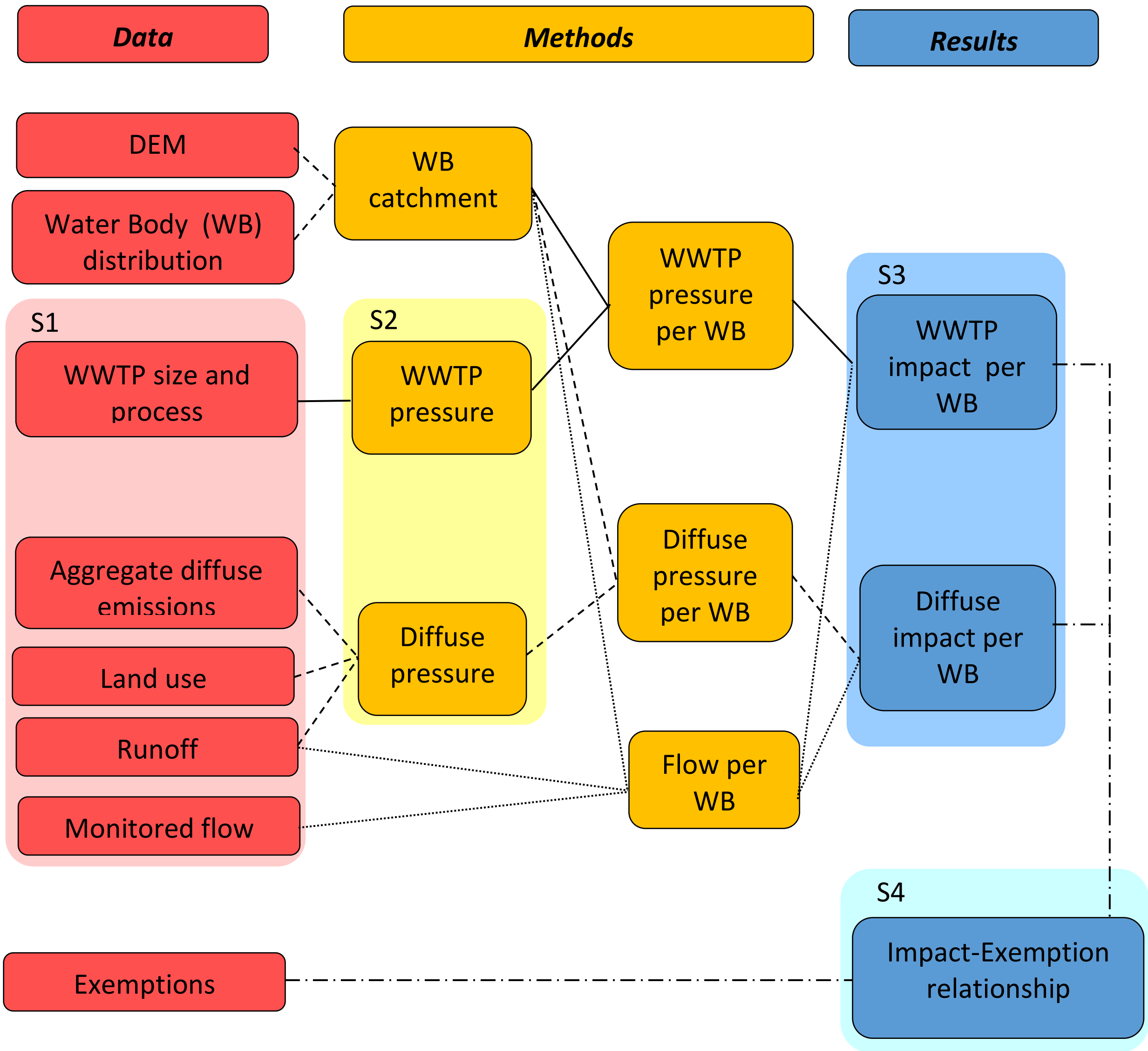


Figure 5. Calculation steps. Line type code: solid line for point source calculations. dashed line for diffuse source, dotted line for flow and dash-point line for exemptions.

Our model uses the water bodies as discretization units of the continental waters. A water body is defined in the WFD as a "discrete and significant element of surface water"

(Art. 2.10), its main purpose being to enable the status of the water body to be accurately described and compared to environmental objectives (EC 2003a).

3.1. Diffuse load pressures

The lack of suitable methods to estimate diffuse nitrogen load onto each surface water contributes to jeopardize the achievement of the WFD environmental objectives (Yang and Wang 2010). Existing legislation on nitrogen emissions, in particular the European Directive concerning the protection of waters against pollution caused by nitrates from agricultural sources (Council of the European Communities 1991b), mandates that nitrogen surplus reaching surface and groundwater bodies should be monitored. Average livestock density in the last reported period (2012-2015) has decreased by 2.6% with respect to the previous period (2008-2011), while mineral nitrogen fertilizer use has increased a 4% (EC 2018b). Only 12 countries reported the nitrogen balance, showing an average decrease of 3% of nitrogen emission. The Directive does not limit emissions, and reporting is not fully comprehensive. Therefore, nitrogen emission from diffuse sources cannot be measured with precision and existing studies turn to statistical descriptions of emissions of different soil types (Ongley 1996; Lin 2004; Elrashidi et al. 2004; Yang and Wang 2010).

Among the existing methods in the literature, fully distributed GIS-based models (Zhang and Huang 2011; Munafò et al. 2005) present the best geographical accuracy potential (El-Nasr et al. 2005), and they can be as accurate as the input data. Nonetheless, semi-distributed models (R. A. Smith, Schwarz, and Alexander 1997; Grizzetti, Bouraoui, and De Marsily 2008) lumping parameter values to sub-basin scale are easier to calibrate given the discrete and scarce nature of water quality monitoring networks.

The first step for the assessment of diffuse load applied to each water body consists in the delimitation of its catchment area. That is, the geographic area where applied nutrients may be drained by runoff to the water body before reaching any other outlet. Previous efforts have been made to define the European catchments, most notably the EEA Catchments and Rivers Network System - ECRINS (EEA 2012b), which was developed as a GIS database delimitating the Functional Elementary Catchments of each river stretch. Yet, river segments were defined following hydrological considerations instead of the WFD water bodies. Therefore, water body catchments for this study had to be calculated based on the DEM (Copernicus Land Monitoring Service 2019) and using a watershed delimitation algorithm based on flow direction considerations (ESRI 2011).

On the one hand, statistical regression models are able to predict the aggregate load of a RBD (R. A. Smith, Schwarz, and Alexander 1997). Some of these, such as the GREEN model (Grizzetti, Bouraoui, and De Marsily 2008) have been calibrated for the geographical scope of the WFD.

On the other hand, GIS-based methods assign a relative weight to the diffuse emission as a function of explanatory variables such as soil use, runoff, distance, etc. (Zhang and Huang 2011). Soil use includes nutrient loads from urban drainage, agriculture fertilizers and forestry among others (EC 2003b). These are often expressed as raster maps. Being data demanding, these methods often require inputs that are not readily available as secondary data. An exception is the Potential Non-Point Pollution Index (PNPI) method (Munafò et al. 2005), which can be directly applied in the EU using the Corine Land Cover, Copernicus DEM and runoff data. PNPI model works under the assumption that diffuse loads reaching water streams depend on three factors: Land

Cover indicator (LCI), measuring load emitted at each portion of the terrain (characterized by its Corine Land Cover nomenclature); Distance Indicator (DI), measured following a curve of maximum slope considering that a polluter will degrade along the land phase before reaching the stream; and Runoff Indicator (ROI), since surface runoff originated by precipitation is considered the main process transporting polluters from soil to surface water bodies.

The PNPI method considers that the nitrogen emission is a linear combination of the three factors. Another characteristic of non physically-based GIS methods is that the output is normally not expressed in physically meaning variables (such as kgN/year). As a matter of fact, PNPI index is dimensionless (Munafò et al. 2005; Cecchi et al. 2007).

Other methods like the Soil and Water Assessment Tool (Arnold et al. 1998; Neitsch et al. 2011) rely on physically-based processes (nitrogen cycle in the land phase, surface and subsurface transport mechanisms) but require an extensive set of hypothesis and data and are not suitable for large scale studies.

In our model, we have chosen a mixed approach. The overall emitted load for each RBD in the study area is first calculated using the calibrated GREEN method (Grizzetti, Bouraoui, and De Marsily 2008). As a result, the aggregate diffuse load on each catchment in the study area is known with an acceptable degree of certainty.

A geographically distributed load index is calculated using the PNPI GIS-based method (Munafò et al. 2005). The hydrology toolset of ArcMap (ESRI, 2011) is used to calculate the DI factor based on the local DEM (Copernicus Land Monitoring Service 2019) and the catchment of each water body. The Spatial Analyst tool is used to reclassify the Corine Land Cover (Copernicus Land Monitoring Service 2018) raster to calculate the LCI factor, and the Runoff map (European Environment Agency 2012a) to calculate the ROI factor. The raster calculator (ESRI, 2011) is used to combine these indicators, which are then integrated within each water body catchment to calculate the load factor. This load factor is then scaled so the total catchment load matches the load predicted by the statistical regression model. With this approach, our method takes advantage of both the geographical distribution of GIS-based models and the aggregate accuracy of the statistical regression models. The result of the calculation is a value of diffuse load in t/year of nitrogen, applied to the midpoint of each water body.

3.2. Point source pressures

Point source pollution from urban waste water is regulated by a Directive (Council of the European Communities 1991c) that requires MS to treat urban wastewaters before the discharge to the environment. The level of the treatment depends on the size of the served population and the protection status of the receiving waters. The applied loads from point sources can be quantified to a higher degree of certainty than diffuse source pressures. For each WWTP, the nitrogen load in the influent is calculated taking into account the following factors:

- The size of each treatment plant expressed in population-equivalent, as declared in the databases of the Urban Waste Water Treatment Directive dissemination platform (EC 2019b).

- The average nitrogen load emitted per person. It is considered to be 12 gN/day/person, according to the existing literature (Grizzetti and Bouraoui 2007; Mulder 2003).

The nitrogen load in the influent is then reduced in the WWTP according to the treatment level (see the considered reduction percentages in Table 1). The Waste Water Treatment Directive only limits nitrogen emission to WWTPs discharging to water bodies sensitive to eutrophication by nitrogen. More stringent requirements can be applied to ensure that the receiving waters meet their environmental objectives. Therefore, nitrogen removal processes may be needed in practice in all the WWTPs where the nitrogen level in the receiving waters is above the established limits.

Table 1. Nitrogen load reduction in WWTPs. Based on Grizzetti and Bouraoui (2007) and Mulder (2003).

Treatment	*Primary*	*Secondary*	*N removal*
Total N reduction (%)	15	30	90

The resulting point source load is therefore calculated in t/year of nitrogen, applied to the midpoint of the water body. It takes into consideration the aggregate sum of nitrogen emissions of all the WWTPs that discharge their effluent in the catchment of each water body.

3.3. Circulating flow

The flow of water circulating (in hm^3/year) is calculated, either by averaging the existing data in the monitoring stations (EEA 2019c; CEDEX 2016), or by aggregation of runoff in the catchment of each water body (EEA 2012a) when no gauging stations are available.

3.4. Transport and fate in streams: assessment of impact on water bodies

The total impact value, i.e. the concentration of nitrogen in the receiving waters (step S3 above), is then estimated. A matrix is created for each RBD to relate each water body to those directly upstream. The total flow on each water body is calculated as the sum of the runoff generated in the catchment of the water body plus the flow coming from upstream water bodies, expressed in hm^3/year.

Previous literature (Thomann and Mueller 1987; Pelletier, Chapra, and Tao 2006; Chapra 2008) characterizes the continuous degradation of a percentage of the existing nitrogen load along the river, which results in an exponential decay. Our model considers that nitrogen quantity evolves along the length of each water body as per equation (1) (Grizzetti, Bouraoui, and De Marsily 2008).

$$N_{outlet} = N_{inlet}\, e^{-\frac{\alpha_L}{L_{max}} L_i} \tag{1}$$

Where α_L is a coefficient calibrated with existing observations and L_i is the water body length. The total load on each water body is then calculated as the load from point and diffuse sources plus the degraded load coming from upstream water bodies.

Each water body is characterized by the pressures and flow in the outlet. Pressures generated in the water body itself or its watershed are assumed to travel on average half the water body length until the outlet, while pressures transmitted in the water body inlet from upstream water bodies (i.e. the summation of the outlet pressures of water bodies directly upstream) will travel and degrade through the full water body length.

Finally, an impact level indicator *Im* is calculated for each water body as the ratio between the applied loads (N_{outlet}) and the circulating flow (Q_{wb}):

$$Im = \frac{N_{outlet}}{Q_{wb}} \tag{2}$$

In this pressure-impact study, the pressure is measured in t/year of nitrogen reaching the surface continental waters. The impact on the water bodies is expressed as the average yearly concentration of nitrogen, in tons of nitrogen per cubic hectometer of flow (t/hm^3 equals the more common unit of mg/l used in the legislation). This equivalence should be handled with some caution, since observed concentrations in mg/l will be subject to high intra-annual variations that are not reflected in this model.

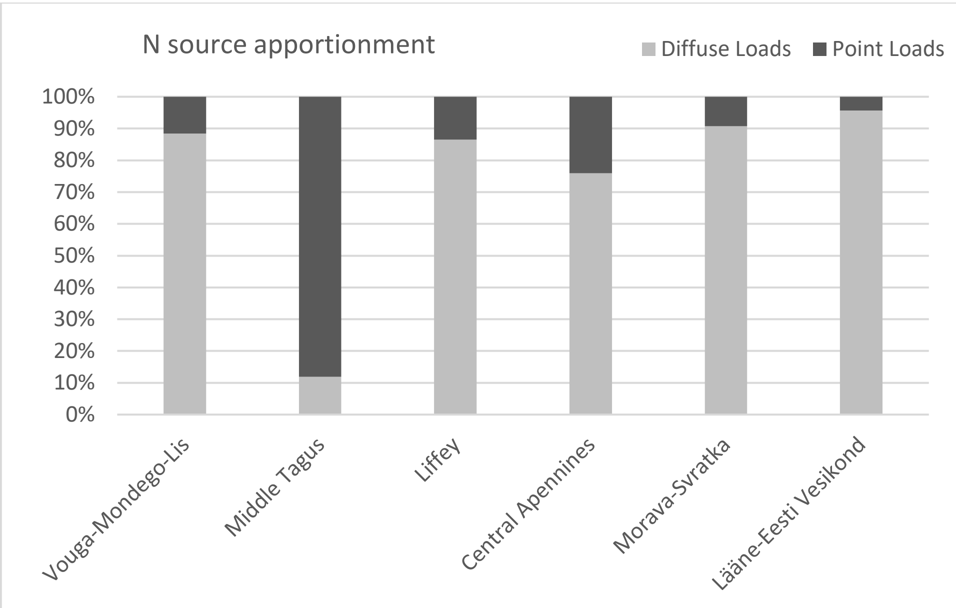


Figure 6. Aggregate pressure from diffuse and point sources in each catchment.

3.5. Comparing the impact with the declaration of exemptions to nitrogen environmental objectives

The estimated impact on each water body (in tN/hm^3) is confronted with the ecological status declared in the official databases (EEA 2019c) and with the declaration of exemptions to nitrogen environmental objectives, which is a water policy decision taken independently by each MS or even by each RBD authority. The study assesses the decisions taken by RBD, water body type (river or lake), load source (diffuse or point) and exemption type (technical feasibility or cost disproportionality, time extension or less strict objective).

4. Results and discussion

The model yields the aggregate level of pressure due to nitrogen in each of the studied RBDs. As shown in figure 6, diffuse source pollution is the main nitrogen pressure in all the catchments of the study area except for the Middle Tagus, where the urban waste waters of more than 6 million inhabitants (INE 2018) largely overshadow any other pressure. This effect in the Tagus RBD was found also in previous studies (Bolinches, De Stefano, and Paredes-Arquiola 2020a). The results are in consonance with the main pressures declared by MS in their WFD implementation reports (European Environment Agency 2019c).

4.1. Impact level vs. ecological status

The correlation between the impact of nitrogen loads and the ecological status of the water body (which is derived not only from nitrogen concentrations but also from other physico-chemical, biological and hydro-morphological elements) is explored in this section.

Figure 7 shows that water bodies with a higher nitrogen load tend to have a lower ecological status, which is in line with the previous study of Grizzetti et al. (2017). The range for each group depends on the dominant ecotype of the rivers in each RBD. In the Vouga, Mondego and Lis basins, water bodies with poor ecological status have a relatively low nitrogen loads. This suggests that in those water bodies other concomitant pressures (e.g. hydromorphological ones) may be affecting the status, in particular the especially sensitive biological elements. The high correlation between ecological status and nitrogen load confirms its condition among the main pressures on surface water bodies.

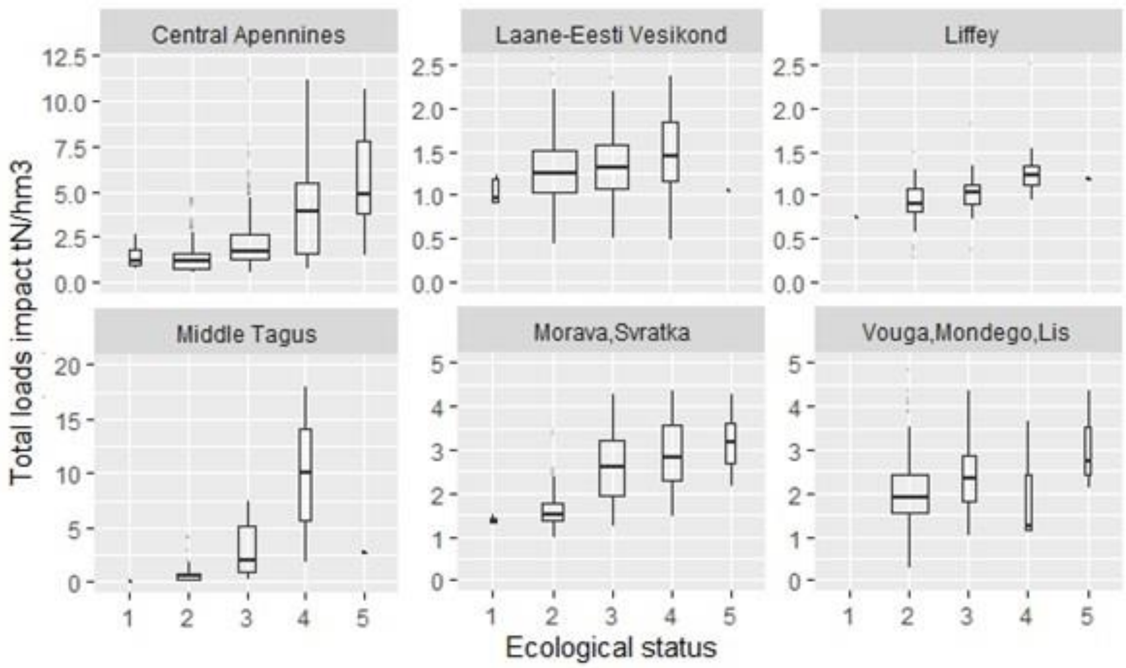


Figure 7. Statistical distribution of the impact of nitrogen pressures for each ecological status class, by RBD. Ecological status 1=High, 2=Good, 3=Moderate, 4=Poor, 5=Bad. Box width is proportional to the number of water bodies in each category. Note the different vertical scales.

4.2. Impact level vs. declared exemptions to nitrogen environmental objectives

When comparing the magnitude of nitrogen load with the declaration of disproportionate cost or technical feasibility exemptions to nitrogen environmental objectives, we find a tendency to have a large number of exemptions in water bodies with higher nitrogen impact (Figure 8).

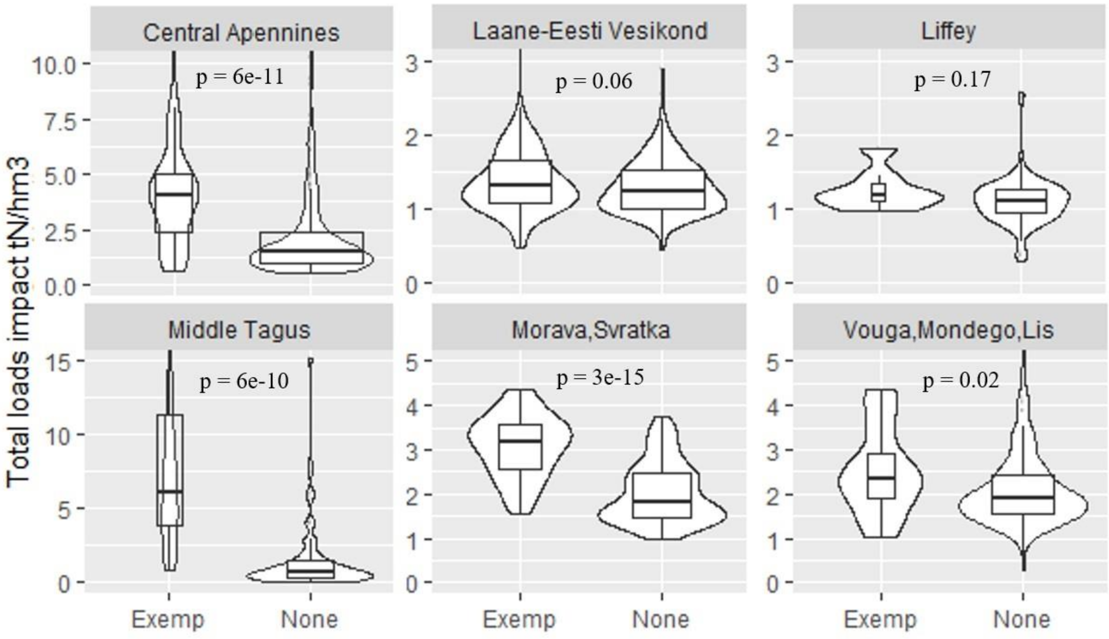


Figure 8. Impact of nitrogen pressures and application of exemptions to the nitrogen indicator. The width of each plot represents the quantity of water bodies for each impact level, while the box and whiskers show the position of the quartiles of the distributions.

The Kruskal-Wallis test applied in R (R Core Team 2019) to each RBD revealed that the group of water bodies with exemptions to nitrogen environmental objectives are significantly different from the group without exemptions in the Vouga, Mondego and Lis, Middle Tagus, Central Apennines and Svratka-Morava river basins, with a high level of confidence ($p < 0.05$). No difference was detected in the Laane-Eesti Vesikond and Liffey river basins. One explanation may be that the PNPI load factor described by (Munafò et al. 2005) for each Corine land use type represents better the nitrogen diffuse load distribution in lower latitudes than in higher ones. Another reason could be that input data for Laane-Eesti Vesikond was extrapolated, which may affect its quality.

Figure 8 also identifies the threshold above which each RBD Authority has applied the exemptions to the nitrogen concentration objectives. The model shows that in the RBDs with significant differences, the RBD Authorities have resorted preferably to exemptions to nitrogen environmental objectives above an impact of 2.4 +/- 0.2 t/hm^3 (calculated using the categorical method proposed in EC 2018a). In the case of Laane-Eesti Vesikond and Liffey catchments, the average impact level in exempted water bodies is significantly lower than in the other catchments.

The effect of point and diffuse pollution on each water body can also be singled out. Figure 9 plots the impact of diffuse loads (x axis) against point loads (y axis) on each water body in the study area. With the exception of the Central Apennines and the Middle Tagus, the predominant pressure for most water bodies is the diffuse source pollution, as already noted in sub-section 4.1 above.

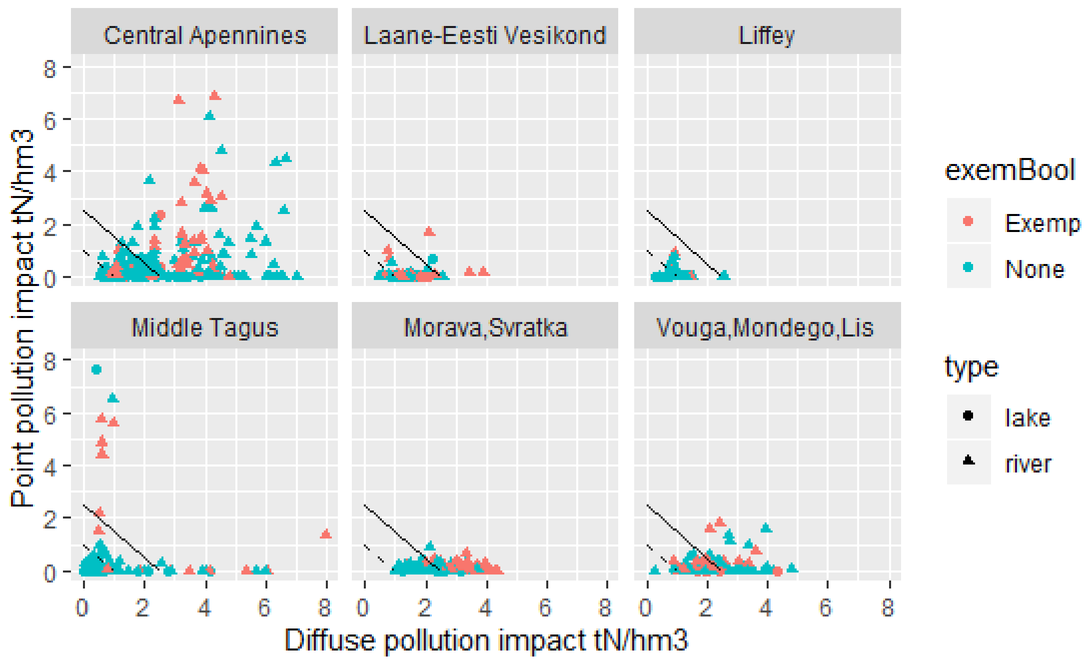


Figure 9. Impact of nitrogen pressures and application of exemptions to the nitrogen indicator.

The inclined lines represent the locus of water bodies with impacts below the total 1 mg/l (dashed) and 2.5 mg/l (solid) found by Poikane et al. (2019) as the average boundary between good and moderate status for lakes and rivers, respectively.

Figure 9 helps identifying the main pressure for each water body, hinting at the type of mitigation measures that may help attain the environmental objectives. For example, the graph suggests that the water bodies in the Middle Tagus with high point pollution impact may meet the environmental objectives with an upgrade of WWTPs. Instead, in the Central Apennines most of the water bodies require action to mitigate not only point pollution but also diffuse pollution, in order to meet the objectives and minimize the need of exemptions to nitrogen environmental objectives. In the remaining RBDs, Figure 9 shows that most mitigation measures should concentrate on diffuse source pollution. It is worth noting that, with respect to water body categories, the threshold for exempted water bodies of lake type tend to be stricter than for water bodies of river type, which is consistent with the stricter limits set for lentic systems (Poikane et al. 2019).

4.3. Exemption types

Our model can also characterize the water bodies according to the type of exemption applied. According to the WFD, if the achievement of the environmental objectives before the end of a given 6-year planning cycle is disproportionately expensive or technically unfeasible, time extensions can be granted. If the environmental objectives are considered to be unattainable in a specific water body, less stringent ones can be set. Arguably, in water bodies with relatively low levels of impact, exemptions should be declared preferably for cost disproportionality rather than for technical unfeasibility. Similarly, they should materialize as time extensions rather than as less stringent objectives.

Figure 10 shows that each RBD Authority has declared exemptions following a different approach. Only the Central Apennines have used all the options available. In this RBD, the lowest level of impact in exempted water bodies is associated to time extensions due to cost disproportionality, while higher levels lead to the use of the technical feasibility clause and to the establishment of less stringent objectives due to disproportionate cost.

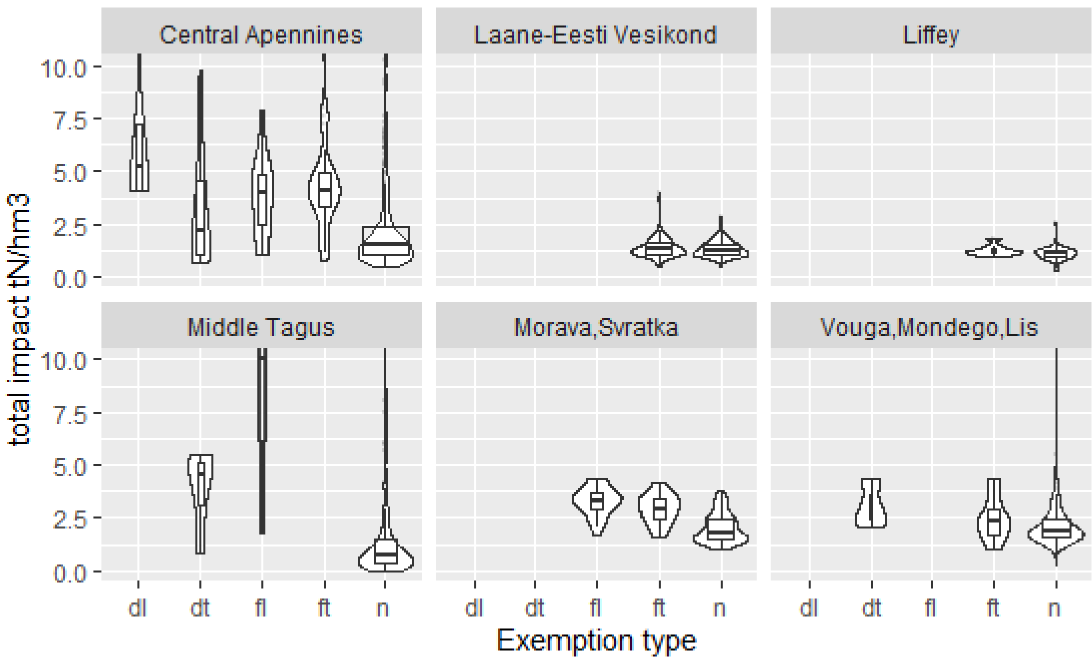


Figure 10. Total loads and exemption type. dl = disproportionate costs, less stringent objectives. dt = disproportionate costs, temporary extension. fl = technical feasibility, less stringent objectives. ft = technical feasibility, temporary extension. n = none.

Lääne-Eesti Vesikond and Liffey water basins only apply the technical feasibility clause for time extensions. In the case of Vouga, Mondego and Lis, only temporal extensions are applied, with disproportionate costs claimed for water bodies with a relatively higher level of impacts. This may be linked to the measures needed for each water body, and their cost. For example, the same point pollution level may be caused by several small WWTPs (with high mitigation cost), or by one large WWTP (which is cheaper to upgrade due to scale economy considerations), an effect not captured in this model.

In the case of the Svratka and Morava basins, only technical feasibility exemptions are implemented, with less stringent objective clauses applied to a higher level of impact, as expected. In the Middle Tagus, less stringent objectives clause is also associated with a higher level of impacts.

Overall, no clear trend can be identified across RBDs (Figure 11), leaving the impression that some arbitrariness has been used in the declaration of each claimed exemption. The same analysis is performed for diffuse pollution, where mitigation measures are technically more complex than in the case of point pollution.

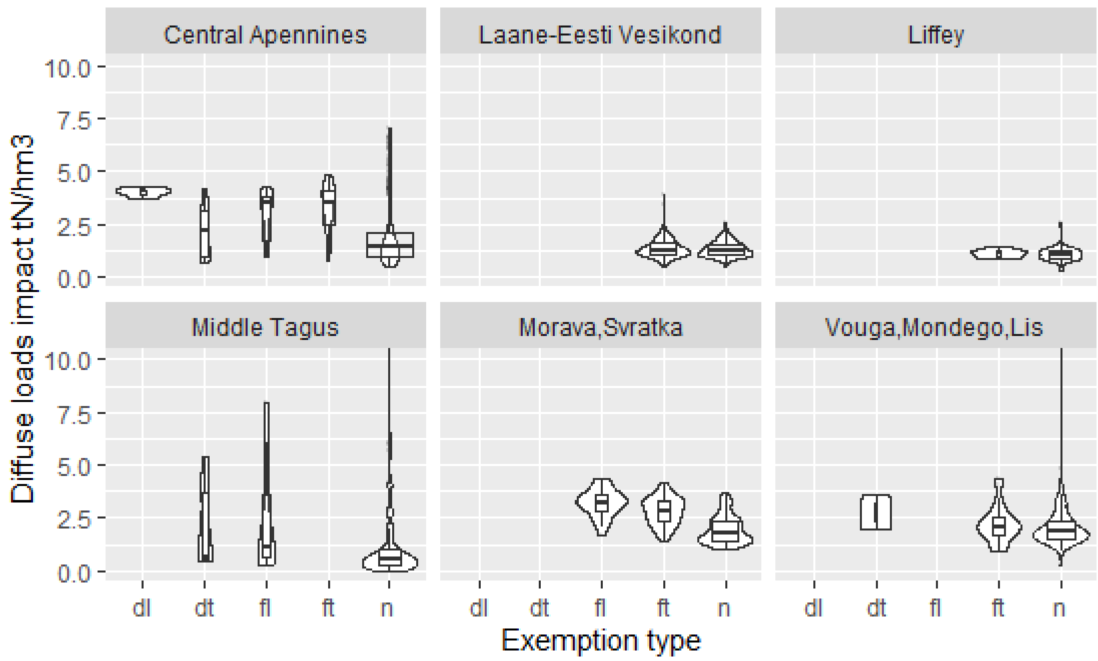


Figure 11. Aggregate pressure from diffuse and point sources for the watersheds with distinct groups, per continental water body type.

4.4. Expected effect of mitigation measures

The model not only describes the current situation but also enables the prediction of the status of the water bodies after the application of specific mitigation measures. As an example, in the Middle Tagus basin, where point source pollution from urban WWTPs driving the declaration of exemptions, the model can predict the expected evolution of the impacts if these pressures were reduced through the implementation of a nitrogen reduction process in some of its WWTPs (see table 1).

In Figure 12, the left bar shows the current situation, while the right bar represents the expected impact level in water bodies after the upgrade of all the WWTPs. The slope of the connecting lines represents the expected amelioration in each water body. This shows the drastic impact reduction, which would render many of the exemptions to nitrogen environmental objectives declared in the current River Basin Management Plan unnecessary. The economic implications of mitigation measures can be assessed through a Cost and Benefit Analysis (EC 2003c). For a given cost of measures, Figure 12 helps identifying the expected benefit in terms of pollution reduction and minimization of the need of resorting to exemptions.

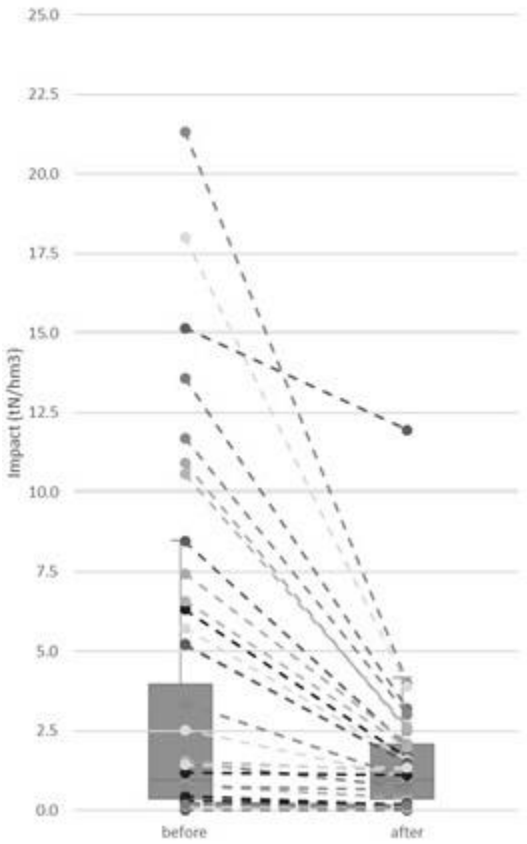


Figure 12. Expected change in the impact of total loads in Middle Tagus water bodies, if all WWTPs apply a nitrogen reduction process. The points represent the impact level for each water body before and after the change.

A similar approach applied to diffuse pollution may identify in which areas a pressure reduction would bring about a large amelioration in the status of the receiving waters, taking the costs into account (Vinten et al. 2012). The implementation of this reduction, however, should be based on voluntary practices by farmers or implementation of riverside buffer zones, since there is no legal instrument in the EU to limit the nitrogen emissions of agricultural land. A better characterization of diffuse pollution through intensive monitoring (Bouraoui and Grizzetti 2014), and a development of the legislation effectively defining an upper limit to the nitrogen emissions – or, indirectly, taxing nitrogen fertilizers to account for cost externalities (Pearce and Koundouri 2013) would be required to reduce the pressure. However, the evidence shows that the control of organic and inorganic nitrogen contamination using market instruments, implemented in some MS like The Netherlands and Denmark, has not been effective in reducing this diffuse contamination.

4.5. Prioritization of interventions

A scrutiny of the distribution of water bodies according to the impact level and the declaration of exemptions to nitrogen environmental objectives shows that there are many bodies with declared exemptions and a relatively low level of impact. The exemptions in those cases are more difficult to justify. In some cases, a limited investment in mitigation measures may help achieve the environmental objectives, thus revoking the need for exemptions to nitrogen environmental objectives. From the point of view of an authority in charge of monitoring, these are the exemptions that may require special attention in the inspections since the presence of exempted water bodies with low levels of declared impacts may help identify pressures that had not been accounted for. Figure 13 shows this type of analysis in the Middle Tagus. A Cost and Benefit Analysis (EC, 2003c) can be conformed combining this assessment of the expected improvement in water quality with an estimation of the cost of each measure.

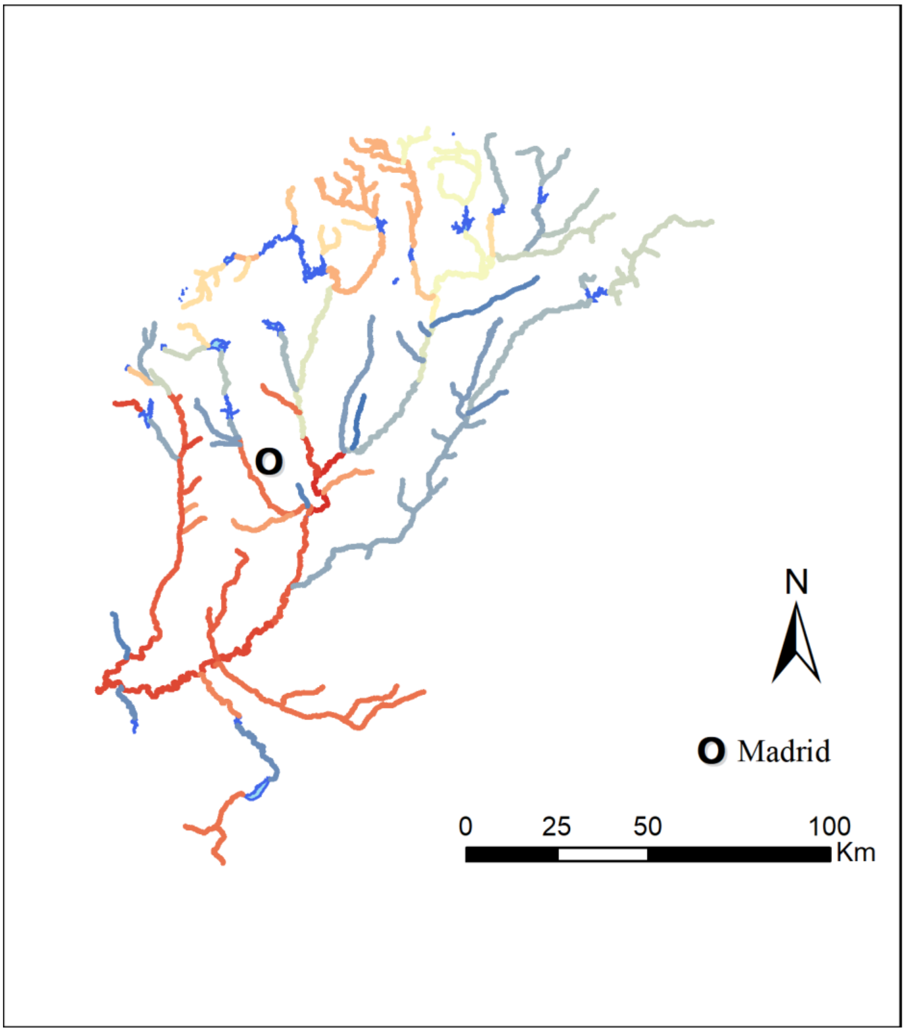


Figure 13. Classification of the Middle Tagus water bodies according to their declared exemption and the modelled nitrogen load. Orange colour represents water bodies with exemptions (darker if relatively low impact), and blue colour represents water bodies without exemptions (darker if relatively high impact). Yellow colour represents water bodies without exemptions and relatively low impact.

Conversely, there are water bodies without any declared exemption to nitrogen environmental objectives but with a high level of impact and thus with a high probability of failing to meet the WFD environmental objectives. The assessment of impact should prioritize these water bodies. Additionally, and in case these water bodies had no data measured on the ground, they would be preferential sites to check the achievement of the objectives.

The prioritization of measures should also take account of the different protection mechanisms that affect water bodies in the European Union, i.e. Natura 2000 sites, areas designated for abstraction of water for human consumption, recreational waters and nutrient-sensitive areas. Figure 14 presents the application of exemptions on water bodies according to their protection status. It shows that except for Central Apennines RBD, exemptions are more widespread in protected water bodies.

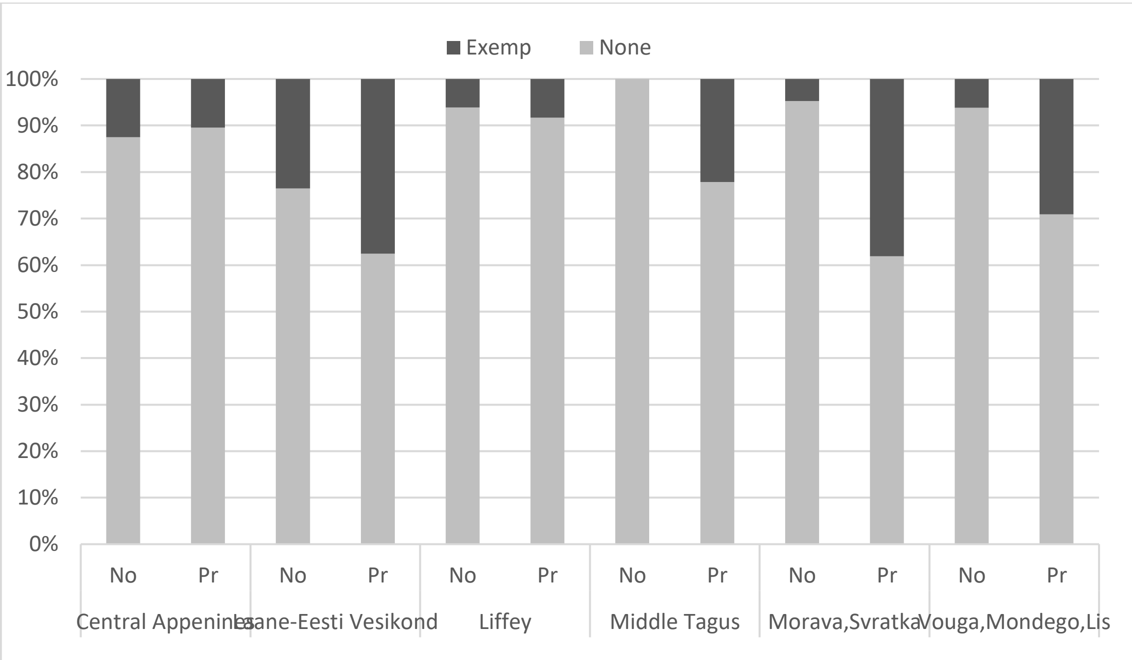


Figure 14. Exemptions and protected water bodies. No: water bodies without any protection status. Pr: water bodies with at least one protection status.

4.6. Action screening matrix

Since there are thousands of water bodies exempted from their nitrogen environmental objectives, a one by one study by the inspection authorities would require intensive resources. In this context, a screening tool that exploits the secondary data already available in the existing databases (therefore with a low implementation cost) and indicates the best course of action can be useful. The previous considerations can be summarized in the action screening matrix presented in Table 2, where each water body is classified according to the level of impact of the nitrogen pressures, and the declaration or not of exemptions to the nitrogen environmental objectives.

Table 2. Proposed action for water bodies according to the impact level of nitrogen pressures and presence of exemptions to the nitrogen environmental objectives.

WB: water body. ES: ecological status of a given water body as declared in the RBMP.

	No exemption declared	Exemption declared
N impact level is high	a) ES is good or better based on direct observation in the WB → *no further action required* b) ES is good or higher, not based on direct observation → *check actual quality, as real status is probably worse than good* c) ES is worse than good: most probably the main cause for the status is high N level → *define relative weight of point and diffuse load and design mitigation measures*	→ *Define relative weight of point and diffuse loads* →*Design mitigation measures and calculate the cost* → *Check feasibility and cost proportionality*
N impact level is low	a) ES is good or better → *no further action required* b) ES is worse than good → *investigate other pressures*	→ *Investigate the reason behind exemption declaration, such as undeclared point pressures; diffuse pressures not accounted for in the model; and other pressures*

4.7. Implications on the number of exempted water bodies

The fact that water authorities seem to set the threshold of exemption in the vicinity of 2.4 tN/hm^3 (see section 4.2 above) has strong implications with regards to the percentage of water bodies that are exempted from the environmental objectives. The cumulative plot of water body percentage below a given impact level (Figure 15) shows that 15% of water bodies in our study area have an impact above 2.5 tN/hm^3. If the percentage of exempted bodies is to be reduced, the average impact level associated to the declaration of exemption should be significantly higher. For instance, for a target 5% of exempted bodies, the average impact level should be above 4.5 tN/hm^3. It must be noted, though, that this consideration is valid only in aggregate values and context-specific factors may apply to each water body.

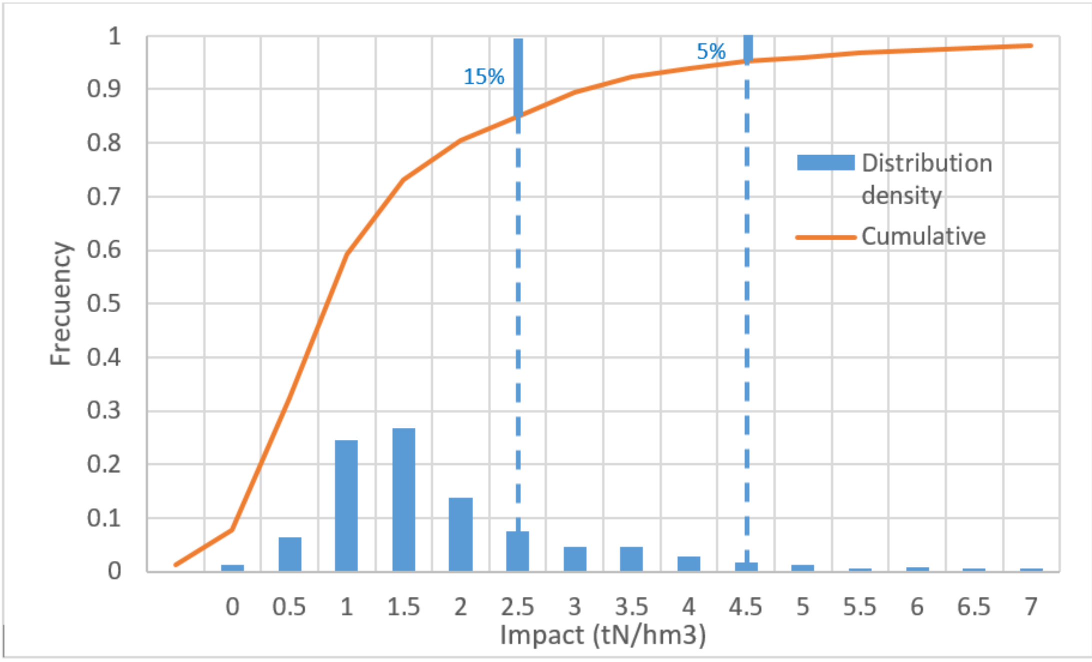


Figure 15. Distribution density and cumulative plots of water bodies according to their impact factor.

4.8. Caveats and further developments

In light of the results, several questions remain unanswered and need further research. The transmission of nitrogen from land surface to water bodies is very complex and requires further research to be fully understood. In this paper, we have used a mixed approach combining statistical regression models with reliable aggregate values, and GIS based models with a higher geographical accuracy. Although the result identifies the trends in four of the six RBDs, the lower accuracy in the Liffey and Laane-Eesti Vesikond river basins show that a refinement of the diffuse load assessment is still needed.

In this paper, we only model nitrogen evolution. In future research, other pollutants may be included to complete the description of indicators that contribute to the physico-chemical indicators of water body status. In particular, phosphorus should be studied, although the quantification of diffuse load pressures entails even more uncertainties than the case of the nitrogen.

Being a discretized, stationary model, effects like the evolution within a water body or transitionary phenomena are beyond the scope of our work. Furthermore, only continental surface water bodies are covered in this paper. Although the effect of pollution on groundwater bodies and transitional and coastal water bodies has been thoroughly studied in the scientific literature, to our knowledge no study has been conducted so far to assess the criteria used to define exemptions in those types of water bodies. Such research line would offer a more complete picture of the environmental health of water bodies and help prioritize policy interventions.

Future research could focus the modelling of other elements that complete the description of the ecological status of the water bodies, in particular the biological ones. To date the response of biological elements to the implementation of measures are not well understood (Palmer, Menninger, and Bernhardt 2010; Hering et al. 2010; Jähnig et al. 2010).

The declaration of exemptions is in many cases directly linked to the costs and benefits of the mitigation measures that would alleviate the pressures on the water bodies. However, there is little information about the actual costs of mitigation measures such as the implementation of a nitrogen reduction cycle in a WWTP, or the cost of diffuse pollution reduction measures. Furthermore, existing methods face difficulties in the monetization of the environmental benefits of the measures.

5. Conclusions

The application of exemptions to the environmental objectives of the Water Framework Directive was conceived as an option to be applied in exceptional situations where meeting those objectives would be extremely difficult or costly. Two decades after the approval of the WFD there are still no common methods to delimit in which cases the exemptions are justified. Furthermore, the EC is still granting exemptions extensively. The apparent arbitrariness of the application and the limited transparency of the justification can compromise the achievement of the goals of the Directive, and prevents a consistent application of its requirements across the Member States of the European Union.

In this paper we have developed an approach to estimate nitrogen pressures on each water body, assess their impacts and confront them with the application of exemptions. The methodology can be applied to all the water bodies in the geographical scope of the WFD, and helps assess the need for exemptions even in water bodies with scarce monitoring.

The results show that in the catchments of the study area with statistically significant differences, the average threshold above which exemptions are prevalent in rivers and lakes is 2.4 tN/hm^3. The threshold in river-type water bodies is close to an annual average of 2.5 tN/hm^3 or an instantaneous value of 2.5 mg/l, which is similar to the average limit between good and moderate status found in the literature. In the case of lakes, the limit is more stringent, as found by previous studies. These results hint at the possibility that, when confronted to water bodies with nitrogen concentrations above the limits, RBD Authorities have often resorted to the declaration of exemptions rather than the definition of mitigation measures. These thresholds imply an approximate proportion of 15 % of water bodies with exemptions. In order to reduce this proportion to 5%, the impact level threshold to declare exemptions should be raised to 4.5 t/hm^3. No clear trends have been identified for the different exemption types (cost disproportionality vs. technical feasibility, time extension vs. less stringent objectives).

In the case of point source pollution, the model can predict the evolution path that the impact on water bodies may have if an amelioration of processes in WWTP was implemented. This effect may also be calculated for the mitigation of diffuse pollution, although the implementation of measures would be more complex.

The model has been able to single out the water bodies where the application of exemptions is not consistent with the level of impacts of the existing pressures. On the one hand, it identifies the water bodies where exemptions are applied but the impact is relatively small. These would be potential candidates for rejecting the exemptions. On the other hand, the model delimitates the water bodies with a high level of impacts but with no exemption declared. If no monitoring sites are operative in those water bodies, they could represent cases where failure to meet the environmental objectives goes unnoticed.

The current tools for diffuse nitrogen load quantification are still approximate and in need of better accuracy and prediction capacity. A similar methodology should be developed for phosphorus quantification to be applied in conjunction with the nitrogen assessment. Furthermore, a better understanding of the evolution of biological indicators to external pressures would help define the Programme of Measures to achieve the good ecological status required by the WFD.

Statement

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9. Transferencia a la sociedad

En el desarrollo de esta Tesis se ha otorgado una especial importancia a la transferencia activa a la sociedad de los resultados de la investigación, no solo mediante la producción de artículos científicos, sino también mediante la organización de seminarios, reuniones y la difusión de artículos divulgativos. Durante la Tesis se ha mantenido un diálogo fluido con las autoridades competentes, empresas de gestión de infraestructuras de depuración y grupos ecologistas.

A continuación, se presentan las actividades de transferencia a la sociedad desarrolladas durante el transcurso de la Tesis.

9.1. Seminarios y jornadas con los actores implicados

Se han organizado cuatro seminarios y jornadas para compartir el avance del estudio con las administraciones y los actores principales. En dichas jornadas se ha promovido el diálogo interadministrativo en aquellos aspectos donde se han identificado carencias, y la puesta en común de ideas y conocimientos. Además, se han organizado numerosas reuniones con técnicos e investigadores la Dirección General del Agua del Ministerio para la Transición Ecológica, la Confederación Hidrográfica del Tajo, el Ayuntamiento y la Comunidad Autónoma de Madrid, el Centro de Estudios Hidrográficos del Centro de Estudios y Experimentación de Obras Públicas (CEH-CEDEX), el Canal de Isabel II, la Universidad de Castilla la Mancha, el World Wildlife Fund (WWF), la Fundación Nueva Cultura del Agua y diversas empresas del sector. En estas reuniones se ha trabajado para entender la perspectiva de cada actor implicado, compartir los resultados y proponer soluciones a los diversos problemas del área de estudio. A continuación se describen los seminarios y jornadas organizados:

I. **Seminario sobre depuración de aguas residuales urbanas: retos y oportunidades**, 13 de septiembre de 2018. En este primer seminario, de asistencia bajo invitación, se presentaron los objetivos del estudio y algunos resultados parciales sobre la calidad del río Manzanares. Además, se aprovechó la presencia de ponentes especializados para entender los condicionantes legales y competenciales, los aspectos financieros y algunos casos aplicados de depuración de aguas residuales urbanas. Se pueden consultar los detalles de la jornada y las ponencias en https://www.fundacionbotin.org/observatorio-contenidos/seminario-sobre-depuracion-de-aguas-residuales-urbanas-13-de-septiembre-de-2018.html. Las ponencias trataron sobre:

   - Visión de conjunto: Víctor Arqued, Subdirector General de Planificación y Uso Sostenible del Agua, Ministerio para la Transición Ecológica.
   - Marco legal y competencial: Beatriz Setuáin, Profesora titular en la Universidad de Zaragoza e investigadora en el grupo AGUDEMA: Agua, Derecho y Medio Ambiente.
   - Financiación:

- Alejandro Gil, Director-Gerente de Infraestructuras del Agua, Consejería de Fomento de Castilla La Mancha.
- Jaime Morell, Secretario General de la Asociación Española de Operadores Públicos de Abastecimiento y Saneamiento.
- Philippe Rougé, Product Manager de Desalación, Potabilización y Depuración de Aqualogy, SUEZ España.

- Monitoreo y modelización:
  - Javier Paredes, profesor titular en la Universitat Politècnica de València e investigador en el Instituto de Ingeniería del Agua y Medio Ambiente.
  - Antonio Bolinches, investigador en el Observatorio del Agua y la Universidad Complutense de Madrid.
- Casos de estudio:
  - Depuradora de Fabara: Francisco Doménech. Alcalde de Fabara.
  - Casos de depuración en Badajoz: Álvaro Jiménez, Gerente de Promedio, Diputación de Badajoz.
  - Depuradoras de Interés General del Estado en Madrid: Ángel Cajigas, Subdirector General Adjunto de Infraestructuras y Tecnología, Ministerio para la Transición Ecológica.
  - Casos de depuración en el Segura: Miguel Ángel Ródenas, Confederación Hidrográfica del Segura.

II. **Jornada de puertas abiertas sobre depuración de aguas residuales**, 7 de mayo de 2019. En esta jornada se añadieron ponencias del sector industrial para entender las posibilidades tecnológicas actuales en materia de depuración, y se abrieron las puertas a la asistencia de público para promover la transferencia de conocimiento. Los detalles están disponibles en https://www.fundacionbotin.org/observatorio-contenidos/jornada-de-debate-sobre-depuracion-de-aguas-residuales-mayo-de-2019.html. Las ponencias trataron sobre:

- Presentación del Plan Nacional de Depuración, Saneamiento, Eficiencia, Ahorro y Reutilización: Víctor Arqued, Subdirector General de Planificación y Uso Sostenible del Agua, Ministerio para la Transición Ecológica.
- Efluentes de depuración y afección al medio receptor:
  - Antonio Bolinches, Investigador en el Observatorio del Agua, Universidad Complutense de Madrid.
  - Diego Moxó: Director del Área de Gestión del Medio, Agencia Catalana del Agua.
- Tecnologías de depuración:
  - David Ambrona, Treatment Manager Iberia, Xylem.
  - Joan Marc Ponsoda Mauri, Ingeniero de Procesos, Acciona Agua.
  - Lucía Soriano, Consultora, Profesora asociada en la Universidad de Zaragoza.

- Financiación:
  - Ángel Cajigas, Subdirector General Adjunto de Infraestructura y Tecnología, Ministerio para la Transición Ecológica.
  - Alejandro Gil, Director Gerente de Infraestructuras del Agua, Consejería de Fomento de Castilla La Mancha.
  - Álvaro Jiménez, Gerente de Promedio, Diputación de Badajoz.

III. **Jornada de debate sobre los retos económicos para alcanzar el buen estado de las aguas superficiales**, 29 de octubre de 2019. Con el foco puesto en los aspectos económicos de la depuración, se convocó otra jornada abierta al público en la que expertos con experiencia a nivel europeo expusieron los aspectos a tener en cuenta para analizar los costes y beneficios de los proyectos de depuración. Asimismo, se presentaron los resultados de la investigación sobre métodos de estudio de costes desproporcionados (capítulo 7). Los detalles y las presentaciones están disponibles en https://www.fundacionbotin.org/observatorio-contenidos/jornada-de-debate-sobre-los-retos-economicos-para-alcanzar-el-buen-estado-de-las-aguas.html. Los temas tratados fueron:

- Introducción a los retos económicos a nivel español y europeo:
  - Víctor Arqued, Subdirector General de Planificación y Uso Sostenible del Agua, Ministerio para la Transición Ecológica.
  - Manuel Lago, Consultor de Economía de Recursos Medioambientales, Ecologic Institute.
- El análisis económico en la Directiva Marco del Agua:
  - Carlos Benítez, Coordinador Técnico, Emgrisa.
  - Josefina Maestu, Asesora de Agua del Secretario de Estado de Medio Ambiente del Ministerio para la Transición Ecológica.
  - Amelia Pérez Zabaleta, Directora de la Cátedra Economía del Agua, Fundación Aquae-Universidad Nacional de Educación a Distancia – UNED.
- Las exenciones por costes desproporcionados:
  - Antonio Bolinches, Investigador del Observatorio del Agua, Universidad Complutense de Madrid.
  - Francesc Hernández Sancho, Profesor Catedrático, Universidad de Valencia.

IV. **Reunión de trabajo sobre depuración y calidad de aguas en los ríos Manzanares y Jarama**, 9 de marzo de 2020. Tras identificar cierta dificultad para acelerar la puesta en común de datos y resultados entre actores, se propuso la celebración de una jornada a puerta cerrada entre los actores implicados. Esta jornada se convocó bajo un acuerdo de confidencialidad de los aspectos tratados, y se contó con asistentes de la Confederación Hidrográfica del Tajo, Ministerio para la Transición Ecológica, Comunidad Autónoma de Madrid, Ayuntamiento de

Madrid, Canal Isabel II, Centro de Estudios Hidrográficos del Centro de Estudios y Experimentación de Obras Públicas (CEH-CEDEX), Aguas de las Cuencas de España (ACUAES), Universidad Politécnica de Valencia, Universidad Politécnica de Madrid y Universidad Complutense de Madrid.

9.2. Participación en congresos científicos

Asimismo, los resultados parciales generados se han ido compartiendo con otros investigadores del sector en diferentes congresos científicos.

- Primer Encuentro Informal en I+D+i en Economía y Políticas en la Gestión del Agua (Universidad Politécnica de Valencia), 26 de julio de 2018. Comunicación "Interacciones entre cantidad y calidad en la consecución del buen estado de las aguas continentales del Tajo Medio según la Directiva Marco del Agua". https://www.iiama.upv.es/iiama/en/newsroom/news/jueves-26-de-julio-primer-encuentro-informal-en-i-d-i-en-economia-y-politicas-en-la-gestion-del-agua.html
- Congreso Ibérico de Gestión y Planificación del Agua (Coimbra, Portugal), 7 de septiembre de 2018. Comunicación "Grandes volúmenes de vertidos urbanos y ríos poco caudalosos: El caso de Madrid y el Manzanares". https://fnca.eu/congresos-ibericos/x-congresso-iberico
- European Geosciences Union (EGU) General Assembly, Water quality at the catchment scale: measuring and modelling of nutrients, sediment and eutrophication impacts, 10 de abril de 2018. Poster "Large Wastewater Volumes in Small Rivers: the Case of Madrid, Spain". https://meetingorganizer.copernicus.org/EGU2018/session/26594

- Segundo Encuentro Informal en I+D+i en Economía y Políticas en la Gestión del Agua (Universidad de Alcalá), 24 de junio de 2019. Comunicación "Los costes desproporcionados en la Directiva Marco del Agua: El caso del Tajo Medio".
- 11th World Congress on Water Resources and Environment (EWRA 2019), 27 de junio de 2019. Comunicación "Modeling surface water quality with limited data: A calibration approach applied to the Middle Tagus Basin (Spain)". http://ewra.net/pages/EWRA2019_Proceedings.pdf

- European Geosciences Union (EGU) General Assembly 2020. Session on Water resources policy and management - systems solutions in an uncertain world, 6 de mayo de 2020. Comunicación "Setting the threshold: An analysis of different approaches for the definition of exemptions to water quality objectives in the European Union". https://meetingorganizer.copernicus.org/EGU2020/session/35481

9.3. Conferencias y charlas divulgativas

El doctorando ha impartido diversas charlas divulgativas, cuyo registro se ha adaptado a la audiencia, desde conferencias para un público especializado en un entorno universitario, hasta charlas para un público general para contribuir a una concienciación de los efectos de los vertidos:

- Semana de la Ciencia. Caminar el agua: travesía a pie por las orillas del río Manzanares, entre lo rural y lo urbano, 18 de noviembre de 2017. Charla "La depuración de las aguas residuales: Viveros de la Villa".
https://www.ucm.es/data/cont/docs/1334-2017-11-06-SC2017_Geol%C3%B3gicas_20171106.pdf
- Grupo de investigación "El río Tajo: hacia un enfoque holístico de sus problemas y soluciones" (Universidad de Castilla-La Mancha) y Real Fundación de Toledo. Ciclo de conferencias Investigando el Tajo, 30 de octubre de 2018. Conferencia "Grandes volúmenes de vertidos urbanos y ríos poco caudalosos: el caso de Madrid y el Manzanares".
http://www.toledo.es/agenda/ciclo-de-conferencias-investigando-el-tajo-5/
- Escuela T.S. de Ingeniería de Montes, Forestal y del Medio Natural de la Universidad Politécnica de Madrid. Ciclo sobre las aguas continentales en España, 23 de noviembre de 2018. Conferencia "Modelización de la calidad físico-química del agua. Aplicaciones en España".
https://www.youtube.com/watch?v=8_dhv545DdM

- Semana de la Ciencia. Travesía a pie desde Somontes a Madrid río siguiendo las aguas del río Manzanares, 6 de noviembre de 2018. Charla "La depuración de las aguas residuales urbanas".
https://geologicas.ucm.es/data/cont/docs/19-2018-10-26-Cartel%20SemanaCiencia%20Martes6_CAMINAR%20EL%20AGUA_02.pdf

- Universidad de Castilla la Mancha (Toledo). Curso de Gestión y planificación del agua en la cuenca del Tajo, 10 de abril de 2019. Clase "Las aguas residuales de la Comunidad Autónoma de Madrid y la presión sobre el Tajo".
https://blog.uclm.es/grupotajo/curso-en-gestion-y-planificacion-del-agua-en-la-cuenca-del-tajo/

- Ministerio para la Transición Ecológica. Workshop OECD: Strategies to close the financing gap for water supply, sanitation and flood protection in Spain, 25 de abril de 2019. Intervención "Harnessing the effect of urban sewage on Europe's surface waters: A tale of two directives".
- Comunidad Autónoma de Madrid, Consejería de Medio Ambiente, Ordenación del Territorio y Sostenibilidad. Ciclo sobre la problemática generada por el desarrollo urbano en la red hidrográfica de la Comunidad de Madrid, 8 de octubre de 2019. Conferencia "Efluentes de depuración y afección al medio receptor: El caso del Tajo Medio".
https://www.fmmformacion.es/cursosver.php?cur_id=4147

- Coloquios online organizados por el Programa Observatorio del Agua de la Fundación Botín, 30 de abril de 2020. Webinar "El efecto de las depuradoras urbanas y el trasvase Tajo-Segura sobre la calidad de las aguas del Tajo Medio. Implicaciones para el Tercer Ciclo de Planificación". https://www.fundacionbotin.org/noticia/coloquios-online-del-observatorio-del-agua.html

- Presentación iAgua Magazine 28, 28 de mayo de 2020. Entrevista sobre " La depuración en Madrid y la calidad de las aguas del Tajo Medio". https://youtu.be/Hq9r14h1BhE
- Mesa redonda del Observatorio El Economista "El agua después de la pandemia", 26 de junio de 2020. https://www.eleconomista.es/empresas-finanzas/noticias/10648963/07/20/El-agua-puede-crear-43000-empleos-nuevos-durante-los-proximos-10-anos.html

9.4. Artículos de divulgación

Se han publicado los siguientes artículos de divulgación con el objetivo de transmitir los resultados de la investigación a una audiencia lo más amplia posible:

- "Las aguas residuales de Madrid y el río Manzanares", 10 de febrero de 2019. https://www.iagua.es/noticias/fundacion-botin/aguas-residuales-madrid-y-rio-manzanares

- "Cómo las depuradoras de Madrid y el Trasvase Tajo-Segura afectan a la calidad del agua del Tajo", 30 de abril de 2019. https://www.iagua.es/noticias/fundacion-botin/como-depuradoras-madrid-y-trasvase-tajo-segura-afectan-calidad-agua-tajo

- "Los retos económicos de la implementación de la Directiva Marco del Agua", 10 de octubre de 2019. https://www.iagua.es/noticias/fundacion-botin/retos-economicos-implementacion-directiva-marco-agua

- "Escenarios de mejora de la calidad del agua del Tajo Medio", 14 de febrero de 2020. https://www.iagua.es/noticias/fundacion-botin/escenarios-mejora-calidad-agua-tajo-medio

- "La depuración en Madrid y la calidad de las aguas del Tajo Medio", 24 de junio de 2020.

  https://www.iagua.es/blogs/antonio-bolinches/depuracion-madrid-y-calidad-aguas-tajo-medio

10. Discusión y conclusiones generales

El Tajo Medio es una zona donde las concentraciones de algunos contaminantes están por encima de los límites establecidos por la legislación. En esta Tesis se ha realizado un análisis sistemático de los problemas que afectan a la calidad de sus aguas, se han propuesto límites a la concentración de contaminantes de los efluentes de las depuradoras (la presión más significativa del sistema) y se ha desarrollado una metodología para identificar las medidas con costes desproporcionados.

En relación con la consecución del primer objetivo parcial de esta Tesis, se ha cuantificado el cambio requerido en el nivel de presiones para alcanzar el buen estado de las aguas superficiales. La legislación vigente describe el objetivo a cumplir (el buen estado de las aguas receptoras, medido con el valor de los indicadores de calidad descritos) y los medios para conseguirlo en el caso de contaminación puntual. Los modelos de evolución de contaminantes generados en la Tesis han permitido relacionar las presiones físico-químicas sobre las aguas del Tajo Medio (efluentes de depuración, contaminación difusa, abstracciones, etc.) con el estado de las aguas receptoras. Los resultados nos muestran las concentraciones de contaminantes en los efluentes de las depuradoras del área de estudio (ríos Henares, Manzanares y Jarama) que serían compatibles con el buen estado de las aguas receptoras. Se demuestra que el efecto de las presiones difusas es despreciable respecto al de los efluentes de depuradora. En el caso de los efectos del Acueducto Tajo-Segura, se cuantifica el impacto y se demuestra que actuando solamente sobre los caudales transvasados no se alcanzaría el buen estado de las aguas superficiales del eje del Tajo aguas abajo de su confluencia con el río Jarama.

Una vez caracterizados los cambios requeridos en las presiones, en cumplimiento del segundo objetivo parcial se han identificado y se ha dado un orden de prioridad a las medidas correctoras necesarias para llegar al buen estado de las masas de agua estudiadas. Asimismo, los modelos nos muestran la concentración de contaminantes esperable en las aguas superficiales a medida que se implementan las acciones correctoras propuestas.

Los límites requeridos en los efluentes de depuración son más restrictivos que los genéricos de la Directiva de tratamiento de las aguas residuales urbanas, por lo que deberían fijarse en las autorizaciones de vertido de las depuradoras. Por otro lado, la Tesis sugiere la necesidad de que la Directiva de tratamiento de las aguas residuales urbanas se adapte para definir los mismos contaminantes que la DMA, y dé una prioridad explícita e ineludible a la consecución de los objetivos medioambientales en las aguas receptoras.

La legislación está menos desarrollada en el caso de la contaminación difusa (solo para nitrógeno de fuentes agrarias se prescribe el seguimiento, ni siquiera los límites máximos de vertido) por lo que no existen instrumentos legales claros para mitigar este tipo de emisiones, que en todo caso tienen un efecto marginal en el área de estudio.

Los cambios necesarios en la infraestructura de depuración de la zona de estudio implican grandes inversiones. En el plan hidrológico vigente se hace uso de exenciones para flexibilizar los objetivos medioambientales. Dichas exenciones deben estar basadas en datos observables y accesibles, de modo que se facilite la comparación con otras demarcaciones europeas. Sin embargo, aún existe una laguna científica y normativa en cuanto a la definición de una metodología que permita establecer este tipo de exenciones de una forma objetiva y común para todos los Estados Miembros de la UE.

A tal efecto, en cumplimiento del tercer objetivo parcial se ha desarrollado una metodología para cuantificar un índice de desproporcionalidad del coste de las medidas estudiadas, de manera que se pueda comparar el esfuerzo ejercido por los distintos Estados Miembros de la Unión Europea.

La comparativa de la aplicación de exenciones entre diferentes demarcaciones europeas (cuarto objetivo parcial de la Tesis) nos muestra que el nivel de impacto de presiones a partir del cual se han aplicado exenciones de manera generalizada en las cuencas europeas de estudio es relativamente bajo. Ello implica que hay un gran porcentaje de masas de agua con exenciones declaradas, lo cual supone una amenaza para los objetivos estratégicos de la Directiva Marco del Agua. Se debería aumentar el umbral de impacto de las presiones a partir del cual se empiecen a plantear exenciones, y dicho umbral debería consensuarse para todas las demarcaciones de la Unión Europea.

Con la combinación de modelos matemáticos de evolución de contaminantes y métodos de análisis de coste y beneficio se ha pretendido conferir al estudio un enfoque integrador, desde el que proponer actuaciones tanto desde el punto de vista técnico como administrativo.

Aparte de las contribuciones de la investigación desarrollada en la Tesis en materia de valores cuantitativos admisibles de las presiones y propuestas metodológicas en el tratamiento de exenciones, en la presente Tesis se ha otorgado un peso especial a la transferencia de conocimiento a la sociedad. Los resultados se han transmitido mediante numerosas reuniones con los actores implicados y la organización de seminarios y jornadas de puertas abiertas, dando cumplimiento al quinto y último objetivo parcial.

10.1. Reflexiones cualitativas

En base a las conclusiones de la Tesis, se sugiere la revisión del procedimiento de definición de las exenciones, que en su forma actual no conduce al buen estado de las aguas superficiales ni garantiza un esfuerzo comparable entre Estados Miembros.

En el caso del área de estudio, la particular distribución demográfica en el interior de España resulta en una gran concentración urbana en la zona metropolitana de Madrid. Dicha posición es idónea para el abastecimiento debido a la alta precipitación en el Sistema Central, que gracias a su mayor elevación respecto a las zonas urbanas permite el transporte por gravedad a los usuarios con un consumo mínimo de energía. Además, la buena calidad de las aguas (poco mineralizadas debido a la litología del

Sistema Central) implica un gasto mínimo de potabilización. Sin embargo, la posición es poco idónea para el drenaje de las aguas residuales generadas. El bajo caudal de los ríos, especialmente durante su estiaje, requiere unos altos niveles de depuración para evitar concentraciones de contaminantes que afecten a su estado ecológico. La infraestructura de depuración debe paliar los efectos de esta ubicación, con un balance racional entre los costes implicados y la calidad resultante. Este efecto, combinado con el efecto favorable de una economía de escala que se traduce en un menor coste por habitante de la depuración en grandes aglomeraciones (para calidad de efluentes equivalentes) genera un problema de optimización que debe tratarse para garantizar el cumplimiento de los objetivos de la Directiva Marco del Agua.

Por otro lado, se constata que la definición de competencias de los diferentes niveles de la administración pública dificulta la gestión, especialmente en casos de cuencas intercomunitarias. Mientras el principio de subsidiariedad (Consejo de Europa 1985; BOE 1999) atribuye a la administración local (la más cercana al ciudadano) la titularidad de los vertidos urbanos, muchos estatutos de autonomía de comunidades autónomas se atribuyen competencias en materia de aguas residuales, y en el caso de obras declaradas de Interés General del Estado (Ministerio de Medio Ambiente 2001), la Administración General del Estado toma el papel principal en el diseño y ejecución de las mismas.

Finalmente, se detecta un decalaje entre las competencias atribuidas por la legislación a las Confederaciones Hidrográficas, y los recursos de que disponen para desarrollarlas. La infrafinanciación de las Confederaciones dificulta la redacción de los estudios necesarios para establecer autorizaciones de vertido ad hoc para cada depuradora, y la verificación que las aguas superficiales responden como esperado a la implementación de nuevas infraestructuras de depuración.

10.2. Líneas de trabajo futuras

El presente estudio identifica algunas líneas de trabajo que habría que desarrollar para ampliar el conocimiento de los fenómenos implicados en la calidad del agua del área de estudio.

En la parte de modelización, las observaciones disponibles solamente permitían establecer modelos con medias en estado estacionario. Aun así, existen fenómenos transitorios de cierta importancia cuyo estudio podría aportar una mayor profundidad a la comprensión del problema.

Las bajas temperaturas en invierno puede afectar a la eficiencia de los procesos de tratamiento en las depuradoras, y por tanto a la calidad del efluente (Antoniou et al. 1990). En particular, el rendimiento del proceso de nitrificación y desnitrificación se reduce sensiblemente cuando la temperatura del influente disminuye a temperaturas bajas. En ese caso, la cantidad de amonio vertido podría aumentar por encima de los valores de diseño de la depuradora, y la concentración en las aguas receptoras se incrementaría sensiblemente.

Otro fenómeno transitorio importante lo constituyen los casos de desbordamiento de la red de saneamiento causados por episodios de lluvias torrenciales (Lastra 2017). La mayor parte de la red de saneamiento urbano del área de estudio está constituida por redes unitarias, donde se mezclan las aguas residuales de los domicilios con el agua de escorrentía superficial generada por lluvia sobre zonas urbanas. Mientras una precipitación leve tras una temporada lluviosa provoca un simple aumento de caudal que puede ser absorbido por las depuradoras, un episodio de lluvias torrenciales tras una época sin precipitaciones puede provocar una serie de fenómenos que implican un aumento de la carga vertida. Primero, el lavado de la suciedad acumulada en los pavimentos urbanos provoca que en las primeras horas de lluvia la concentración de contaminantes en el influente de las depuradoras es mucho mayor que el de diseño. Segundo, el caudal influente es mucho mayor que el de diseño y las depuradoras pasan a trabajar en caudal de pico. Y tercero, como el caudal circulante puede desbordar la capacidad de pico de las depuradoras, los operadores pueden verse obligados a verter el sobrante directamente a las aguas receptoras, sin tratamiento. Para tratar este fenómeno, el Real Decreto 1290/2012 (Ministerio de Agricultura 2012) incluyó consideraciones sobre la contaminación por episodios de lluvia.

En todo caso, el tratamiento de la legislación del efecto de los episodios temporales está estrechamente ligado a los requerimientos específicos de mediciones. En el caso de los efluentes de depuradoras, la Directiva 91/271/CEE en su Anexo I, Sec. D.3 pide para depuradoras de más de 50 000 habitantes equivalentes (como es el caso de las grandes depuradoras de nuestra área de estudio) la recogida de un mínimo 24 muestras de efluente al año para su análisis. En el caso de las demandas biológica (DBO) y química (DQO) de oxígeno, y de los sólidos en suspensión (TSS) solo se permitirían 3 incumplimientos (valores observados por encima de los máximos permitidos) dentro de las 24 mediciones, es decir un 12.5% de incumplimientos. En el caso de que se efectuaran más mediciones el porcentaje de incumplimientos permitidos bajaría, aceptándose solamente 25 incumplimientos de 365 mediciones (un 6.8%). En el caso de nitrógeno y fósforo (Anexo I, Sec. D.4), simplemente se requiere el cumplimiento de la media anual.

En las aguas receptoras, la DMA en su Anexo V, Sec. 1.3.4 exige una medición cada tres meses de los indicadores físico-químicos. El Real Decreto 817/2015, en su Anexo III, Apartado B.2.2 de clasificación del estado pide que se cumpla la media de 6 años si las mediciones de concentración son homogéneas. En cambio, si la tendencia de las mediciones es creciente, decreciente o variable se requiere contabilizar la media del último año. Por tanto, la naturaleza discreta y escasa de las mediciones hace que el cumplimiento formal de la legislación dependa de las circunstancias particulares en que se toman las muestras.

Un análisis estadístico de la incidencia media a lo largo del año de estos efectos transitorios contribuiría a la consideración de sus efectos en unas autorizaciones de vertido más completas. En ese sentido, la legislación sobre el tratamiento de aguas residuales urbanas necesita un desarrollo en la definición de mecanismos que relacionen las mediciones puntuales (en efluentes y aguas receptoras) con los límites establecidos.

Por otro lado, se necesita un mayor desarrollo de las herramientas de modelización que expliquen la evolución de otros indicadores (en especial los indicadores biológicos) tras la aplicación de las medidas correctoras. En el caso de contaminantes emergentes, debe avanzarse en la comprensión de sus efectos sobre los ecosistemas de las aguas receptoras.

Finalmente, sería conveniente estudiar los posibles efectos del cambio climático sobre la calidad del agua de la zona de estudio. Un descenso de las precipitaciones medias anuales conllevaría una disminución de la escorrentía superficial (CEDEX 2017), que acarrearía una mayor concentración de contaminantes, requiriendo un esfuerzo adicional de depuración (Estrela 2020). Además, un agravamiento de los fenómenos extremos puede dificultar la gestión de la infraestructura de depuración al traducirse en una mayor acumulación de contaminantes en pavimentos urbanos en períodos de sequía y una mayor frecuencia de desbordamiento de las redes de saneamiento en episodios de lluvias torrenciales. Un análisis cuantitativo de estos efectos permitirá entender y controlar sus consecuencias sobre la calidad de las aguas receptoras.

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