European Commission

JRC SCIENCE FOR POLICY REPORT

# Water quality in Europe: effects of the Urban Wastewater Treatment Directive

*A retrospective and scenario analysis of Dir. 91/271/EEC*

Pistocchi, A., Dorati, C., Grizzetti, B., Udias, A., Vigiak, O., Zanni, M.

2019

Joint Research Centre

EUR 30003 EN

**Contact information**
Name: Alberto Pistocchi
Address: via E.Fermi 2749, 21027 Ispra (VA) Italy
Email: alberto.pistocchi@ec.europa.eu
Tel.: +390332783658

**EU Science Hub**
https://ec.europa.eu/jrc

JRC115607

EUR 30003 EN

PDF ISBN 978-92-76-11263-1 ISSN 1831-9424 doi:10.2760/303163

Luxembourg: Publications Office of the European Union, 2019



How to cite this report: Pistocchi, A., Dorati, C., Grizzetti, B., Udias, A., Vigiak, O., Zanni, M., *Water quality in Europe: effects of the Urban Wastewater Treatment Directive. A retrospective and scenario analysis of Dir. 91/271/EEC*, EUR 30003 EN, Publications Office of the European Union, Luxembourg, 2019, ISBN 978-92-76-11263-1, doi:10.2760/303163, JRC115607.

# Contents

## Foreword

The **Joint Research Centre's technical and scientific work plays a major role in its support to developing EU policies**. The Joint Research Centre both manages and develops knowledge, ensuring that emerging policies of the European Commission are based on the **best available scientific evidence, a cornerstone for robust policymaking**.

This *Science for Policy* report *Water quality in Europe: effects of the Urban Wastewater Treatment Directive* demonstrates how science informs policy directly.

Assessing the effects of EU legislation over the duration of their implementation is an essential element of the European Commission's work, for both the evaluation of policies and informing future policymaking. Only when the best available data and knowledge is used, can it be readily established whether a legislative tool has achieved its objectives. And where there has been no contemporaneous tracking of the effects of individual Directives, it is essential to use solid analysis tools to establish retrospectively the impact an intervention has had.

The Urban Waste Water Treatment Directive (UWWTD) is a key element of the water *acquis* in EU legislation. This report lays down a methodological approach, with results of the assessment of the effects of the UWWTD. The findings in this report substantively support the REFIT Evaluation of the Directive, with the assessment as to whether it has delivered on its objectives and remains fit for purpose.

The results demonstrate that the UWWTD has played a crucial role in the reduction of pollutant loads due to the resulting increase in the treatment of waste water across member states, thus protecting the water quality of EU's water bodies. The implementation of the UWWTD supported the increase in the number of bathing sites that used for recreation, and has also resulted in in the reduction of pollutants in urban waste water that were not directly targeted by the Directive's provisions.

The Joint Research Centre's report played a **pivotal role in the delivery of a strong Evaluation** that demonstrates clearly the effects of the UWWTD. The evidence from this report provides a thorough basis for any further work that may be carried out with regards to the Directive.

In light of the forthcoming **European Green Deal,** that will provide the impetus for Europe to become the first climate neutral continent, support a **Biodiversity Strategy for 2030,** and move Europe **towards a zero-pollution ambition**, more of this calibre of work is needed, thereby ensuring that **policies can be both ambitious and grounded in the best available scientific evidence**.

Daniel Calleja
The Director-General

## Acknowledgements

This work was developed in support to the fitness check of the Urban Wastewater Directive, in connection with the Blue2 project (DG ENV and JRC), and grounds on a close collaboration with Nele-Frederike Rosenstock, Michel Sponar and other colleagues at DG ENV. The Authors wish to thank Fayçal Bouraoui, Antonio Franco and Arwyn Jones of the JRC for their review of the work.

***Authors***

Pistocchi, A., Dorati, C., Grizzetti, B., Udias, A., Vigiak, O., Zanni, M.

## Executive summary

### *Policy context*

This report was prepared in support to the retrospective evaluation of the Urban Wastewater Treatment Directive (UWWTD) 91/271/EEC, aimed at understanding inter alia the effectiveness of this piece of legislation, its relevance and its value added for Europe. The main questions addressed in this report are:

- To what extent are the main pollutants released by urban areas collected and treated?
- What have been the (quantitative and qualitative) effects of the UWWTD?

In order to answer these questions, we have considered past (pre-Directive) and present (baseline) conditions of wastewater treatment in Europe, and we have compared these with scenarios of full compliance with the Directive and a hypothetic evolution of the pre-Directive situation in the absence of the UWWTD. Under each of these scenarios, we have used models to estimate loads and concentrations of contaminants in order to appraise the effects of the Directive on the status of European waters.

### *Main findings and Key conclusions*

The UWWTD has achieved a significant reduction of urban wastewater emissions of organic matter, nutrients and coliforms to surface waters. The improvement in the ecological status of inland water bodies has been less than proportional to the reduction of these emissions, because of the presence of other sources of pollution (in particular agriculture and livestock), not affected by the Directive.

More apparent is the consequence of this significant reduction of emissions on the conditions of bathing waters in Europe, where the UWWTD has allowed a clear improvement in meeting excellent and good standards for faecal coliforms in inland as well as coastal waters.

For chemicals present in urban wastewater, the treatment of organic matter and nutrients required by the Directive may allow a significant reduction of emissions only if the molecules are either removed in the bioreactors, or sorbed to solids treated as sludge. In this case, however, application of sludge to soils may cause a transfer of pollution from the receiving surface waters to soils and, consequently, groundwater. In addition, there are chemicals which are virtually unaffected by conventional wastewater treatment. For these, the UWWTD does not contain any specific provision.

In the absence of the Directive, it can be imagined that certain EU Member States would have implemented wastewater treatment in a way similar to the requirements of the UWWTD, while other countries would have made much lesser efforts in this direction, leading to potentially very inhomogeneous situations across the EU.

In spite of the good progress made by most Member States, implementing wastewater treatment to full compliance with the Directive would still bring significant reductions of pollutant loads. However, the loads of wastewater still to be treated are comparable with other sources of pollution, namely urban runoff and combined sewer overflows. Both of these sources are not specifically addressed by the Directive, which contains only general principles regarding diffuse urban pollution and stormwater.

The Directive allows Member States to adopt individual and appropriate systems (IAS) to treat parts of the agglomerations served by sewer networks, where connections would not be economically justified. When such IAS are managed in a way that does not guarantee compliance with the standards of the Directive, there may be significant local impacts on the quality of the receiving water bodies. However, as IAS treat only a small part of the EU population equivalents, their impact at the EU scale is small. Similar considerations apply to the case of scattered dwellings, i.e. individual households or agglomerations of less than 2000 population equivalents (PE). These fall outside of the scope of application of the UWWTD. While certain Member States have provisions in place applicable to scattered dwellings, others do not, hence local impacts on water bodies may be significant.

Finally, while reducing urban loads is important, for both BOD and nutrients an improvement of the conditions of the receiving water bodies requires addressing also agricultural sources.

### *Related and future JRC work*

This report grounds on data and model developments described in Pistocchi and Dorati, 2018, Vigiak et al., 2018, and Vigiak et al., 2019. Part of the work was developed in the context of the Blue2 project[1]. A European-wide assessment of combined sewer overflows (CSO) and the management of urban runoff is forseen in the current workprogramme of the JRC as a follow-up of the preliminary assessment presented here. Particularly for CSO, it is expected that new knowledge may become available in the near future, which may require revisiting the assessment presented here.

[1] http://ec.europa.eu/environment/blue2_en.htm

***Quick guide***

After introducing the context of this study, §2 illustrates the models and scenario assumptions adopted in the assessment ; §3 presents the outcomes of the assessment in terms of water quality in Europe ; and finally §4 discusses these outcomes and draws conclusions for policy making based.

# 1 Introduction

The Council Directive of 21 May 1991 concerning urban wastewater treatment (91/271/EEC), hereinafter UWWTD, has given the framework for the development of the European wastewater treatment system. More than a quarter of a century after its entry into force, the European Commission has undertaken to subject this Directive to a retrospective evaluation in the framework of the Better Regulation[2] initiative and the Regulatory Fitness and Performance (REFIT) Programme[3].

In the context of the REFIT evaluation of the UWWTD, this report contributes to addressing the following, interrelated evaluation questions:

- To what extent have the objectives of the UWWTD been achieved?
- To what extent are the main pollutants released by urban areas collected and treated?
- What have been the (quantitative and qualitative) effects of the UWWTD?

This study quantifies the reduction of nutrients and organic matter in European waters as a consequence of the implementation of the UWWTD. To this end, we first calculate loads and concentrations of organic matter, measured as 5-days Biochemical Oxygen Demand ($BOD_5$), total nitrogen (N), and total phosphorus (P) in different scenarios representative of the past (before entry into force of the Directive), present and future under assumptions of a full implementation of the Directive. $BOD_5$, N and P are, together with total suspended solids (TSS), specific targets for urban wastewater treatment.[4]

In addition, we estimate concentrations of indicator coliforms in European rivers and coastal areas. Coliforms are not a target of the UWWTD, but they are directly relevant for the quality of bathing waters.

Finally, we estimate the reduction of loads of selected micropollutants achieved through implementation of the UWWTD. Micropollutants are also not a target of the UWWTD, but their environmental fate and transport may be very significantly affected by the wastewater treatment system, as many of them are emitted primarily in urban areas and pass through wastewater treatment plants. Europe-wide quantification of loads is not possible at present, in absolute terms, for the majority of micropollutant. However, based on the properties of the substances it is possible to quantify the percentage of the micropollutants entering the wastewater treatment system, which are eventually released in the environment.

In this study, we refer to “ordinary” conditions intended as those of dry weather, where all wastewater collected in European agglomerations is expected to undergo treatment. It is well known that, during storm events, a part of the collected wastewater may be released untreated to the environment through combined sewer overflows (CSO). While local studies have been conducted in several cases to characterize and minimize CSO, no systematic and comprehensive information exists to date about the “big picture” of pollution from CSO in Europe. In this study we attempt a first-ever modelling of CSO volumes at the large scale (whole EU), and their interpretation in terms of pollutant loads (N, P, $BOD_5$, coliforms and micropollutants).

In the remainder of the report, after explaining in detail the methods, assumptions and data used in this study, we present the results and provide a first contribution to the answer of the abovementioned evaluation questions.

[2] https://ec.europa.eu/info/better-regulation-guidelines-and-toolbox_en

[3] https://ec.europa.eu/info/law/law-making-process/overview-law-making-process/evaluating-and-improving-existing-laws/reducing-burdens-and-simplifying-law/refit-making-eu-law-simpler-and-less-costly_en

[4] TSS originate from various sources and undergo a multitude of environmental processes, and their modelling in the European stream network is very complex. Due to the limited resources available, TSS could not be included in this study.

# 2 Methodology

## 2.1 Assessment endpoints

The UWWTD addresses pollution which in turn affects the quality of waters for human use, and the status of aquatic ecosystems. The legislation of the EU sets quality standards for different parameters depending on the protection goals (Table 1). In order to appreciate the impact of the UWWTD, in principle, we need to quantify the improvement with respect to the quality standards that can be attributed to this Directive.

*Table 1- assessment endpoints, and corresponding protection goals set in the EU legislation*

| | Protection goal | | |
|---|---|---|---|
| **Pollutant** | **Aquatic ecosystems** | **Drinking Water** | **Bathing Waters** |
| N | X | X | |
| P | X | | |
| BOD | X | | |
| Indicator coliforms | | | X |
| Chemicals | X | X | |

Directive 2006/7/EC (the Bathing Water Directive, BWD) sets limits to the concentration of Intestinal Enterococci and Escherichia Coli in bathing waters, with limits in inland waters about twice as high as in coastal and transitional waters (Table 2).

*Table 2 – quality standards for Intestinal Enterococci and Escherichia Coli in the Bathing Waters Directive (BWD)*

| Quality class | Intestinal enterococci (CFU/100 ml) | | Escherichia Coli (CFU/100 ml) | |
|---|---|---|---|---|
| | Freshwaters | Coastal and transitional waters | Freshwaters | Coastal and transitional waters |
| Excellent | 200[5] | 100[5] | 500[5] | 250[5] |
| Good | 400[5] | 200[5] | 1000[5] | 500[5] |
| Sufficient | 330[6] | 185[6] | 900[6] | 500[6] |

The model used in this exercise refers to faecal coliforms, which are typically well correlated to Escherichia Coli. Francy et al., 1993, present regression models to predict Escherichia Coli from faecal coliform concentration in American rivers, highlighting a typical quasi-linear relationship where the former is typically between 0.6 and 1.5 times as high as the latter. Given the uncertainties in the emission factor and die-off rates adopted for faecal coliforms, it appears reasonable to assume the two to have the same numerical value. Therefore quality standards for Escherichia Coli will be applied *tout court* to faecal coliforms.

The Drinking Water Directive (DWD)[7] includes parameters and parametric values used to assess the quality of water intended for human consumption. A recent Commission proposal for a revision of the DWD[8] expands the list of parameters and modifies some of the limits. In this report, we focus on those substances that can pollute drinking waters, and whose origin may be related to wastewater. These include metals, nitrate, and household or industrial chemicals. We deliberately ignore the cyanotoxin

[5] 95th percentile of measured concentrations.
[6] 90th percentile of measured concentrations.
[7] Directive 1998/83/EC
[8] http://ec.europa.eu/environment/water/water-drink/review_en.html

Microcystin-LR, pesticides[9], disinfection by-products (Haloacetic acids, trihalomethanes) and some inorganic parameters not expected to be specifically associated to wastewater pollution (fluoride, chlorate, chlorite, boron, bromate, cyanide).

*Table 3– parameters of the DWD considered in this study*

| Parameter | Parametric value |
|---|---|
| Nitrate (NO3) | 50 mg/L (11.3 mg/L as N) |
| Cu | 2 mg/L |
| U | 30 ug/L |
| Cr | 25 ug/L |
| Ni | 20 ug/L |
| As, Se, Tri- and Tetrachloroethene | 10 ug/L |
| Cd, Sb, Pb | 5 ug/L |
| 1,2 - Dichloroethane | 3 ug/L |
| Hg, Benzene | 1 ug/L |
| Vinyl chloride | 500 ng/L |
| Nonylphenol | 300 ng/L |
| Acrylamide, PFAS, PAHs, Epichlorohydrin | 100 ng/L |
| Bisphenol A, Benzo(a)Pyrene | 10 ng/L |
| Beta-Estradiol | 1 ng/L |

In the context of the Water Framework Directive (WFD)[10], the ecological status of water bodies is expected to reflect also the conditions of nutrients and organic matter. Concentrations of $BOD_5$ may be interpreted having in mind the criteria presented in Table 4.

Concentrations of nutrients are more difficult to interpret in terms of the status of water ecosystems, exhibiting a significant dispersion of values for the thresholds between good/moderate and high/good status. The ECOSTAT Guidance document on purpose[11] provides an overview of nutrient standards set across different regions in Europe (tables from p. 60 onwards of the Guidance), highlighting a broad variability. Grizzetti et al., 2017, show that the median of predicted concentrations for rivers in good status is normally about 4 mg/L for total N and 0.1 mg/L for total P, and we use these thresholds for the purposes of this exercise.

For what concerns chemicals, the Environmental Quality Standards Directive (EQSD)[12] sets limits for 45 priority substances including pesticides, household and industrial chemicals. Of these, some are already included in the parameters of the DWD (Benzene, Cd, 1,2-Dichloroethane, Pb, Hg, Ni, Nonylphenols, PAHs, PFAs). Here we deliberately exclude pesticides and we focus on those substances that can be expected to relate to urban wastewater (taking also into account assimilated industrial wastewater). The EQS for the priority substances considered here are reported in Table 5.

The EQSD introduces also a “watch list” of chemicals of emerging concern, including pesticides (which we do not further consider) and chemicals widely used in households: macrolide antibiotics (erythromicyn, clarithromycin, azithromycin), hormones

[9] Pesticides are commonly reported to be present in raw and treated wastewater, and they may end up more quickly in water after use in urban settings than in agriculture. However, in this assessment we assume their main source to be agricultural use, which we expect to dominate in comparison with domestic use and use in urban green spaces. A more specific assessment is beyond the scope of this work.

[10] Dir. 2000/60/EC

[11] Common Implementation Strategy For The Water Framework Directive And The Floods Directive - Best Practice for establishing nutrient concentrations to support good ecological status https://circabc.europa.eu/sd/a/5aa80709-9ce8-411d-94e8-f0577f3632fa/Nutrient_standard_Guidance_for_CIS_final%20with%20links%209%20May.pdf

[12] Directive 2013/39/EU

(ethinylestradiol, estradiol – already considered by the DWD- and estrone), diclofenac, BHT and EHMC. For these substances there is no EQS.

*Table 4 – thresholds for good and high ecological status with respect to BOD5.*

| **Region**[13] | **High-good threshold (mg/L BOD5), i.e. reference conditions** | **Good-moderate threshold (mg/L $BOD_5$)** |
|---|---|---|
| General EU-wide indication | 2 to 3 | 4 to 5 |
| Alpine reference criteria | Mean < 2<br>90% < 2.75 | |
| Central-Baltic reference criteria | Mean < 2.4 (< 2 for R-C3)<br>90% <3.6 (< 2.75 for R-C3) | |
| Eastern Continental reference criteria | Mean < 2.4 (< 2 for R-E1a and R-E1b) | |
| Northern rivers (applied by UK and IE) reference criteria | Mean < 1.6 | |
| Median value at sites in harmonised high status (Central-Baltic) | 1.5 - 1.9 | |
| Danube basin countries: criteria for high and good status | < 2.5 | <5 |
| Belgium –Flanders | | < 4 |
| France[14] | <3 | < 5 |
| Italy[15] | <2.5 | < 4 |
| Estonia (very large rivers) | < 2 | |
| Hungary (very large rivers) | <3 | |
| Latvia (very large rivers) | < 2.7 (BOD7) | |
| EEA[16] | <2 | <5 |
| Various national and international classification schemes[4] | | <5 |

*Table 5 – EQS for the priority substances of the WFD considered in this study*

[13] "Alpine", "Central-Baltic", "Eastern Continental" are Geographic Intercalibration Groups (GIG) – see http://publications.jrc.ec.europa.eu/repository/bitstream/JRC99231/lb-na-27707-en-n%20.pdf

[14] Brunel et al., 1997 classifies: very good <3, good 3-5, average 5-10; bad 10-25; very bad >25 mg/l

[15] The Italian legislation antecedent to the WFD (Legislative Decree 152/1999, annex I, table7) contains 4 threshold values for BOD, corresponding approximately to high, good, moderate, poor and bad ecological status. The threshold values are 2.5, 4, 8 and 15 mg/L, respectively.

[16] Kristensen & Hansen 1994, report that river reaches little affected by human activities generally have BOD < 2 mg/l, whereas BOD5 exceeding 5 mg generally indicates pollution (BOD is lower than 2 mg/l in catchments with less than 15 inhabitants km2, it generally exceeds 5 mg/l in catchments with more than 100 inhabitants km2)

[4] ICPDR, 2004; Newman, 1988; MMGA, 2006

| Parameter | Parametric value (ug/L) |
|---|---|
| Anthracene | 0.1 |
| Brominated diphenylethers | 0.14 (maximum concentration only. For non-inland waters, 0.014) |
| Chloroalkanes C10-13 | 0.4 (annual average concentration), 1.4 (maximum concentration) |
| Dichloromethane | 20 (annual average concentration only) |
| DEHP | 1.3 (annual average concentration only) |
| Fluoranthene | 0.0063 (annual average concentration), 0.12 (maximum concentration) |
| Hexachlorobenzene | 0.05 (maximum concentration only) |
| Hexachlorobutadiene | 0.6 (maximum concentration only) |
| Naphtalene | 2 (annual average concentration), 130 (maximum concentration) |
| Octylphenols | 0.1 or 0.01 (annual average concentration only) |
| Pentachlorobenzene | 0.007 or 0.0007 (annual average concentration only) |
| Tributyltin compounds | 0.0002 (annual average concentration),<br>0.0015 (maximum concentration) |
| Trichlorobenzenes | 0.4 (annual average concentration only) |
| Trichloromethane (chloroform) | 2.5 (annual average concentration only) |
| Hexabromocyclododecanes | 0.0016 or 0.0008 (annual average concentration), 0.5 or 0.05 (maximum concentration) |
| Benzene | 8 or 10 (annual average concentration), 50 (maximum concentration) |
| Cd | 0.08 to 1.5 (depending on water hardness and water body type) |
| Pb | 1.2 or 1.3 (annual average concentration), 14 (maximum concentration) |
| Hg | 0.07 (maximum concentration only) |
| Ni | 4 or 8.6 (annual average concentration) ; 34 (maximum concentration) |
| PAHs | 1.7 × 10–4 (annual average concentration), 0.27 or 0.027 (maximum concentration) ; for Benzo(a)pyrene |
| PFOS | 6.5 × 10–4 or 1.3 × 10–4 (annual average concentration), 36 or 7.2 (maximum concentration) |
| Nonylphenols | 0.3 (annual average concentration),2 (maximum concentration) |
| 1,2-Dichloroethane | 10 (annual average concentration only) |

In the last years, increasing concern is being raised by a multitude of chemicals, particularly pharmaceuticals, undergoing widespread household use, and therefore strongly associated to wastewater. In this exercise we take into consideration the following as examples: medicines Ibuprofen, Carbamazepine, Ciprofloxacin, Fluoxetine, Sertraline, Atorvastatin, Simvastatin, antibacterial Triclosan, and Octamethylcyclotetrasiloxane (D4) used in cosmetics. Also for these substances there is no EQS.

It should be stressed that the above chemicals do not necessarily represent a list of substances of priority concern, but are considered as examples of substances affected to a different extent by wastewater treatment.

## 2.2 Approach to modelling

### 2.2.1 Nutrients and organic matter

In this study, we use the Geospatial Regression Equation for European Nutrient losses (GREEN) model to simulate scenarios of emissions of nutrients (Grizzetti et al. 2012; Bouraoui et al. 2011) to predict annual loads of nitrogen (N) and phosphorus (P) from point and diffuse sources, at the spatial resolution of catchments. Additional details on the model setup can be found in Pistocchi et al., 2017.

The GREEN model has been generalized to describe organic matter (expressed as biochemical oxygen demand after 5 days - $BOD_5$, or BOD *tout court*). To this end, the parameterization of retention in rivers and lakes has been made following the concept of the MAPPE model (Pistocchi, 2014), which can be regarded as a generalization of the GREEN model (see Pistocchi, 2019). In this application, point sources of BOD comprise urban waste water treatment plant (UWWTP) discharges and industrial discharges, and are considered to be discharged directly in the main subbasin reach. Diffuse sources encompass organic waste from livestock, scattered dwellings, wash-off from urban areas, plus organic matter originated from natural areas, like forests and inland waters. Diffuse sources are considered to be attenuated by residence time in the subbasin before reaching the main stream network. The residence time in the subbasin was based on time lag estimated as a simple empirical function of subbasin area (Gericke and Smithers, 2014). However, diffuse sources may follow other pathways, e.g. subsurface lateral flow, with longer travel time. To adapt travel time by different pathways, source specific delay times [days] were added to basin travel time to obtain the travel time in the subbasin per source. A delay for domestic waste from disconnected dwellings (#DD; in days) and one for extensive livestock system waste (#LVST; days) were added to consider waste leaching in the soil and transport via subsurface pathways. Residence time in the rivers [days] was calculated on the basis of reach length and water velocity following Pistocchi and Pennington, 2006. Attenuation is computed as an exponential decay function of residence time. The water flow used to compute velocities, as well as concentrations from loads, is the estimate of long-term average runoff according to the model of Budyko (see Pistocchi et al., 2019). As in Wen et al. (2017), the first order degradation rate coefficient was derived from the decay rate at 20 °C ($k_{20}$, Table 1) adjusted for water temperature. Water temperature across Europe was assessed with a log-linear relationship with mean annual air temperature (Vigiak et al., 2017). Table 6 summarizes the model parameters adopted in this study. An extensive presentation of the model is offered in Vigiak et al., 2019.

In this work, we examine different scenarios of discharges of domestic wastewater (see §2.3), but the models for nutrients and BOD are calibrated taking into account also non-domestic sources of emissions. The GREEN model accounts for mineral and manure fertilizers, scattered dwellings and atmospheric deposition of nitrogen (see Pistocchi et al., 2017, for additional details). For BOD, we include industrial emissions, livestock, and runoff from urban and natural areas.

Industrial releases from large facilities that directly treat and discharge their waste were taken from the European Pollutant Release and Transfer Register database (E-PRTR; EEA, 2018). UWWTPs included in this database were removed to avoid duplications with domestic waste. Annual facility releases to water (net to transfers) of Total Organic Carbon (TOC) for the seven-year period 2010-2016 were averaged to obtain mean annual facility TOC emissions [t/y]. BOD was estimated from TOC based on the equivalence BOD=1.85 TOC. This is a simplification, as the relationship between the two measures depends on type of industrial waste (e.g. Dubber and Gray 2010; Christian et al., 2017).

Livestock waste is a major source of BOD pollution globally (Wen et al., 2017) and in Europe (Malve et al., 2012). The 1 km2 global distribution of livestock maps for the reference year 2005 (Robinson et al., 2014) were used to assess livestock heads per type and subbasin. Livestock waste was estimated following Wen et al. (2017). Livestock waste was divided into intensive or extensive production systems. Intensive systems were considered of industrial type and treated at secondary level before being discharged to the main reach as point sources. Intensive systems comprised all pig and chicken, as well as cattle and goat/sheep of density higher than 25 Livestock Standard Units (LSU)/km2. Conversely, extensive systems were considered diffuse sources of BOD and abated through basin retention before reaching the stream network. Extensive systems comprised cattle and goat/sheep bred at density lower than 25 LSU/km2.

To account for BOD washed off by urban runoff collected in separate sewers, we considered urban land as an additional diffuse source estimated as proportional to annual urban runoff volume ($m^3$) occurring in a subbasin assuming a effective mean concentration $EMC_U$ (Table 1). Urban runoff was derived from Pistocchi et al., 2017.

Organic matter washed off from natural areas to freshwater systems contributes to biodegradable material and to BOD monitored in running waters (e.g. HELCOM, 2004). While this is not a source of pollution, it nevertheless contributes to BOD in freshwater systems. Soils in natural areas are richer in litter and organic matter content than agricultural fields and may contribute to organic matter in freshwater systems significantly. Additionally, lakes (including artificial reservoirs) can act as sinks or sources of BOD to downstream waters. Their role depends on several factor, among which trophic status, hydrology and retention time, type and amount of incoming organic matter, and location within the freshwater network. Autochthonous releases of organic matter are generally more biodegradable than incoming sources, for examples when particulate organic matter in sediments is transformed into more liable dissolved organic components. As a result, BOD outflowing flux may be higher than incoming BOD influx, although switches may occur with rainfall events and seasonality. BOD emissions from natural areas (including lakes) were assumed to be proportional to the extent of natural land by a simple export coefficient method, adopting a constant area emission ($E_{NAT}$; t/km2, Table 1).

*Table 6- BOD Model parameters description and adopted value.*

| Parameter | Description | Value |
|---|---|---|
| Eff.1 [-] | BOD removal efficiency for primary treatment | 0.50 |
| Eff.2 [-] | BOD removal efficiency for secondary treatment – applied also to high density livestock treatment | 0.94 |
| Eff.3 [-] | BOD removal efficiency for tertiary treatment | 0.96 |
| Eff.SD[-] | BOD removal efficiency for septic tanks | 0.40 |
| #DD [days] | Retention days in excess of basin travel time employed by disconnected domestic waste to reach stream network | 7 |
| #LVST [days] | Retention days in excess of basin travel time employed by low density livestock waste to reach stream network | 7 |
| $k_{20}$ [days$^{-1}$] | Freshwater BOD retention | 0.56 |
| $EMC_U$ [mg L$^{-1}$] | Effective mean concentration of BOD in urban wash-off | 11 |
| $E_{NAT}$ [t km$^{-2}$ y$^{-1}$] | BOD export coefficient of natural areas | 0.16 |

Wastewater treatment (WWT) is represented through a removal efficiency of pollution, specific for $BOD_5$, N and P. the parameters adopted in this study are shown in Table 7.

*Table 7 – WWT removal efficiency for organic matter and nutrients*

| substance | Primary | Secondary | More stringent |
|---|---|---|---|
| N | 25% | 55% | 80% |
| P | 30% | 60% | 90% |
| BOD5 | 50% | 94% | 96% |

### 2.2.2 Coliforms

For indicator coliforms, no specific model is presently available off-the-shelf for application at European scale; therefore we use the same model used for BOD with the assumptions explained hereafter. This application of the model was not calibrated due to current unavailability of homogeneous data on coliforms at European scale. Moreover, we include only emissions from domestic wastewater, thus neglecting other potentially important sources such as urban runoff and livestock. The model is therefore valid not in absolute terms, but only for comparison among scenarios. For the generic WWTP at site (x,y), the load of fecal coliforms (used as indicator) is computed as:

$$L(x,y) = E_0 PE(x,y)\, \eta_T(x,y) \qquad \text{Equation 1}$$

where $E_0$ is the coliform excretion rate, assumed to be constant across Europe, assumed $E_0$=450 $10^{10}$ CFU/PE/year following Reder et al., 2015; $\eta_T$ is (1 - treatment efficiency) of treatment level T (primary, secondary, or more advanced treatment)[17], assumed to be constant for a given level of treatment across Europe, PE is the population-equivalent treated by the WWTP. Concentrations in the generic w-th receiving river water bodies is estimated as:

$$C_w = E_0 \frac{\int_{A_w} PE(x,y)\eta_T(x,y)e^{-k_w\tau(x,y)}dxdy}{Q} \qquad \text{Equation 2}$$

where $A_w$ is the catchment area of the w-th water body, $\tau(x,y)$ is the water travel time from site (x,y) to the w-th water body, Q is river water discharge (estimated with the Budyko model as for nutrients and BOD), and $k_w$ is a decay constant accounting

[17] Treatment efficiency is meant as the ratio of concentrations after (numerator) and before (denominator) treatment.

for coliform die-off in rivers and lakes. For rivers, $k_w$is expected to be in the range 0.1 to 30 day$^{-1}$ (Metcalf & Eddy, 1991) and can be represented through the Mancini equation (see Metcalf and Eddy, 1991, p. 1206) written in the form:

$$k_w = 0.2067e^{0.0677\,T} \qquad \text{Equation 3}$$

with T=river water temperature (°C). We assume the values of $\eta_T$ summarized in Table 8. The above simple model is applied to simulate the same scenarios discussed above. The level of treatment in place at each WWTP determines the treatment efficiency. For wastewater discharged raw to the environment, $\eta_T$=1 (i.e. zero treatment efficiency). Water discharge Q and water travel time $\tau(x, y)$ are the same as used to model $BOD_5$.

*Table 8. Removal efficiencies and $\eta_T$ for coliforms after wastewater treatment.*

| **Level of treatment** | **Lower removal eff.**[18] | **Higher removal eff.**[18] | **Assumed** |
|---|---|---|---|
| No treatment | 0% | 0% | 0% |
| Primary | 29.2% | 68.4% | 40% |
| Secondary | 90.0% | 99.9% | 95% |
| More stringent | Up to 100% | | 99.9% |

For marine coastal waters, we assume that concentrations of coliforms decrease with the distance from the point of release of inland emissions, following an exponential curve (see Annex I for details):

$$C_c = C_0 e^{(-\alpha R)} \qquad \text{Equation 4}$$

where $C_0$ is the concentration at the emission point, R is the distance from the point, and $\alpha$ is a coliform dissipation constant in coastal waters. In this exercise, we set $\alpha$ = 0.002 m$^{-1}$ based on the arguments presented in Annex I. The distance where $C_c = T$ can be obviously computed as:

$$d_T = \frac{\ln(C_0) - \ln(T)}{\alpha} \qquad \text{Equation 5}$$

We may consequently estimate that, in a stretch of coastline of length L where there are n emission points, each having a representative concentration $C_{0,i}$ (i=1:n), the percentage of the coastline with concentrations below a threshold T can be estimated as:

$$B_T = 1 - \frac{\sum_{i=1}^{n} \ln(C_{0,i}) - n\ln(T)}{\alpha L} \qquad \text{Equation 6}$$

In this exercise, we compute the percentage of European coastlines, as well as the percentage of coastlines for the EU's NUTS2 regions[19], where the thresholds for excellent and good quality of the BWD, T=250 CFU/100 ml and T=500 CFU/100 ml, are met, assuming that the only sources of coliforms are the emissions from wastewater in the sub-basins. In this way, we completely neglect the presence of devices for the disposal of wastewater, such as offshore diffusers. We compute the concentration of faecal coliforms that we expect in the stream at the outlet of each coastal sub-basin, based on the calculation of $C_w$ discussed above. This reflects the level of treatment in the sub-basin, as well as the emission sources upstream having accounted for the die-off of coliforms along their trajectories.

### 2.2.3 Chemicals

Chemical micro-pollutants include a very high number of substances, and are not an explicit target of urban wastewater treatment. However, it is broadly recognized that many of these substances are emitted to the environment from urban areas and households, hence WWTPs may be a very important component of any strategy to control their concentration in receiving waters.

In this study, we consider as an example a set of 54 representative micropollutants as discussed in §2.1. These are present in raw wastewater in highly variable concentrations, and their removal efficiency is also highly variable at the different levels of

[18] Range suggested for primary treatment and conventional activated sludge processes by Oakley, 2018, and Naughton and Rousselot, 2017, respectively.
[19] https://ec.europa.eu/eurostat/web/nuts/background

treatment. Table 10 provides examples of reported concentrations in wastewater and treatment efficiencies for some of the micro-pollutants. Each of these chemicals may undergo removal in WWTPs to a different degree, and the removal efficiency of each single molecule appears to depend considerably on the technology and operating conditions of the WWTP, making it impossible to directly relate the removal efficiency to the level of treatment (mechanical, biological and removal of nutrients) as categorized in the UWWTD, in a simple way.

Given the high variability of removal efficiencies in real WWTPs, in order to achieve a conceptually consistent representation we modelled a "reference" removal efficiency for selected micropollutants, under conventional assumptions. The removal efficiency is modelled on the basis of the physico-chemical properties of the molecules, assuming a schematic layout of a WWTP.

To date, a model used for the prediction of chemical removal in WWTPs in a European regulatory context is SimpleTreat (Struijs, 2014). In SimpleTreat, a WWTP is described as a combination of a primary settler, a biological activated sludge bioreactor and a secondary settler (Figure 1).

Using SimpleTreat with the default values of the operational parameters of a WWTP (Table 9), we conducted an exploratory analysis of the behaviour of chemicals in WWTPs with reference to 28 representative example chemicals out of the 54 considered in this study (Table 11).

It should be stressed that chemicals feature a rather complex behaviour in wastewater treatment processes, particularly when they are ionisable. A realistic simulation with SimpleTreat requires an accurate compilation of physicochemical property data, entailing an effort beyond the limits of the illustrative exercise presented here.

*Figure 1 – schematic of the WWTP represented in SimpleTreat. Source: Struijs, 2014.*

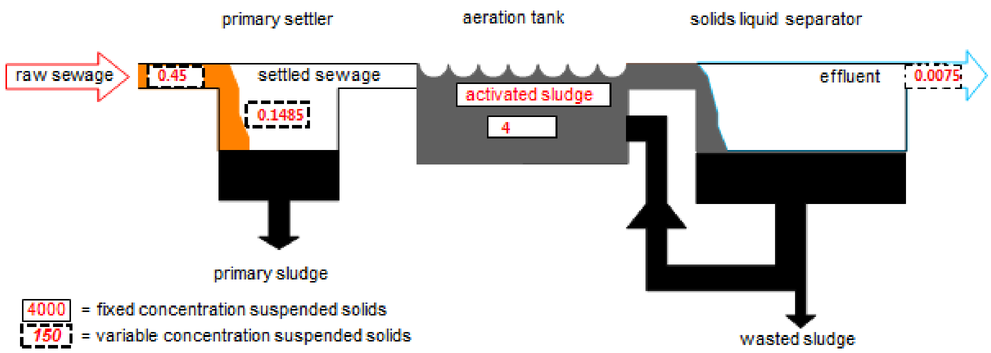



*Table 9 – default operational parameters assumed for the WWTP*

| **Operational parameter of WWTP** | **Default** |
|---|---|
| Sewage flow | 0.2 $m^3$ $d^{-1}$ $PE^{-1}$ |
| Mass of sewage suspended solids | 0.09 kg $d^{-1}$ $PE^{-1}$ |
| BOD5 in sewage suspended solids | 60 g $O_2$ $d^{-1}$ $PE^{-1}$ |
| Fraction of sewage suspended solids removed in primary settler | 2/3 |
| pH | 7 |
| Concentration of suspended solids in effluent | 7.5 mg/L |
| Aeration system | Surface aeration |

As we expect a higher removal of degradable micropollutants in WWTPs equipped for nutrient removal, compared with plants designed only for $BOD_5$ and TSS removal (though this may not always be true in reality), we computed the removal of a unit load inflowing to a WWTP for selected chemicals, under the assumption of a food-to-mass ratio (F/M) equal to 0.05 and 0.2 kg O2 $kg_{dw}^{-1}d^{-1}$, respectively, to conventionally represent plants with and without nutrient removal. This reflects the general tendency to have higher sludge retention times (lower F/M ratios) in the former compared to the latter. The physico-chemical properties of the substances, used as input to the calculation, and the resulting modelled removal efficiencies are summarized in Table 11. The calculation highlights a broad variability in the removal mechanisms. In general, a trade-off can be observed between sorption in sludge and removal through degradation and/or volatilization (Figure 2). Moreover, there are many chemicals not appreciably retained in solids but, at the same time, rather persistent.

The relative importance of removal through sorption and through degradation/volatilization allows categorizing 6 classes of chemicals in terms of their behaviour in WWTPs:

1) Chemicals practically bypassing wastewater treatment, virtually unaffected
2) Slowly removed chemicals, with limited or no accumulation in sludge;
3) Moderately removed, with limited or no accumulation in sludge;
4) Removed, with limited or no accumulation in sludge;
5) Sorbed to sludge and slowly removed;
6) Sorbed to sludge (but not appreciably removed).

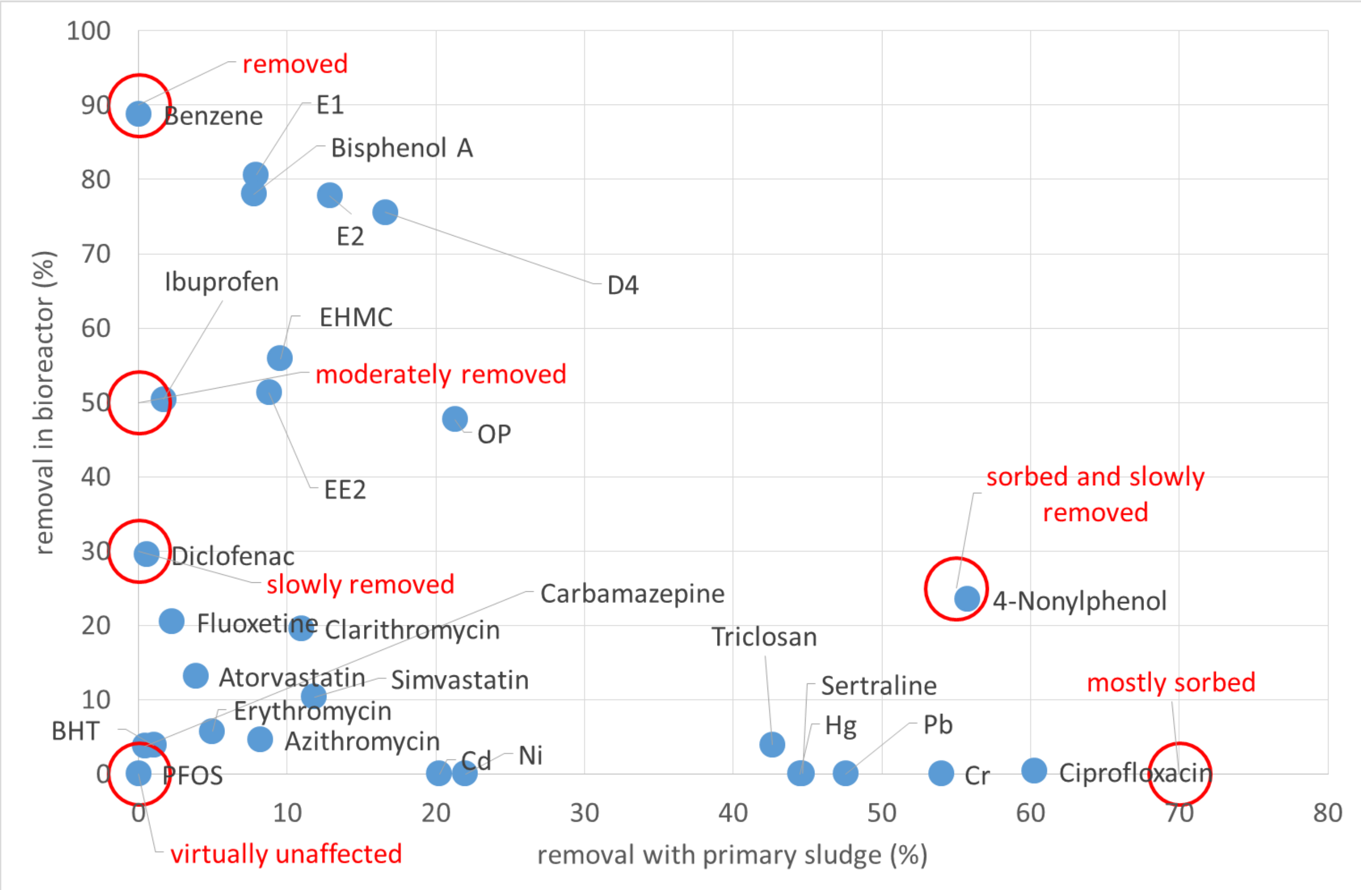
These categories define 6 prototypal behaviours of chemicals in WWTPs, possibly encompassing the whole "chemical universe": the UWWTD may have enabled the removal of chemicals of all classes except the first one, to an extent depending on the properties of the chemical. For chemicals in the last two classes, though, sorption to sludge may indicate that pollution is transferred with the use life of sludge.

The 6 types of fate in the WWTP may be associated to representative removal rates in bioreactors and via sorption to sludge; one possible and practical parameterization, although completely conventional, may be as shown in Figure 2.

Besides treatment efficiency, the impact of the UWWTD on chemicals depends on their persistence in the stream network, which can be represented by a "dissipation half-life" expressing the speed of disappearance of a chemical from the water column. "Dissipation" does not imply that the chemical is degraded or volatilized, as it could also be sorbed to sediments. The half-life of chemicals in river water is as highly variable as their removal efficiency in WWTPs, as clearly shown in Table12, reporting some values of half-lives from the literature. Usually, half-life in rivers cannot be determined with more certainty than within one order of magnitude. Therefore we argue that, for European-scale, screening-level modelling, it may be sufficient to refer to three categories of persistence of chemicals, reflecting orders of magnitude of dissipation rates (DT50=1 day, DT50 = 10 days and DT50 = 100 days).

These considerations suggest that, in spite of the complexity and variety of chemicals present in untreated and treated wastewater, it is possible to study the expected impact of wastewater treatment with reference to a limited list of prototypal substances with an environmental behaviour representative of a whole class of molecules. We propose a list of such prototypal substances, that we call "metachemicals", in Table 13, spanning the range of dissipation in rivers and removal in a WWTP expected for real chemicals. In addition, still relevant is the case of substances virtually bypassing wastewater treatment, for which apparently no impact can be expected from the UWWTD.

*Figure 2 – scatter plot of removal with primary sludge and removal (degradation and/or volatilization) in the activated sludge bioreactor. Red circles represent the properties conventionally attributed to the 6 classes of chemicals: removal with sludge is 0 and removal in bioreactor is 30, 50 and 90% respectively for chemicals slowly removed, moderately removed and removed, respectively; removal with sludge is 25% and removal in bioreactor is 55% for chemicals sorbed and slowly removed; removal with sludge is 70% and removal in bioreactor is 0 for chemicals that are sorbed; both are 0 for chemicals virtually unaffected by WWTPs.*



removal in bioreactor (%)
removal with primary sludge (%)
100
90
80
70
60
50
40
30
20
10
0
0
10
20
30
40
50
60
70
80
removed
Benzene
E1
Bisphenol A
E2
D4
Ibuprofen
EHMC
moderately removed
OP
EE2
sorbed and slowly removed
Diclofenac
slowly removed
Carbamazepine
4-Nonylphenol
Fluoxetine
Clarithromycin
Triclosan
Atorvastatin
Simvastatin
Sertraline
mostly sorbed
Erythromycin
Hg
Pb
BHT
Azithromycin
Cd
Ni
PFOS
Cr
Ciprofloxacin
virtually unaffected

*Table 10 – Chemicals considered in this study. Typical concentrations in wastewater, and removal efficiency for selected chemicals, are provided as a compilation of reported values by Wood Plc. For each chemical we describe a typical expected behaviour, which guides its identification with a "metachemical" as explained in the text below (see Table 13).*

| Chemical | Modelled with SimpleTreat | Metachemical | Behavior | Concentration in wastewater reported in literature[20] | Treatment eff. (primary) from the literature[20] | Treatment eff. (secondary) from the literature[20] | Treatment eff. (more stringent) from the literature[20] |
|---|---|---|---|---|---|---|---|
| Acrylamide | No | c#7 | Highly soluble chemical, can be degraded but not as fast as other chemicals[21] | | | | |
| Sb | No | c#15 | General assumption for metals[22] | | | | |
| As | No | c#15 | General assumption for metals | 0.4-27.8 μg/L[23] | 18-81% (up to 72% from primary treatment) [23] | | |
| Benzene | Yes | c#8 | Volatilized in WWTPs; degraded in rivers but not as fast as other chemicals | 1.3 - 14 mg/L[24] | | 99%[25] | 99%[25] |
| Benzo(a)pyrene | No | c#10 | Non-polar and persistent but settling | | | | |
| Bisphenol A | Yes | c#3 | Degraded in both WWTP and river | | | | |
| Cd | Yes | c#15 | General assumption for metals | 0.4-40 μg/L[23] | 10-79% (up to 41% from primary treatment) [23] | | |
| Cr | Yes | c#15 | General assumption for metals | 8.1-59.2 μg/L[23] | 50-72% (up to 58% from primary treatment) [23] | | |
| Cu | No | c#15 | General assumption for metals | 9.8-301 μg/L[23] | 35-88% (up to 40% from primary treatment) [23] | | |
| 1,2-dichloroethane | | c#8 | Volatilized in WWTPs; degraded in rivers but not as fast as other chemicals | | | | |
| Pb | Yes | c#15 | General assumption for metals | 2-600 μg/L[23] | 31-95% (up to 73% from primary treatment) [23] | | |
| Hg | Yes | c#15 | General assumption for metals | 0.4-27.8 μg/L[23] | 47-90% (up to 42% from primary treatment) [23] | | |

[20] Compilation by Wood Plc.
[21] http://www.who.int/water_sanitation_health/dwq/chemicals/acrylamide.pdf
[22] Metals are not degraded and tend adsorb in sludge; assumed to persist in rivers
[23] P. Cantinho et al. (2016)
[24] Parkerton, T. (2001)
[25] USEPA Drinking Water Treatability Database https://oaspub.epa.gov/tdb/pages/general/home.do

| Chemical | Modelled with SimpleTreat | Metachemical | Behavior | Concentration in wastewater reported in literature[20] | Treatment eff. (primary) from the literature[20] | Treatment eff. (secondary) from the literature[20] | Treatment eff. (more stringent) from the literature[20] |
|---|---|---|---|---|---|---|---|
| Ni | Yes | c#15 | General assumption for metals | 3.5-770 µg/L[23] | 0-44% (up to 43% from primary treatment) [23] | | |
| 4-Nonylphenol | Yes | c#9 | Partly removed and sorbed in WWTP, undergoes some degradation in rivers | 0.029-5.50 µg/L[29] | <33% [25] | 60% [25] | Up to 100%[25] |
| PFOS | No | virtually unaffected by wastewater treatment, conservative | | 2.88 to 176 ng/L[26] | | | |
| Tetrachloroethene | No | c#8 | Volatilized in WWTPs; degraded in rivers but not as fast as other chemicals | | | | |
| U | No | unclear behavior in wwtp/river | | | | | |
| Anthracene | No | c#10 | Non-polar and persistent | | | | |
| Fluoranthene | No | c#10 | Non-polar and persistent | | | | |
| Naphthalene | No | c#10 | Non-polar and persistent | | | | |
| Brominated diphenylethers | No | c#10 | Non-polar and persistent but settling | | | | |
| C10-13 Chloroalkanes | No | c#15 | Non-polar and persistent | | | | |
| Dichloromethane | No | c#8 | Volatilized in WWTPs; degraded in rivers but not as fast as other chemicals | | | | |
| Di(2-ethylhexyl)-phthalate (DEHP) | No | c#15 | Slowly degradable (DT50~50 days)[27] and sorbing to sludge[28] | | | | |

[26] Pan, Y. et al. (2011)
[27] https://echa.europa.eu/documents/10162/060d4981-4dfb-4e40-8c69-6320c9debb01
[28] Marttinen et al., 2003

| Chemical | Modelled with SimpleTreat | Metachemical | Behavior | Concentration in wastewater reported in literature[20] | Treatment eff. (primary) from the literature[20] | Treatment eff. (secondary) from the literature[20] | Treatment eff. (more stringent) from the literature[20] |
|---|---|---|---|---|---|---|---|
| (4-(1,1′,3,3′-tetramethylbutyl)-phenol) (OP) | Yes | c#7 | Moderately degraded | 0.005-0.22 µg/L[29] | 30.60%[30] | | |
| Pentachloro-benzene | | c#15 | Non-polar and persistent | | | | |
| Tributyltin compounds (Tributyltin-cation) | | c#10 | Adsorbed to sludge and settling [31] | | | | |
| Trichloro-benzenes | | c#8 | Volatilized in WWTPs; degraded in rivers but not as fast as other chemicals | | | | |
| Trichloro-methane | | c#8 | Volatilized in WWTPs; degraded in rivers but not as fast as other chemicals | | | | |
| Hexabromocyclododecanes (HBCDD) | | c#15 | Non-polar and persistent | | | | |
| 17-Beta-estradiol (E2) | Yes | c#3 | Degraded in both WWTP and river | <1 – 220 ng/L[42] | | | 77-100%[42] |
| 17-Alpha-ethinylestradiol (EE2) | Yes | c#7 | Degraded but more slowly than E1, E2 | <1 – 8 ng/L[42] | <20%[25] | | 80-100%[42, 25] |
| Estrone (E1) | Yes | c#3 | Degraded in both WWTP and river | <1 – 88 ng/L[42] | | 65-85% | 90-100%[42,32] |
| Diclofenac | Yes | c#6 | Moderately degraded | <loq - 1160 ng/L[33] ; 115.4 ng/L[36] ; 14.9 – 4425 ng/L[42] | 0-58% | | 65-100%[42] |

[29] Jonsson, B. Risk assessment on butylphenol, octylphenol and nonylphenol, and estimated human exposure of alkylphenols from Swedish fish http://www.uu.se/digitalAssets/177/c_177024-l_3-k_jonsson-beatrice-report.pdf
[30] ECHA (2011) SVHC SUPPORT DOCUMENT- 4-(1,1,3,3-TETRAMETHYLBUTYL)PHENOL https://echa.europa.eu/documents/10162/4c6cccfd-d366-4a00-87e5-65aa77181fb6
[31] Voulvoulis et al., 2004
[32] Braga, O. (2005)
[33] Dai et al., 20014

| Chemical | Modelled with SimpleTreat | Metachemical | Behavior | Concentration in wastewater reported in literature[20] | Treatment eff. (primary) from the literature[20] | Treatment eff. (secondary) from the literature[20] | Treatment eff. (more stringent) from the literature[20] |
|---|---|---|---|---|---|---|---|
| Ibuprofen | Yes | c#7 | Moderately degraded | 61 ng/L[36] | <20% | 0-99% | 0-99%[34, 25, 38] |
| Carbamazepine | Yes | virtually unaffected by wwt, conservative | | 90 ng/L[35] ; 49 ng/L[36] ; 747 ng/L[37] | 4.40% | | 83-100%[38, 37, 38] |
| Azithromycin | Yes | virtually unaffected by wastewater treatment, but dissipating in river | | 0.4 – 1220 ng/L[42] | 94.60%[42, 25] | | |
| Clarithromycin | Yes | virtually unaffected by wastewater treatment, but dissipating in river | | 551.3 ng/L[39] ; 129 ng/L[39]; 54 – 1890 ng/L[42] | | | 70-100%[42] |
| Erythromycin | Yes | virtually unaffected by wastewater treatment, but dissipating in river | | 16 – 147.9 ng/L[42] | | | 40-99%[42] |
| Ciprofloxacin | Yes | c#5 | Adsorbed to sludge and settling | | | | |
| Fluoxetine | Yes | c#11 | Degraded to some extent, but persistent in rivers | 11 ng/L[37] | 16% | | 100%[42] |
| Sertraline | Yes | c#15 | Sorbing to sludge, persistent in rivers | 13 ng/L[37] | 28% | | 100%[37] |
| Atorvastatin | Yes | virtually unaffected by wastewater treatment, but dissipating in river | | 37 ng/L[40] | 85 – >90 %.[41] | | |
| Simvastatin | Yes | virtually unaffected by wastewater treatment, but dissipating in river | | 17.8 ng/L[36] | 85->90%[42] | | |
| Triclosan | Yes | c#5 | Mostly adsorbed to sludge but fast-dissipating in rivers | | | | |
| 2,6-Ditert-butyl-4-methylphenol (BHT) | Yes | virtually unaffected by wastewater treatment, conservative | | | | | |

[34] Gros et al. (2010)
[35] Gao et al. (2016)
[36] Magnér et al. (2016)
[37] Lajeunesse et al. (2013)
[38] Sim et al. (2010)
[39] Rodriguez-Mozaz et al. (2015)
[40] Hernando, M.D. et al (2006)
[41] Sulaiman, S. et al., (2015)

| Chemical | Modelled with SimpleTreat | Metachemical | Behavior | Concentration in wastewater reported in literature[20] | Treatment eff. (primary) from the literature[20] | Treatment eff. (secondary) from the literature[20] | Treatment eff. (more stringent) from the literature[20] |
|---|---|---|---|---|---|---|---|
| 2-Ethylhexyl 4-methoxycinnamate (EHMC) | Yes | c#2 | Moderately degraded in WWTP, fast-dissipating in rivers | 4.7 – 505 ng/L[42] | 30 –50 %.[42] | | |
| Octamethylcyclotetrasiloxane (D4) | Yes | C#8 | Degraded in WWTPs; degraded in rivers but not as fast as other chemicals | 0.282 – 6.69 µg/L[43] | 55 - 99.8% | | 90-100%[44,43] |
| Zn | No | c#15 | General assumption for metals | 227-2411 µg/L[23] | 12-90% (up to 64% from primary treatment) [23] | | |
| Vinylchloride | No | | | | | | |
| Se | No | | | | | | |
| Hexachlorobenzene | No | | | | | | |
| Hexachlorobutadiene | No | | | | | | |
| Trichloroethene | No | c#8 | Volatilized in WWTPs; degraded in rivers but not as fast as other chemicals | | | | |

[42] Barbosa et al. (2016)
[43] https://echa.europa.eu/documents/10162/2c2ccd95-f160-c714-164a-80a38ab3975c
[44] Zhang et al. (2016)

*Table 11 – physico-chemical properties and modelled removal efficiency for selected representative chemicals. MW=olecular weight; Kow=octanol-water partition coefficient; S=solubility; VP=vapour pressure; pKa= negative logarithm of the acid constant; Kbio=biodegradation rate in activated sludge systems; Kd=partition coefficient with sludge solids; Rx, Sx (x=1,2,3)=% removal from liquid phase and with sludge at treatment level 1, 2, 3, respectively*

| ***Substance*** | ***MW (g/mol)*** | ***Kow*** | ***S (mg/L)*** | ***VP (Pa)*** | ***pKa*** | ***Kbio (1/h)*** | ***Kd (Lkg$_{MLSS}^{-1}$)*** | ***R1*** | ***S1*** | ***R2*** | ***S2*** | ***R3*** | ***S3*** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *17-Alpha-ethinylestradiol (EE2)* | *296* | *7991* | *4.8* | *2.6E-07* | *10.2* | *3.7E-01*[45] | *337*[46] | *0.0* | *8.8* | *51.3* | *1.7* | *76.4* | *0.1* |
| *17-Beta-estradiol (E2)* | *272* | *8710* | *3.6* | *3.0E-08* | *10.7* | *2.5E+00*[46] | *528*[46] | *0.0* | *12.9* | *77.8* | *0.6* | *84.6* | *0.2* |
| *Estrone (E1)* | *270* | *2512* | *13.0* | *2.7E-08* | *10.8* | *2.0E+00*[47] | *298*[46] | *0.0* | *7.9* | *80.6* | *0.4* | *88.9* | *0.1* |
| *Diclofenac* | *296* | *32359* | *17.8* | *1.1E-03* | *4.2* | *1.2E-01*[48] | *19*[46] | *0.0* | *0.6* | *29.5* | *0.2* | *62.5* | *0.1* |
| *Ibuprofen* | *206* | *9333* | *21.0* | *6.2E-03* | *4.9* | *2.7E-01*[49] | *57*[46] | *0.0* | *1.7* | *50.4* | *0.4* | *78.2* | *0.1* |
| *Carbamazepine* | *236* | *282* | *17.7* | *2.4E-05* | *-* | *1.1E-02*[50] | *15*[46] | *0.0* | *0.4* | *3.8* | *0.2* | *13.6* | *0.2* |
| *Azithromycin* | *749* | *10471* | *2.4* | *1.0E-09* | *8.0* | *1.5E-02*[50] | *310*[46] | *0.0* | *8.2* | *4.6* | *3.4* | *16.1* | *2.8* |
| *Clarithromycin* | *748* | *1445* | *0.8* | *1.0E-09* | *8.7* | *8.3E-02*[50] | *434*[46] | *0.0* | *11.0* | *19.6* | *3.8* | *47.3* | *2.1* |
| *Erythromycin* | *733.937* | *1* | *2000.0* | *1.0E-09* | *8.8* | *1.8E-02*[50] | *178*[46] | *0.0* | *5.0* | *5.8* | *2.1* | *19.5* | *1.6* |
| *Ciprofloxacin* | *331.347* | *3* | *30000.0* | *1.0E-09* | | *1.4E-02*[50] | *20000*[46] | *0.0* | *60.3* | *0.4* | *27.4* | *1.5* | *26.1* |
| *Fluoxetine* | *309.33* | *39811* | *38.4* | *1.0E-09* | *-* | *7.4E-01*[51] | *78*[46] | *0.0* | *2.3* | *20.5* | *0.8* | *50.3* | *0.5* |
| *Sertraline* | *306.23* | *125893* | *2.0* | *1.0E-09* | *9.2* | *8.0E-04*[50] | *4437*[52] | *0.0* | *44.6* | *0.1* | *20.7* | *0.4* | *19.1* |
| *Atorvastatin* | *558.64* | *501187* | *0.0* | *1.0E-09* | *4.6* | *4.5E-02*[53] | *136*[46] | *0.0* | *3.9* | *13.2* | *1.5* | *37.4* | *1.0* |
| *Simvastatin* | *418.566* | *47863* | *0.0* | *1.0E-09* | *-* | *3.9E-02*[54] | *476*[55] | *0.0* | *11.8* | *10.4* | *4.6* | *30.8* | *3.2* |
| *Triclosan* | *289.536* | *218776* | *10.0* | *6.1E-04* | *-* | *3.2E-02*[56] | *3890* | *0.0* | *42.6* | *4.0* | *17.8* | *13.3* | *14.0* |
| *2,6-Ditert-butyl-4-methylphenol (BHT)* | *220.356* | *125893* | *0.6* | *6.9E-03* | *11.6* | *0.0E+00*[57] | *35*[58] | *0.5* | *1.0* | *3.9* | *0.5* | *8.7* | *0.4* |
| *2-Ethylhexyl 4-methoxycinnamate (EHMC)* | *290.4* | *1258925* | *0.5* | *1.5E-06* | *-* | *3.0E-01*[59] | *1230*[60] | *0.0* | *9.5* | *55.9* | *2.0* | *78.4* | *0.6* |
| *4-(1,1',3,3'-tetramethylbutyl)-phenol (OP)* | *206.32* | *13183* | *15.0* | *6.4E-02* | *10.3* | *4.9E-01*[46] | *1035*[46] | *0.0* | *21.3* | *47.7* | *3.7* | *67.8* | *1.2* |
| *4-Nonylphenol* | *220.356* | *30200* | *5.7* | *2.0E-01* | *10.0* | *8.0E-01*[46] | *11093*[46] | *0.2* | *55.8* | *23.5* | *12.2* | *36.3* | *4.4* |
| *PFOS* | *500.126* | | *--* | *2.7E-01* | *1.0* | *0.0E+00* | *0*[61] | *0.0* | *0.0* | *0.0* | *0.0* | *0.0* | *0.0* |
| *Cd* | *112* | *89125* | *--* | *1.0E-09* | *-* | *0.0E+00* | *39800* | *0.0* | *20.2* | *0.0* | *7.9* | *0.0* | *7.9* |
| *Cr* | *52* | *125893* | *--* | *1.0E-09* | *-* | *0.0E+00* | *25120*[46] | *0.0* | *54.0* | *0.0* | *6.0* | *0.0* | *6.0* |
| *Hg* | *200* | *199526* | *--* | *1.0E-09* | *-* | *0.0E+00* | *100000* | *0.0* | *44.5* | *0.0* | *20.5* | *0.0* | *20.5* |
| *Ni* | *58.69* | *22387* | *--* | *1.0E-09* | *-* | *0.0E+00* | *3940*[46] | *0.0* | *22.0* | *0.0* | *0.0* | *0.0* | *0.0* |
| *Pb* | *207.2* | *501187* | *--* | *1.0E-09* | *-* | *0.0E+00* | *14996* | *0.0* | *47.6* | *0.0* | *6.7* | *0.0* | *6.7* |

[45] EQS Dossier; Pomies et al 2013
[46] Pomies et al., 2013
[47] Li et al 2008
[48] geomean from Verlicchi et al 2012 and Lautz et al., 2017
[49] Lautz et al., 2017; Pomies et al., 2013
[50] Lautz et al., 2017
[51] computed from Kbio
[52] Styrishave et al 2011
[53] Geometric mean of Lautz et al., 2017 and Ottmar et al 2012
[54] Sulaiman et al., 2015 https://doi.org/10.1080/09593330.2015.1058422
[55] Ottmar et al 2012
[56] Lautz et al., 2017; Baena Nogueras et al 2017
[57] Wang and Kannan, 2018, report that BHT is degraded but metabolites are recalcitrant
[58] Wang and Kannan report 1% BHT in solids; Kd=50 yields approximately this figure
[59] readily biodeg but >10 days: Straub, 2002
[60] KOC in Straub, 2002
[61] we assume sludge to be already in equilibrium

| *Substance* | *MW (g/mol)* | *Kow* | *S (mg/L)* | *VP (Pa)* | *pKa* | *Kbio (1/h)* | *Kd ($Lkg_{MLSS}^{-1}$)* | *R1* | *S1* | *R2* | *S2* | *R3* | *S3* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Benzene* | *78.11* | *135* | *1790.0* | *1.2E+04* | *-* | *1.0E+00*[62] | *0* | *4.5* | *0.0* | *88.8* | *0.0* | *92.3* | *0.0* |
| *Bisphenol A* | *228.291* | *2089* | *300.0* | *4.0E-08* | *9.6* | *1.0E+00*[63] | *292*[46] | *0.0* | *7.8* | *78.0* | *0.5* | *88.2* | *0.1* |
| *Octamethylcyclotetrasiloxane (D4)* | *296.62* | *3090295* | *0.1* | *1.4E+02* | *-* | *0.0E+00*[64] | *0* | *2.4* | *16.6* | *75.6* | *1.4* | *75.6* | *1.4* |

Metachemicals do not represent real, but "virtual" chemicals, each standing for a whole class of substances featuring a similar behaviour in WWTPs and rivers. Therefore, from the modelling of one specific metachemical, we expect to learn about all the chemicals it corresponds to. Based on the expected behaviour of the chemicals considered in this study, we propose a correspondence with metachemicals as shown in Table 10 above.

Metachemicals can be modelled with the same model used for BOD and coliforms (see Equation 2) if we specify a value of $k_w$ and an emission factor $E_0$. The latter is constantly set to 1 ($s^{-1}$), and loads are consequently computed in terms of PE. Concentrations are obtained from loads divided by river flow, hence they are in units of PE $m^{-3}$, which does not have a physical meaning in itself. To compute the decay of a metachemical along the stream network, we set $k_w = \frac{ln2}{DT50}$, where DT50 is the assumed dissipation half-life of the metachemical in rivers.

Finally, the treatment efficiency in Equation 2 must reflect, for the selected metachemical, the behavior expected in the WWTP.

[62] https://echa.europa.eu/documents/10162/be2a96a7-40f6-40d7-81e5-b8c3f948efc2
[63] https://echa.europa.eu/documents/10162/d1d9e186-4385-4595-b6cb-5a1a7a160f07
[64] Capela et al., 2017

*Table12– Example half life reported from measurements in water, and parameters adopted for river waters in modelling studies, for selected substances, highlighting sometimes very large differences.*

| Compound | Reported half life in water based on measurements[65] | Dissipation Process the half life is referred to [65] | Overall modelled river half life (days) |
|---|---|---|---|
| 4-(1,1',3,3'-tetramethylbutyl)-phenol (OP) | 13.9 h | photodegradation[66] | 20.3[81] |
| 4-Nonylphenol | 12 d | degradation[67] | 18.0[81] |
| EHMC | 360 h | degradation[68] | 1.0[69] |
| Benzene | 5 h | evaporation[70] | |
| | 2.7 h | volatilization[70] | 10.0[71] |
| | 33-384 h | aerobic biodegradation[70] | |
| | 28-720 h | anaerobic biodegradation[70] | |
| | 17 d to 36.6 y | photolysis[70] | |
| | 16 d | biodegradation[25] | |
| E2 | 2-3 d | degradation[72] | 6.0[73] |
| Azithromycin | <20 h | photolysis[74] | 0.1[75] |
| Clarithromycin | 76.5 h | photolysis[76] | 0.2[75] |
| Erythromycin | 365 d | degradation [77] | 0.2[78] |
| Atorvastatin | 433.1 d | hydrolysis [79] | 1.1[89] |
| Simvastatin | 12.9 d | hydrolysis [79] | 7.8[84] |
| Diclofenac | 2.14 d | degradation[80] | 3.3[81] |
| E1 | 2-3 d | degradation [72] | 3.1[81] |
| EE2 | 4-6 d | degradation [72] | 17.0[82] |
| Fluoxetine | 122 d | photolysis[83] | 112.5[84] |
| Ibuprofen | 19.7 d | degradation [80] | 4.7[81] |
| Octamethylcyclotetrasiloxane (D4) | 2.9-6 d | hydrolysis[85] | |
| | 49 –588 d | biodegradation [85] | |
| Per- and Polyfluoroalkyl Substances (using PFOS) | >41 y | Hydrolysis | 1000.0[86] |

[65] Source: compilation provided by Wood Plc.
[66] http://www.inchem.org/documents/sids/sids/140669.pdf
[67] http://www.oecd.org/chemicalsafety/testing/2460911.pdf
[68] Gackowska et al. (2018)
[69] Straub, 2002
[70] https://www2.gov.bc.ca/assets/gov/environment/air-land-water/water/waterquality/wqgs-wqos/approved-wqgs/benzene/benzene_overview.pdf
[71] Pistocchi et al., 2018
[72] Ying et al.,2002 within Adeel et al. (2016)
[73] Liu et al., 2011
[74] Tong et al. (2011)
[75] Hanamoto et al., 2018
[76] Nakagawa (1992)
[77] Jessick (2010)
[78] Acuña et al. 2015
[79] Sulaiman, S. et al., (2015)
[80] Araujo et al. (2014)
[81] Pistocchi et al., 2012
[82] EQS dossier
[83] USEPA(2007)
[84] Boxall et al., 2014
[85] RIVM (2012)
[86] assumed persistent: see Pistocchi and Loos, 2009

| | | | |
|---|---|---|---|
| | >3.7 y | Photolysis | |
| Sertraline | 23d | photolysis[87] | 112.5[88] |
| Carbamazepine | | | 127.4[81] |
| Ciprofloxacin | | | 0.3[89] |
| Triclosan | | | 3.9[90] |
| 2,6-Ditert-butyl-4-methylphenol (BHT) | | | 1000.0[91] |
| Bisphenol A | | | 3.4[81] |
| Cd | | | 1000.0[71] |

*Table 13 – Metachemicals proposed for the analysis. Virtual substances c#1, c#13 and c#14 are not supposed to be of practical interest.*

| Fate in WWTP | Environmental fate | | |
|---|---|---|---|
| | Fast dissipation (DT50~1 day) | Intermediate dissipation (DT50~10 days) | Slow dissipation (DT50~100 days) |
| Slowly removed | [c#1] | c#6 | c#11 |
| Moderately removed | c#2 | c#7 | c#12 |
| Removed | c#3 | c#8 | [c#13] |
| Sorbed and slowly removed | c#4 | c#9 | [c#14] |
| Sorbed | c#5 | c#10 | c#15 |

The assumed removal efficiencies for the 15 metachemicals correspond to the categorization of their fate in WWTPs (see Figure 2), while for the DT50 we distinguish only orders of magnitude, i.e. 1, 10 and 100 days. The removal efficiency in more stringent treatment is estimated from that of secondary treatment, considering that the removal efficiency in the activated sludge process with F/M equal to 0.05 and 0.2 kg O2 $kg_{dw}^{-1}d^{-1}$ indicates a fairly simple relationship (Figure 3), well represented by the equation:

$$Removal_{0.05} = 10\sqrt{Removal_{0.2}} \qquad \text{Equation 7}$$

$Removal_x$ being the removal efficiency (%) at F/M=x (x=0.2, x=0.05). This interpolating equation is plotted together with the data in Figure 3. We conventionally assume secondary treatment to have a F/M ratio of 0.2, and more stringent treatment F/M=0.05, with the caveats already expressed about this assumption.

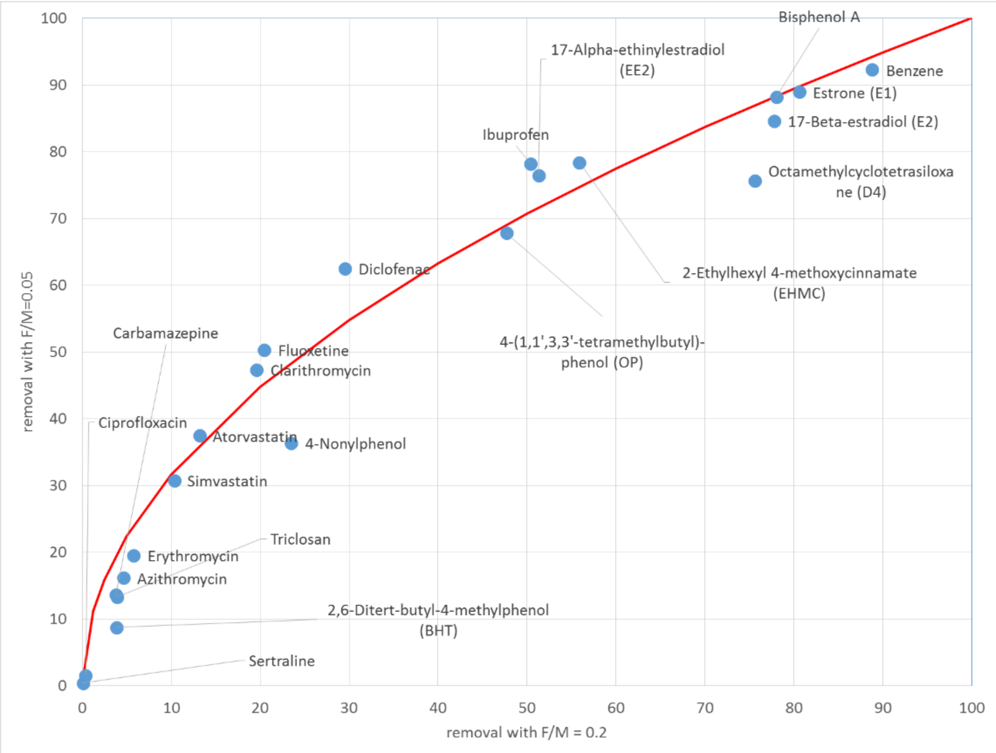
Based on these considerations, we can apply the same model used for coliforms (Equation 2) to any chemical in order to quantify the effect of the UWWTD on the reduction of pollution loads in Europe, assuming chemicals are used uniformly across the EU and, consequently, the emission per PE is a constant in space. If we knew the emission rate of a chemical (e.g. in ng $s^{-1}$ $PE^{-1}$), we could immediately convert the corresponding metachemical loads and concentrations into the chemical's ones (e.g.in ng $m^{-3}$). It may be possible to estimate emission rates

[87] USEPA(2007)
[88] based on analogy with Fluoxetine, Styrishave et al., 2011
[89] Used STP value as a proxy, see Castiglioni et al., 2004 https://doi.org/10.1016/j.yrtph.2003.10.002
[90] Koumaki et al 2018 (average)
[91] see Wang and Kannan, 2018

for a few of the selected chemicals through specific studies, but this would require an effort beyond the scope of this work. However, even in the lack of information on emission rates, the model can still be used to compare different scenarios of wastewater treatment, and quantify the corresponding expected reduction of loads and concentrations.

As we neglect all other (diffuse, industrial etc.) sources of emissions, the model is to be regarded as merely indicative of pollution associated with urban wastewater, and should be used with care when studying chemicals known to have significant non-urban sources.

*Figure 3 – Relationship between the modelled removal efficiency (%) of activated sludge with F/M= 0.2 and F/M = 0.05 kg O2 $kg_{dw}^{-1}d^{-1}$ for selected chemicals. The red line represents the proposed interpolating equation.*




### 2.2.4 Modelling combined sewer overflows

The models described above allow a representation of pollution under "dry weather" conditions, assuming that wastewater is collected and treated to different extents depending on the scenarios. Although in some cases (nutrients and BOD) we account for diffuse sources of pollution (agricultural fertilizers and urban runoff) mobilized following rain events, we do not account for the intermittent and occasional discharge of wastewater occurring in combined sewer networks, when storm discharges exceed the capacity of the network or the WWTP. In this section, we illustrate the model we use to quantify these combined sewer overflows (CSO).

For each urban area, as a first and rough approximation, we imagine the whole combined drainage network to have a single outlet, and we model a single "equivalent" CSO downstream of the latter. We quantify the aggregated CSO water discharge for the urban area with the equation (Pistocchi and Dorati, 2018):

$$Q_{CSO} = max(0,\ fA_U R + (1-d)\, p\, P) \qquad \textit{Equation 8}$$

where $A_U$ is the surface area served by combined sewer networks, R is rainfall, *p* the wastewater discharge per capita in the area, P the total population, f a runoff coefficient (% of rainfall on urban area AU which turns into runoff), and d the combined sewers' design dilution rate, i.e. a multiple of wastewater discharge (usually in the range 2÷10) above which CSO begins. If we assume that urban runoff is initially free from pollution, the CSO mass discharge of a given pollutant can be computed as:

$$L_{CSO} = C\frac{max(0,\ fA_U R\ + (1-d)\ p\ P)}{1+\frac{fA_U R}{p\ P}}$$ *Equation 9*

where C is the concentration of the pollutant in wastewater. In the setup of the model, AU, k and P can be regarded as known input parameters and R is an external forcing, while two parameters remain to be calibrated, one representing land management (f), and the other representing the sewers'safety margin before CSO (d). If we set C=1, Equation 9 is numerically equal to the volumetric discharge of untreated wastewater corresponding to the CSO, i.e. its „untreated wastewater equivalent" or "CSO wastewater content". It should be stressed that this model aims only at the estimation of the wastewater load that is not treated due to overflows. It is well known that urban runoff itself can be a significant carrier of contamination due to the wash-off of polluted surfaces and in-pipe sediments from the sewage networks. Therefore the CSO wastewater content should not be interpreted as an estimation of the overall pollution caused by urban runoff, but only of the "subtraction" of wastewater from the treatment system, caused by the presence of combined sewers. The above model is an acceptable approximation for areas where runoff and wastewater collection in the combined sewer network are uniform within an urban area, and the less realistic, the higher the variability of f, d and P within it.

We apply the above equations to the ~ 700 Functional Urban Areas (FUA) identified across Europe (EEA, 2012). We calculate a time series of precipitation from the three-hourly Multi-Source Weighted-Ensemble Precipitation (MSWEP) version 2.0 global dataset (Beck at al., 2017), from which grid cell values for each 3-hour step are averaged across each FUA. Population is estimated based on GEOSTAT 1km2 population-grid of 2011 (EC, 2011), while *p* is set to 160 l/(inhab.day). Moreover, we assume f=1 for impervious areas and f=0 for non-imprevious areas within each FUA. The degree of imperviousness is estimated for each FUA based on Functional Urban Areas indicators from the LUISA modelling platform (Lavalle et al. 2014). We finally divide for each FUA the load (Equation 9) by the impervious area, yielding a load per unit impervious area served by combined sewers (unit FUA load).

The weighted average of unit FUA loads and total impervious area in a region is eventually used to upscale loads, also taking into account the percentage of combined sewers in the region. The latter is estimated after Milieu, 2016 (see Annex III). On average, about 30% of urban areas are expected to have separate sewers in Europe. This setup allows estimating the total CSO water and pollutant mass discharge from the FUA, for a given value of d. The values of $L_{CSO}$ (Equation 9) upscaled to NUTS2 regions can be interpreted in terms of untreated PE by dividing volumes by 160 l/PE/day.

This model describes a worst-case scenario because (1) it overestimates the contributing impervious area by setting f=1, and (2) it completely neglects the storage and buffering of flow in the sewer network.

## 2.3 Scenarios considered in the analysis

The analysis considers 7 fundamental scenarios, summarized in Table 14. These are briefly described in the following paragraphs.

Vigiak et al., 2018, describe in detail how domestic waste discharges were assessed. The maps of wastewater discharges (in population equivalents – PE) at different levels of treatment are presented in Annex II for reference.

In addition to these scenarios, we consider the impact of combined sewer overflows (CSO) as a separate calculation, as explained below (§2.4).

*Table 14 – Scenarios analyzed in this study*

| **Scenario** | **Short name** |
|---|---|
| Present situation as per the latest WISE report (2014) | Baseline |
| Situation before entry into force of the Directive (circa 1990) | Pre-directive |
| "What-if-no-Directive" | WIND |

| | |
|---|---|
| Full Implementation of the Directive | Full compliance |
| Full implementation with systematically ineffective individual and appropriate systems (IAS) | IAS |
| Full implementation with effectiveness of IAS depending on management | IAS+ |

### 2.3.1 Baseline

In this scenario, we make use of the data reported by the EU Member States on the agglomerations and respective level of treatment, compliant with the UWWTD. The data were taken from the WaterBase – Wastewater database provided by the European Environment Agency (EEA), version 5[92], containing data of the year 2014. The reported data presented a number of issues which required extensive quality control and gap filling. This is documented in Vigiak et al., 2018. One specific issue with the reported data is that they do not include the population living in agglomerations below 2000 person-equivalents (PE) or in scattered dwellings. Vigiak et al., 2018, estimated this part of the population for all EU member states, by comparison between the resident population and the population resulting from the reported PE, assuming 1.23 PE/resident. Although the number of PE per resident is highly variable as a function of the activities in the area served by a wastewater treatment plant, Vigiak et al., 2018, found this simple relationship to be quite stable for Europe, when the target is not an assessment of a specific plant or a small region. The estimated population living in agglomerations below 2000 PE or scattered dwellings is generally small. However, in some countries this may be a relatively large share of the resident population (up to about 10%, see Vigiak et al., 2018).

The person-equivalents (PE) of the different WWTPs in Europe, in this scenario, are attached to a specific location in space representing the discharge point of each WWTP. The population of smaller settlements and scattered dwellings was uniformly allocated in space.

### 2.3.2 Pre-directive

Ideally, comparing the Baseline with the situation before the Directive would require having data for the year 1990 with a granularity similar to that available today (Vigiak et al., 2018). Unfortunately, this information is not available and the only source of information that we could use for the year 1990 is about population at different levels of treatment, by country, provided by EUROSTAT (see Vigiak et al., 2018, for additional details). This information is reckoned to be only to a limited extent consistent with the information reported by the Member states under the UWWTD, and therefore comparisons must be made with care. In spite of these limitations, EUROSTAT data for the year 1990 yield a possible picture of what the situation could have been before the entry into force of the Directive, hence a benchmark for the progress of wastewater treatment in Europe.

EUROSTAT data, which represent population totals by country undergoing no treatment, primary, secondary or more advanced treatments, were used to estimate a spatial distribution of PE with the same levels of treatment in each EU Member state, on the basis of the spatial distribution of population. Further details can be found in Vigiak et al., 2018.

### 2.3.3 Full compliance

This scenario is obtained by bringing all WWTPs of the Baseline scenario to the level required by the UWWTD. This means assuming all WWTPs have biological (secondary) treatment in place, and those falling in the catchments of sensitive areas have more stringent treatment for the removal of N, P or both (depending on whether the WWTP discharges to a normal or sensitive area). WWTPs which, under the Baseline scenario, are already at the required level of treatment or above, stay the same under this scenario.

### 2.3.4 "What-if-No-Directive" (WIND)

In order to represent a hypothetical condition of wastewater treatment in Europe in the absence of the UWWTD, we assume that the different countries would have implemented some wastewater treatment anyway, and that each country would be in an intermediate situation between the pre-directive and baseline scenario conditions

[92] https://www.eea.europa.eu/data-and-maps/data/waterbase-uwwtd-urban-waste-water-treatment-directive-5

depending on their specific capacity to plan, invest in, and manage wastewater treatment plans. This scenario is quantified in a simplified way as a weighted average of the two scenarios (baseline and pre-directive) as usually done in statistical data fusion. The concentration of a generic pollutant in a given water body under the WIND scenario is estimated as :

$$C_{WIND} = w\, C_{baseline} + (1-w)\, C_{pre-directive} \qquad \text{Equation 10}$$

Where $C_{baseline}$ and $C_{pre-directive}$ are the concentrations in the baseline and pre-directive scenarios respectively, and w is a weight. We have computed BOD, N, P and coliform concentrations and loads as the weighted average between the pre-directive scenario (1990) and the current (2015) scenario, using weights based on an evaluation summarized in Annex IV. We consider two slightly different weighting schemes.

Under the first scheme (called "WIND1") a weight of 0 is applied to BE, EL, PT, MT, CY, BG, IT, IE, ES and RO. A weight of 1 is applied to AT, DE, DK, FI, NL, SE. The other countries have weight=0.5. A weight of 0 means the country would be essentially at the level of 1990; a weight of 1 would mean it to be at te same level of 2015 irrespective of the UWWTD. A weight of 0.5 means it would be at intermediate conditions.

Under the second scheme (called "WIND2") the 13 most recent members of the EU (EU13) are assigned a weight of 0.25, while the other members (EU15) are assigned the same weights as above.

### 2.3.5 Quantifying individual / appropriate systems: IAS and IAS+ scenarios

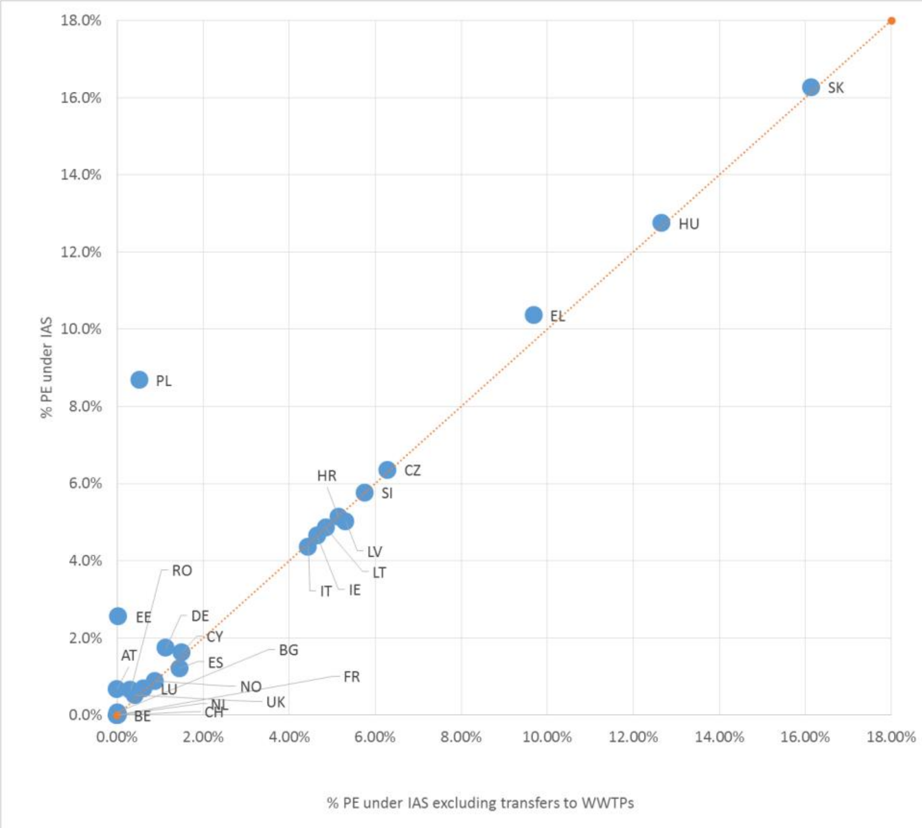
Individual/appropriate systems (IAS) are a solution admitted by the UWWTD for wastewater treatment in certain parts of the agglomerations above 2000 PE, where it can be demonstrated that an extension of collection systems would not be convenient. IAS are expected to have a quantitatively limited impact on concentrations and loads evaluated over the European stream network, due to their small entity compared to the total production of wastewater (see Figure 4). While the UWWTD requires IAS to be effective, there is no specific provision obliging member states to ensure a monitoring of their effluents. Therefore it is possible that IAS show less-than-appropriate performance in treating wastewater, due to inadequate design, maintenance, monitoring and enforcement. In order to quantify the impact of an inadequate functioning of IAS, we compare the full compliance scenario[93], with a modified scenario where IAS are assumed to correspond to a primary treatment, all the rest being as under full compliance. It should be stressed that IAS may be reported by member states as being subject to transfer of wastewater volumes to WWTPs by trucks. In this case, the corresponding PE were not computed under IAS, but rather attributed to the agglomerations' WWTPs. The difference is relevant only for Poland and, to some extent, Estonia (Figure 4).

Under the IAS+ scenario, we have used an evaluation summarized in Annex IV to compute a weighted average of scenarios (1) with IAS equal to a primary treatment and (2) with IAS equal to the WWT level required by the agglomeration (either secondary or tertiary), in a way similar to the WIND scenario (Equation 10). In other words, we assume IAS cannot deliver less than primary treatment, and cannot exceed the efficiency of centralized plants, although both assumptions are not necessarily true in all cases (for instance, certain IAS might have an efficiency of treatment higher than the WWTP of the agglomeration).

The weights to compute the average were chosen as follows: AT, CY, DE, EE, UK have IAS equivalent to the agglomeration's WWTP, hence weight is 1; CZ is midway (50% of IAS are assumed to be equivalent to the agglomeration's WWTP and 50% equivalent to primary treatment), hence weight is 0.5; the other countries are assumed to have IAS equivalent to primary, hence weight is 0.

*Figure 4 – IAS as a % of total PE in different EU member states, considering vs not considering the transfers of wastewater volumes from IAS to WWTPs using trucks.*

[93] Under the full compliance scenario, we assume IAS to have the same level of performance of the treatment level (biological or more stringent) required for the agglomeration they belong to.




## 2.4 The impact of combined sewer overflows (CSO)

Combined sewer overflows are quantified and evaluated in this study as an additional potential source of pollution on top of "dry weather" discharges. For the sake of a first exploration, we consider a reference condition where combined sewer overflows are just allowed to occur without particular management measures. We represent the reference condition through a constant dilution rate for Europe of d=4 in Equation 8 and Equation 9. In some countries, measures may be in place to reduce overflow volumes and/or frequencies. In particular, provisions may exist about a minimum dilution rate d>4. In addition, some countries may prescribe detention measures to reduce overflows, such as detention tanks or surplus storage volume in the drainage network, to limit overflows to a maximum number of events or days per year. An overview of existing measures in EU Member States is provided in Table 15. The higher a dilution rate d>4, the lower the resulting CSO loads. Moreover, comparing the cumulative CSO load (Equation 9) over the n most severe events per year instead of all events yields an estimation of the reduction of load corresponding to measures enabling to ensure a maximum of n events/year. Based on the CSO model run as described in Annex III, we have derived a relationship between the equivalent wastewater volume discharged, and the number of events considered (n) or the dilution rate (d), enabling the calculation of CSO loads as a percentage of loads with d=4 and no detention, as a function of d and n (Table 15). We compute CSO loads for the above countries assuming the most favorable condition, obtaining the correction factors or reduction coefficients reported in Table 15.

*Table 15 – correction factors to apply to default CSO loads taking into account the provisions on minimum dilution rate (d) and number of days of CSO allowed in different EU Member States. Source: compilation by Wood Plc (see Annex IV)*

| *country* | *correction factor* | *d* | *# days* |
|---|---|---|---|
| AT | 44% | 8 | |
| BE | 50% | 7 | 18.71[94] |
| DE | 47% | 4 | 17.32[95] |
| DK | 32% | 4 | 10 |

[94] This is the geometric mean of reported values for Brussels, between 7 and 50 per year.
[95] This is the geometric mean of reported values for Land Bavaria, between 15 and 20 per year.

| *country* | *correction factor* | *d* | *# days* |
|---|---|---|---|
| ES | 44% | 8[96] | |
| FI | 100% | 4 | |
| FR | 52% | 4 | 20 |
| EL | 52% | 7 | |
| IE | 78% | 5 | |
| IT | 62% | 6 | |
| LU | 100% | 4 | |
| NL | 29% | 4 | 10 |
| PT | 62% | 6 | |
| SE | 100% | 4 | |
| UK | 100% | 4 | |
| CY | 100% | 4 | |
| CZ | 44% | 8[96] | |
| EE | 100% | 4 | |
| HU | 100% | 4 | |
| LT | 100% | 4 | |
| LV | 100% | 4 | |
| MT | 100% | 4 | |
| PL | 67% | 4 | 30 |
| SI | 100% | 4 | |
| SK | 100% | 4 | |
| BG | 78% | 5 | |
| RO | 100% | 4 | |
| HR | 100% | 4 | |

Finally, DK prescribes 5 mm of rainfall storage, and NL 9 mm (7 in the network + 2 at the head of the WWTP). In order to account for this, we implemented a variant to Equation 8 and Equation 9, where we replaced rainfall with « net rainfall ». The latter is computed as rainfall exceeding a storage of an amount of S mm during each event. A rainfall event is defined as a series of time steps with non-null precipitation, separated by another series of non-null precipitation by at least m time steps (in our setup, time steps are 3-hourly). For the case of NL, we set S=9 and m=3 ; for the case of DK, S=5 and m=2. At the start of the event, rainfall is considered to be retained until its cumulate reaches storage S; after this, all rainfall contributes to sewer flow.

It should be stressed that our analysis addresses only the potential load from CSO, and neglects the positive impact that may exist when some treatment of CSO is foreseen. CSO treatment (typically a primary/mechanical stage) is required at least in in AT, DE, BE, SK and RO. The efficiency of removal of contaminants with this treatment is not known, though, making it impossible to take into account this aspect in our study.

More in general, our worst-case model calculation is likely to overestimate CSO by a large extent in all cases where the connected impervious surface is significantly different from the total impervious area in a FUA, and where there is significant network storage capacity.

[96] For Spain and the Czech Republic, d is prescribed between 5 and 8, the latter assumed to apply as the most favorable case for these countries.

## 2.5 Considerations on key uncertainties in the assessment, sensitivity analysis and model verification

### 2.5.1 Uncertainty in the quantification of emission scenarios

In the assessment presented here, we compare different scenarios of wastewater treatment in Europe. All these scenarios are affected by uncertainty in the estimation of the population equivalents (PE) subject to different levels of treatment in each European river basin.

The scenario corresponding to current conditions is based on the data for agglomerations above 2000 PE reported by the EU Member States under the UWWTD. While generally this represents the most reliable source of information, issues with the quality of reported data have been highlighted in the past[97]. The uncertainty in this case seems relatively small but difficult to quantify. The same uncertainty necessarily affects also the scenario of full implementation, which is built from a comparison of current treatment levels with those theoretically required by the UWWTD.

Much more uncertain is the quantification of treatment levels under the pre-directive scenario. In this case, no data exist reported by the Member States and the only available information is the one provided by EUROSTAT at the national level. This information has been spatialized with the approach presented in Vigiak et al., 2018, showing that, for the reference year 2015, the national totals are reasonably consistent with the data reported under the UWWTD, although with some discrepancies (see Vigiak et al., 2018). However, when considering the regional level, there is no guarantee of comparability between the two approaches, as shown in Annex II.

This uncertainty in the estimated pollutant emissions in each river basin reflects in the uncertainty in the estimation of concentrations and loads in the stream network and to coastal areas. In the absence of a real benchmark, it is not possible to quantify this uncertainty. However, it is expected that the models cannot be used beyond the target of capturing broad continental trends, statistical distributions of concentrations and hot spots represented by high population with relatively low levels of treatment, and in the presence of low dilution capacity of the stream network.

### 2.5.2 Calibration and verification of models

Models used for organic matter and nutrients were calibrated on the basis of available concentration and load data, as discussed in Vigiak et al., 2019, and Grizzetti et al., 2012. For BOD, Vigiak et al., 2019 show that the model is sensitive to parameters representing the efficiency of secondary treatment and event-mean concentration for urban runoff. The predicted loads are well correlated with observations and with errors within one order of magnitude, while loads per unit catchment area and concentrations show lesser correlation with observations. Model performances for nutrients are similar (Grizzetti et al., 2012). In this case, the model parameters specifically calibrated were those related to the retention of nutrients in the stream network.

For coliforms, the model is not calibrated. However, the emission and die-off parameters for coliforms are the same proposed in Reder et al., 2015. These authors develop a model taking into account livestock and urban runoff as sources, in addition to domestic wastewater, and show that predictions match observation reasonably well within one order of magnitude. As the model used here neglects sources other than wastewater, we may expect a general underestimation of concentrations and loads. At the same time, Reder et al., 2015, increase the rate of decay of coliforms in rivers, that we estimate according to Equation 3, by a term reflecting disinfection by solar radiation and a term reflecting coliform settling. While an in-depth comparison was not conducted in the scope of this work, we expect the tendency of Reder et al., 2015 to estimate faster coliform decay to compensate their higher loadings, thus making our results reasonably consistent with theirs.

### 2.5.3 Representation of IAS and CSO

The approach to quantifying the maximum expected impact of IAS, as presented above, was discussed with experts from various EU Member States. For certain countries there was sufficient evidence to believe that IAS management is effective, and IAS are by all means equivalent to WWTPs, hence fully compliant. For other countries, such evidence was not available. However, the results presented in the next sections need to be interpreted as estimations of potential impacts, virtually absent whenever IAS management is effective.

The modelling approach used for CSO was also discussed with experts from various EU Member States. In the scope of this study, it was not possible to collect sufficient evidence to confirm our estimates, which must be

[97] See http://ec.europa.eu/environment/water/water-urbanwaste/implementation/implementationreports_en.htm

regarded as conservative (i.e. overestimating loads of CSO, by underestimating the buffering capacity of combined sewer networks and overestimating the extent of impervious areas connected to combined sewers). An extensive verification of the model has been initiated and is expected to yield a more accurate picture of CSO in Europe in the near future.

### 2.5.4 Metachemicals

The modelling of chemical fate in WWTPs is extremely complex and uncertain. The approach adopted here aims at exploring the universe of chemical substances entering a WWTP by referring to a limited set of representative molecules. For these molecules, the behavior in WWTPs has been represented in a very schematic way, with constant parameters applied in the whole EU in spite of the high variability expected from the literature. While the modelling chemical fate remains highly uncertain, the proposed approach is merely intended to illustrate the possible impact of wastewater treatment on chemical pollution.

# 3 Water quality in Europe under the different scenarios[98]

## 3.1 Coliforms

When we consider the total length of the European stream network with predicted coliform concentration below the thresholds for bathing water quality (Figure 5), the implementation of the UWWTD is estimated to have almost doubled the river length with at least good quality, from 43.1% in the pre-directive scenario to 80.3% in the Baseline scenario. Under a full compliance, the percentage would further rise to 82.9%. Implementation of the UWWTD is also estimated to have increased the length of coastlines with at least good quality, from 72.5% in the pre-directive scenario to 93.1% in the Baseline scenario. Under a full compliance, the percentage would further rise to 95.2% (Figure 6).

Figure 7 and Figure 8 show the percentage of length of rivers where concentration of faecal coliforms meets the standards of good or excellent quality for bathing. The comparison highlights that, in many regions, there is a clear improvement of water quality when the UWWTD is fully implemented.

Figure 9 portrays the length of the coastline below the thresholds of good or excellent bathing water quantity under the pre-directive, Baseline and full compliance scenarios. Under the WIND scenario, countries with a lesser development of wastewater treatment would have a significantly higher percentage of the length of the stream network not meeting bathing waters standards (Figure 10). At European scale, this translates into an extra 25% of the stream network, approximately, in less than good conditions if there were no Directive (Figure 5 and Figure 6).

It is worth recalling that our assessment neglects contributions from other sources, such as urban runoff and livestock, thus possibly underestimating coliform concentration in many river stretches. The analysis is thus to be regarded in comparative, and not absolute terms.

Another aspect worth mentioning is the importance of CSO on the conditions of bathing waters, which does not appear in this calculation. Harder-Lauridsen et al., 2013, report evidence of impacts on the health of swimmers in CSO-contaminated seawater compared to cleaner waters in Denmark. During storm events, CSO may cause short-term impacts due to high concentrations of coliforms. For instance, in the coastal urban area of Rimini, Italy, otherwise classified in excellent conditions for coliforms, CSO may occur on average 10 times during the bathing season, triggering a “no bathing” order during the day following the overflow, with an estimated direct impact of 1.5 million Euro without considering indirect (e.g. image) damages to tourism (Venier, 2018). This has justified a massive investment locally (worth 155 million Euro) to retrofit the sewer network and treatment systems (*ibid.*).

[98] The percentages of the stream network or coastline meeting certain environmental standards (including good ecological status) as presented in this study are intended for illustration of the results of modelled scenarios, and should be used for scenario comparison only. By no mean they can be interpreted as a judgment on the degree to which specific legal objectives are met in the different regions of Europe.

*Figure 5 – conditions of the European stream network with respect to bathing water standards for E.Coli under three scenarios*

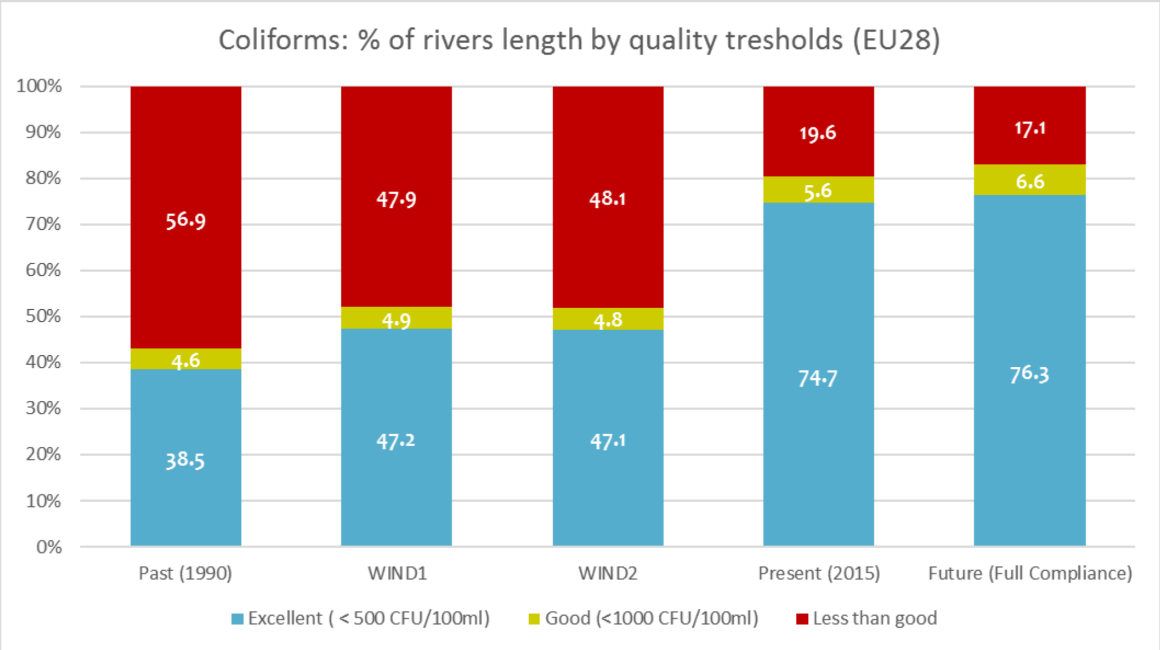



*Figure 6– conditions of the European coastlines with respect to bathing water standards for E.Coli under three scenarios*

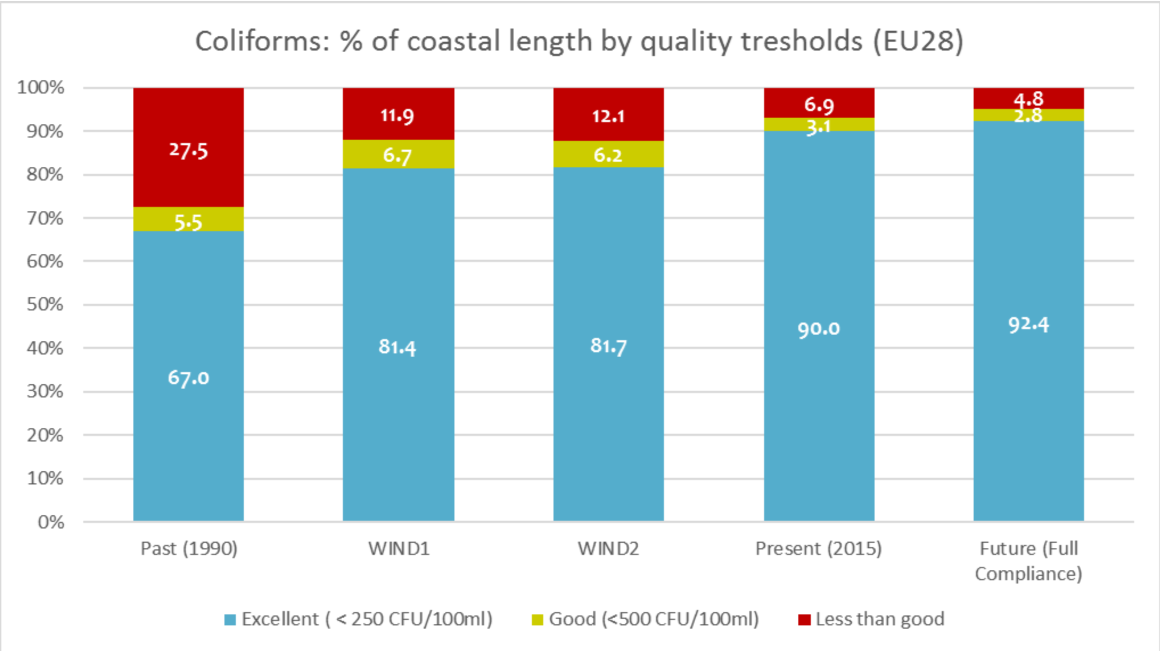
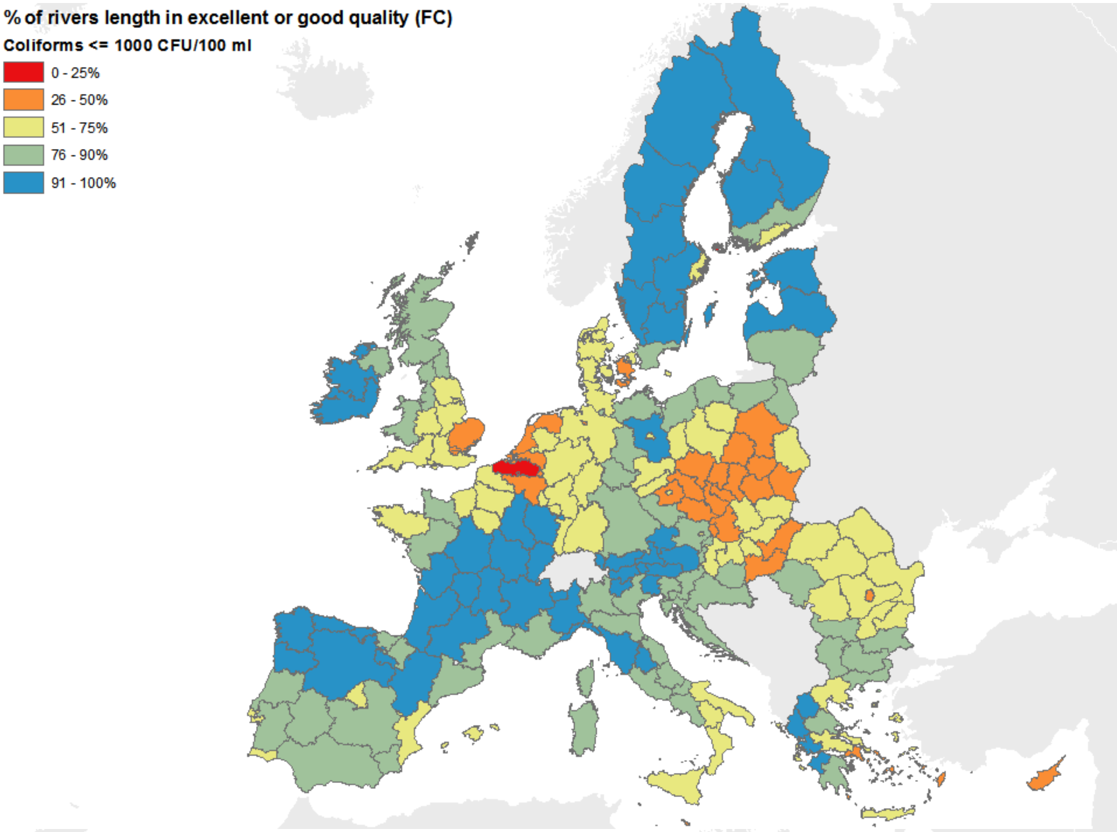


*Figure 7- Percent of the stream network with faecal coliforms below the good quality threshold for E.Coli in bathing waters, aggregated by NUTS2 regions (except for DE, UK, NL, BE, where NUTS1 regions are used for readability) – full compliance (FC) or Baseline from member states' reports (2015).*



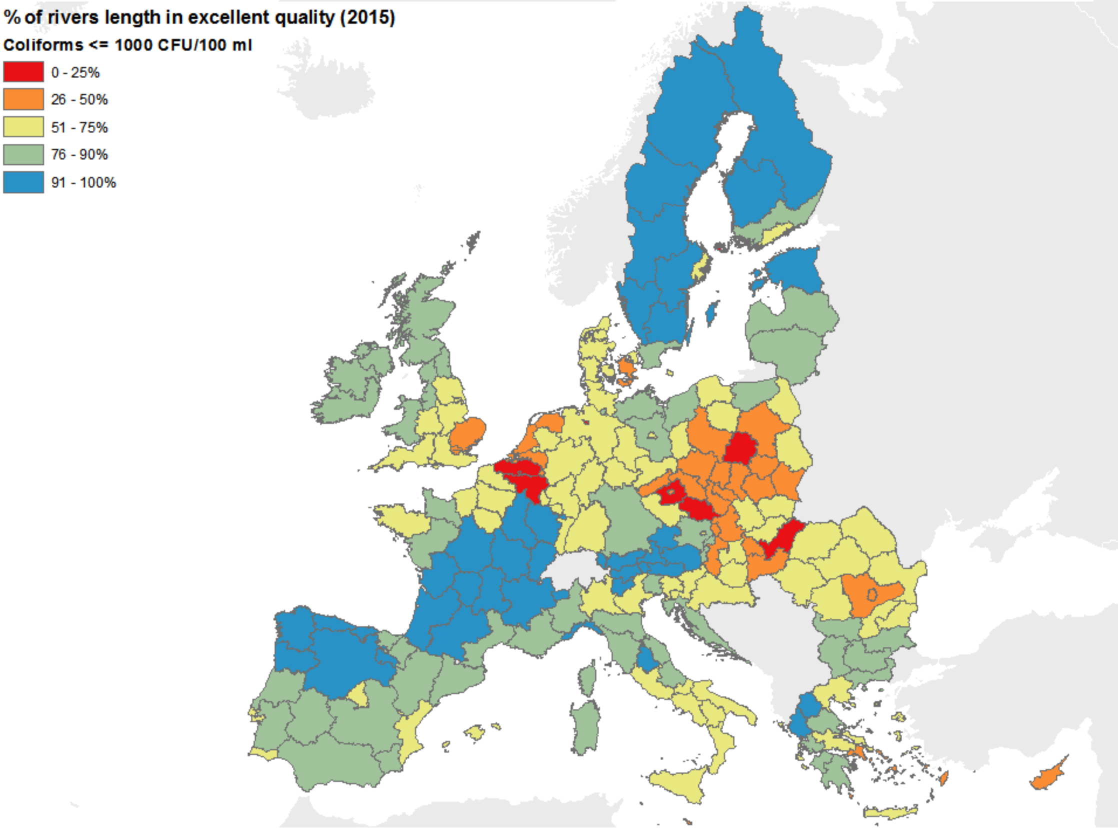
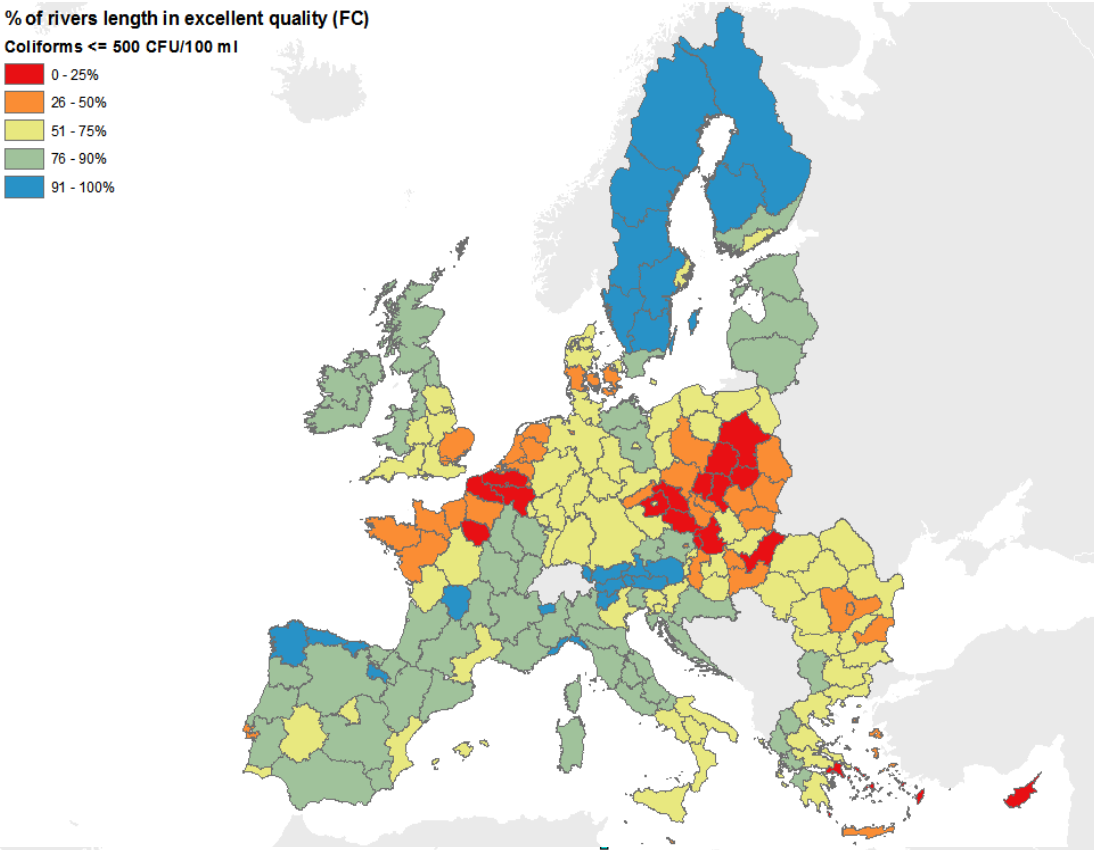



*Figure 8 - Percent of the stream network with faecal coliforms below the <u>excellent</u> quality threshold for E.Coli in bathing waters, aggregated by NUTS2 regions (except for DE, UK, NL, BE, where NUTS1 regions are used for readability) – full compliance (FC) or Baseline from member states' reports (2015).*



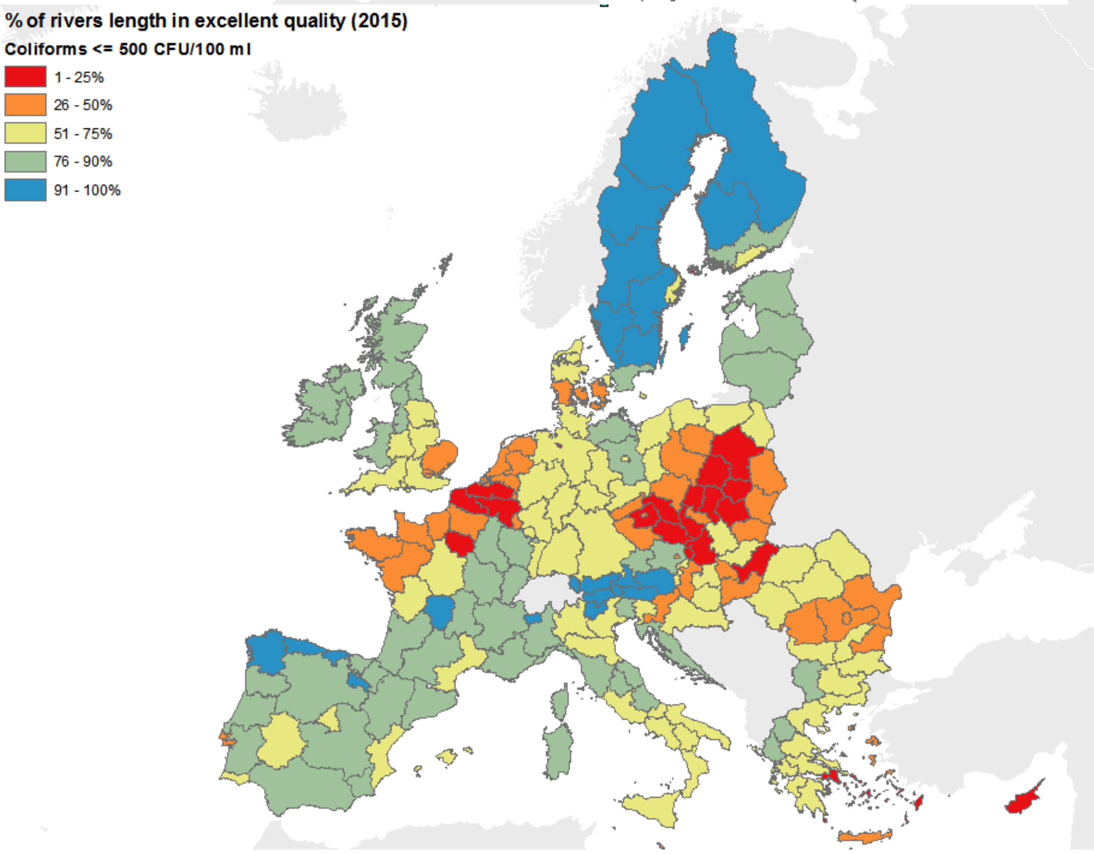



*Figure 9 - Percent of the coastline with faecal coliforms below the good quality threshold for E.Coli in bathing waters, aggregated by NUTS2 regions (except for DE, UK, NL, BE, where NUTS1 regions are used for readability) – full compliance (FC), circa 2015 and circa 1990.*

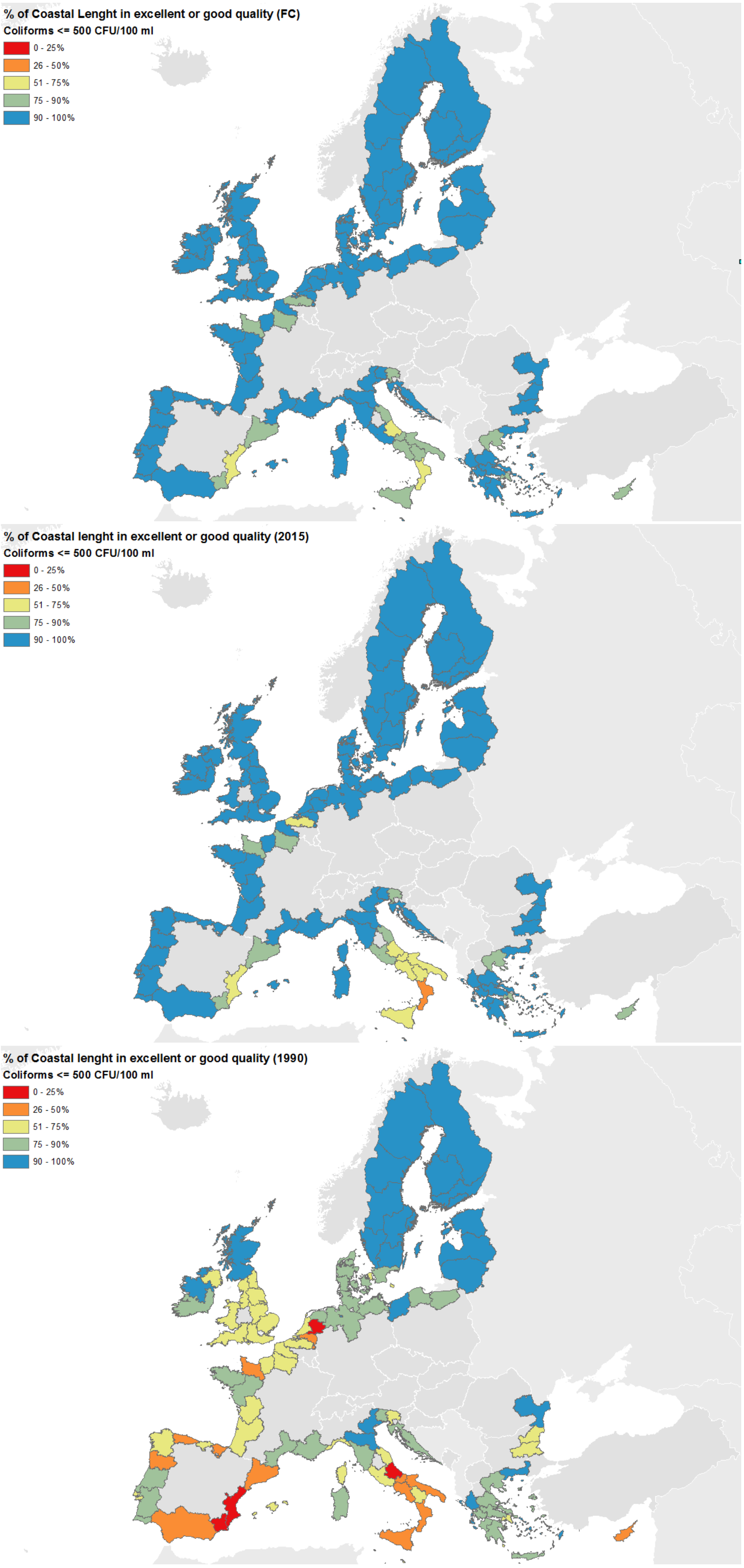
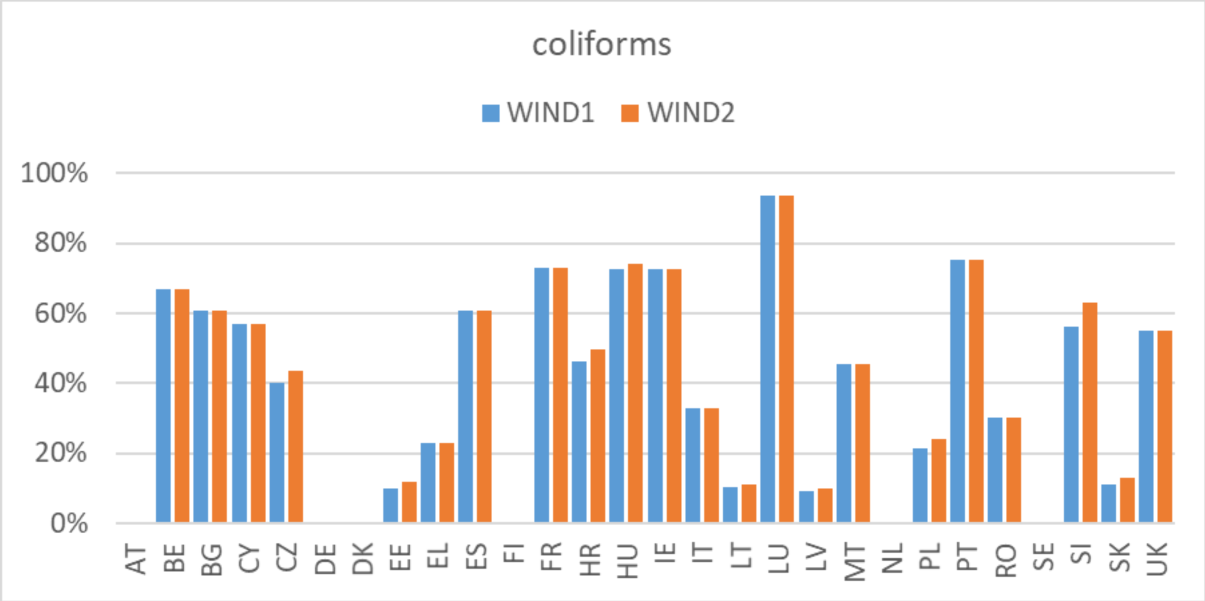
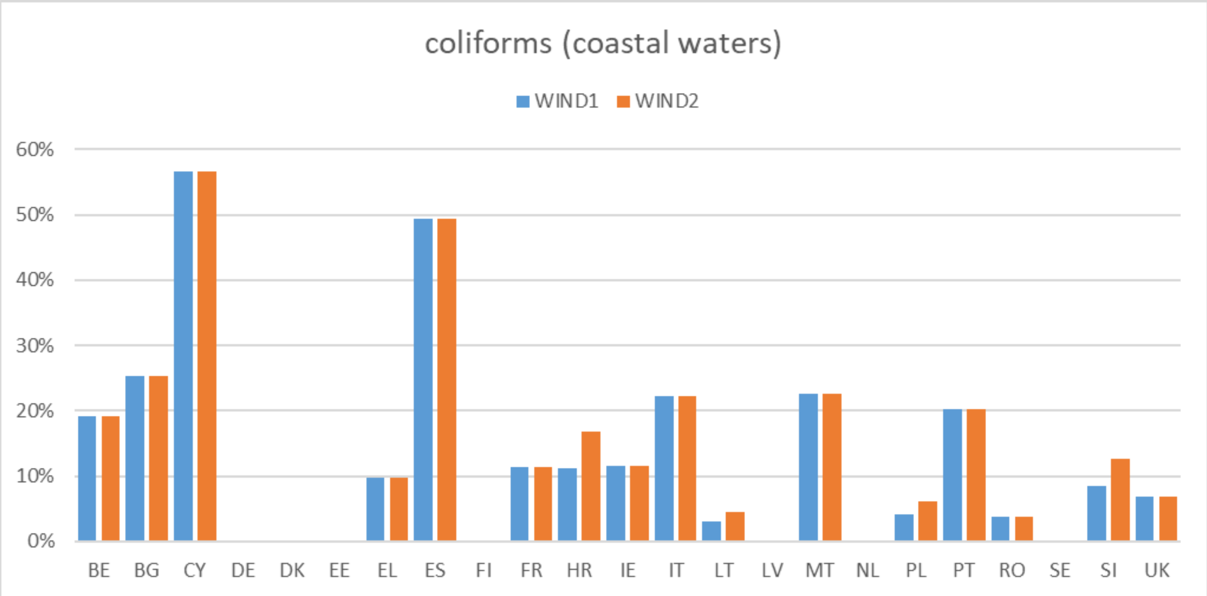


*Figure 10 – percent change of river and coastal length, by country, not meeting bathing waters standards for coliforms under WIND scenario compared to baseline (present) scenario.*



## 3.2 $BOD_5$ and Nutrients

The analysis shows that, for $BOD_5$, the impact of the UWWTD has been significant in terms of the reduction of loads of urban wastewater to the stream network (Figure 11).

In terms of concentrations in rivers, the improvement is sizable but small for $BOD_5$, with about 4% of the stream network moving from above to below the threshold for good ecological status (Figure 12). Under full compliance, another 1.6% of the stream network may similarly improve. A similar impact can be observed by looking at the loads of $BOD_5$ to the European sea regions. This is due to the fact that urban wastewater is not the only source of $BOD_5$.

Under the WIND scenario, certain countries would see an increase in the length of the stream network not meeting good ecological status standards, up to 10% compared to the current scenario (Figure 13). At European scale, this translates into a modest increase in the length of the stream network above good status threshold (from 13.9 to 14.5%, Figure 12).

Similar considerations apply for nutrients, showing a very marked reduction of loads from wastewater to the stream network moving from pre-directive to baseline and full compliance scenarios (Figure 16). Also for total N the impact of the UWWTD has been sizable but small, enabling about 4% of the stream network to improve from above to below the threshold for good ecological status (Figure 17). Under full compliance, another 0.7% of the stream network may similarly improve. Improvements appear smaller when considering higher concentration thresholds, such as those of the DWD (Figure 19).

The improvement is more apparent in the case of total P, where about 10% of the stream network improves from above to below good status threshold. In this case, the additional improvement through full compliance is about 1% of the stream network (Figure 17).

Under the WIND scenario, certain countries would see an increase in the length of the stream network not meeting drinking water standards for N, up to about 20% (Figure 20). The stream network not meeting good ecological status for P or N is expected to increase in certain countries by up to about 25% for N, and 50% for P (Figure 18). At European scale, this translates into smaller changes, though, in the order of 1.5% for N and 5% for P (Figure 17).

*Figure 11 – Loads from urban wastewaters under different scenarios (POP90= estimated loads for 1990 based on population and connection rates; REP15= loads reported by Member States in 2015; FC= full compliance) in tonnes $BOD_5$ per year discharged to the stream network. Details on the loads under the different scenarios are provided in Annex II.*

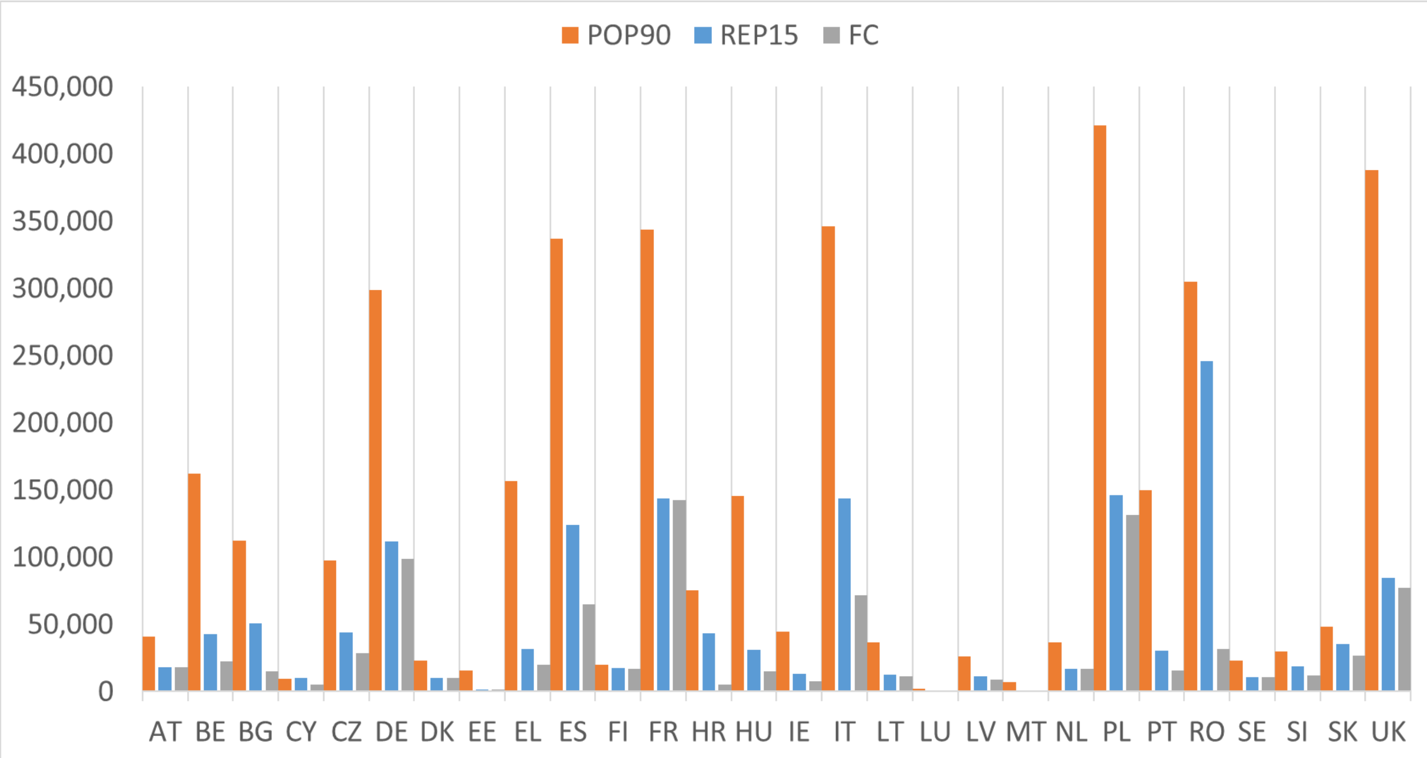


*Figure 12 - conditions of the European stream network with respect to $BOD_5$ quality standards under three scenarios*

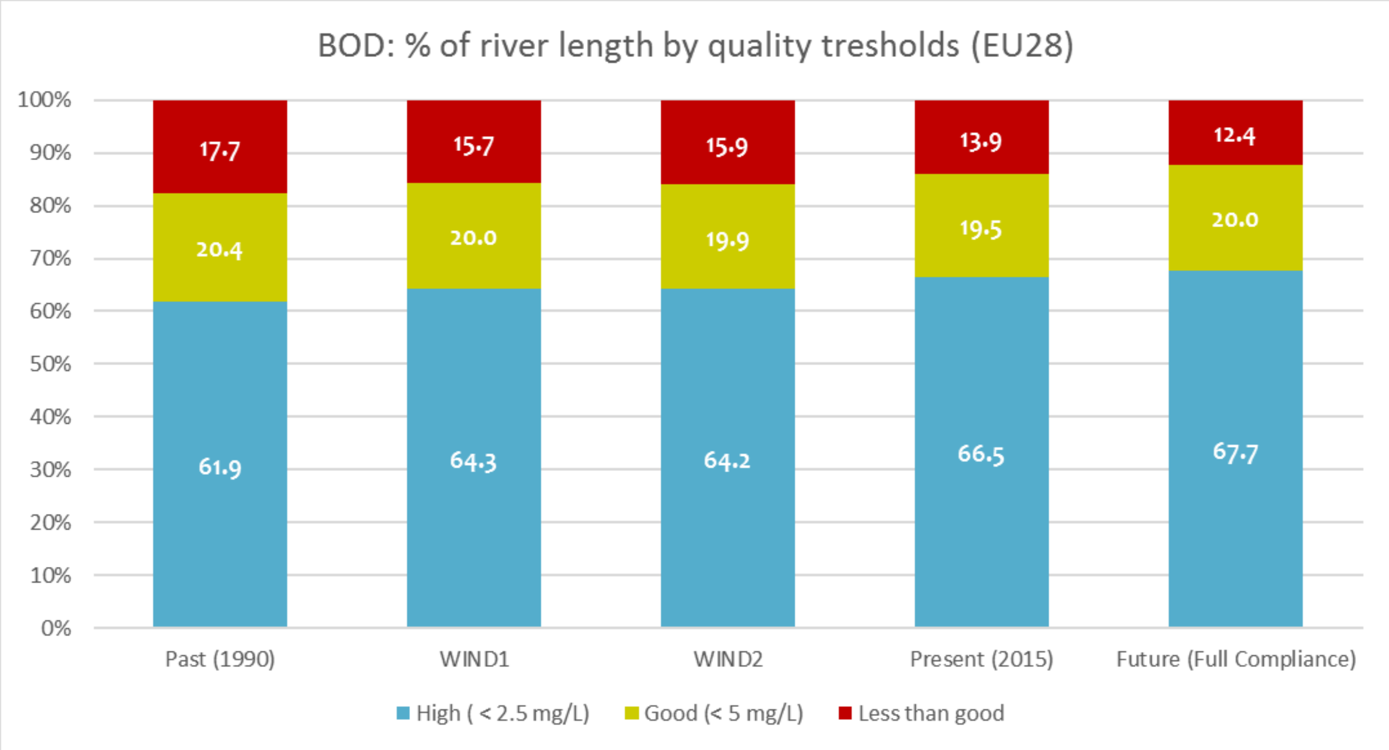



*Figure 13 – percent change of river length, by country, not meeting good ecological status standards for BOD, under WIND scenario compared to baseline (present) scenario.*

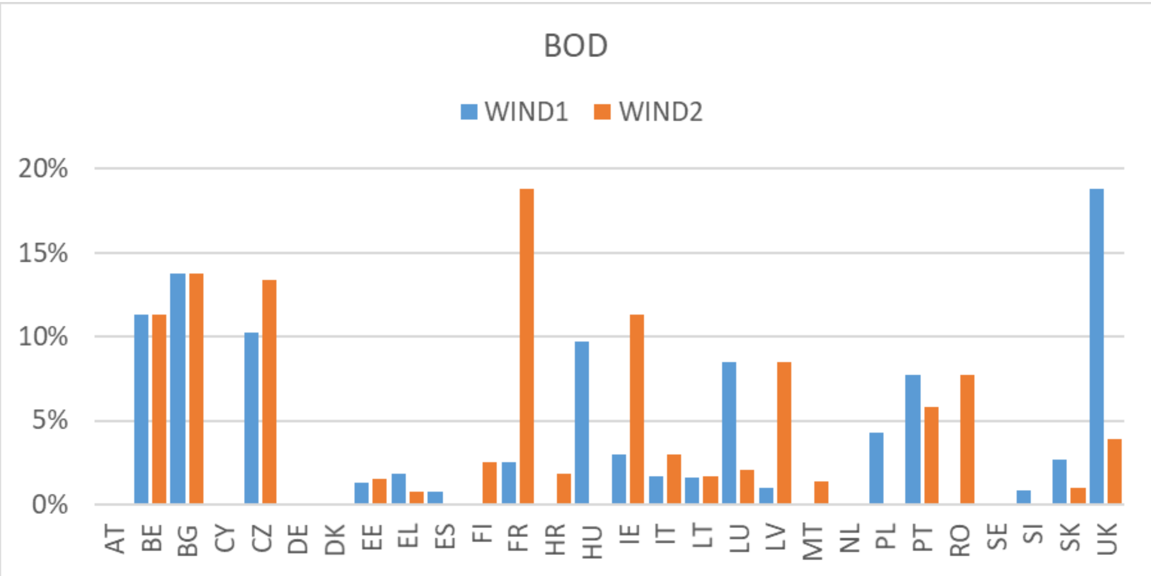
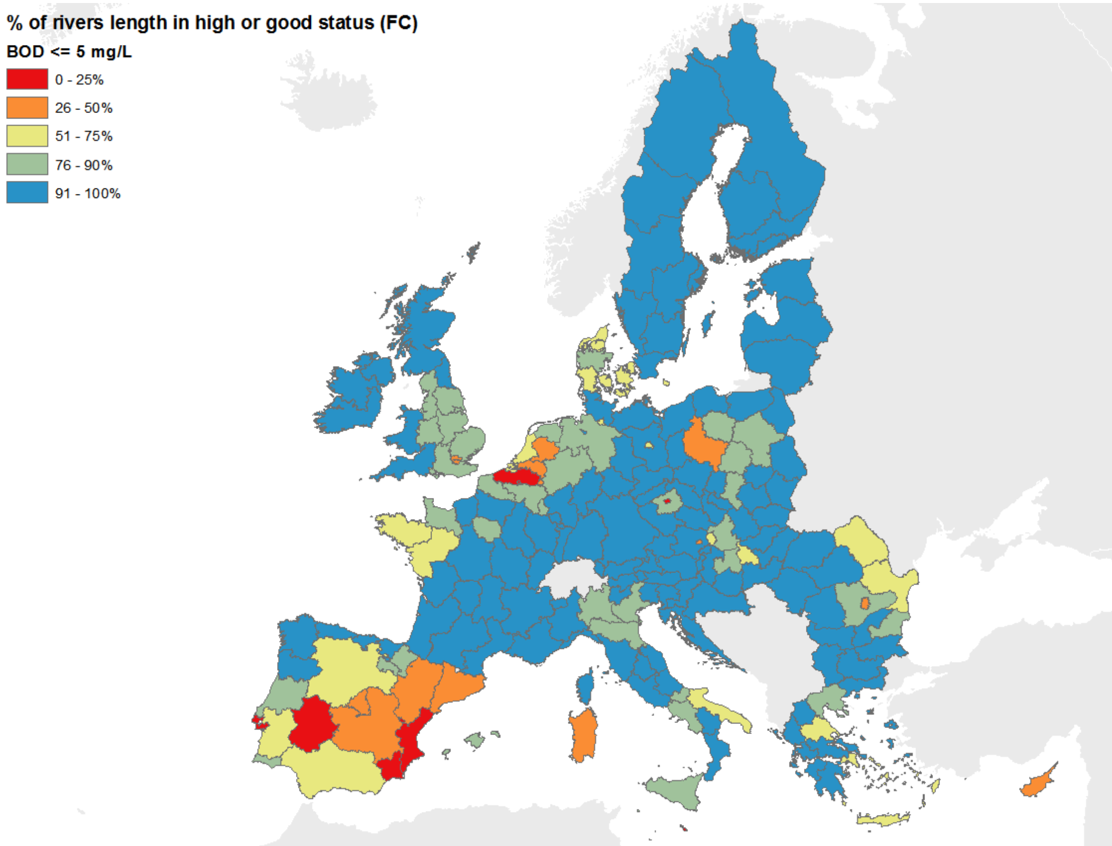


*Figure 14 - percentage of the stream network below an indicative $BOD_5$ threshold for good ecological status (5 mg/L), aggregated at NUTS2 level (NUTS1 for DE, UK, NL, BE). Full compliance (FC), Baseline as reported by Member States (2015).*



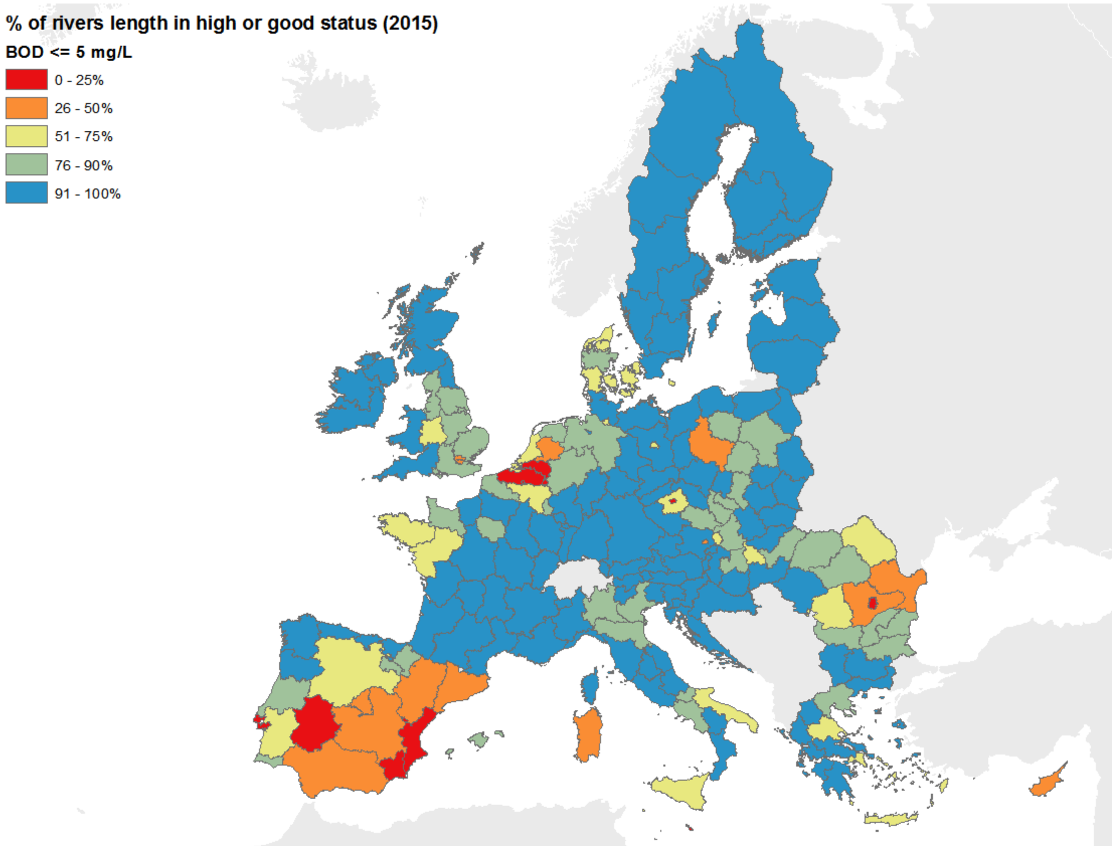
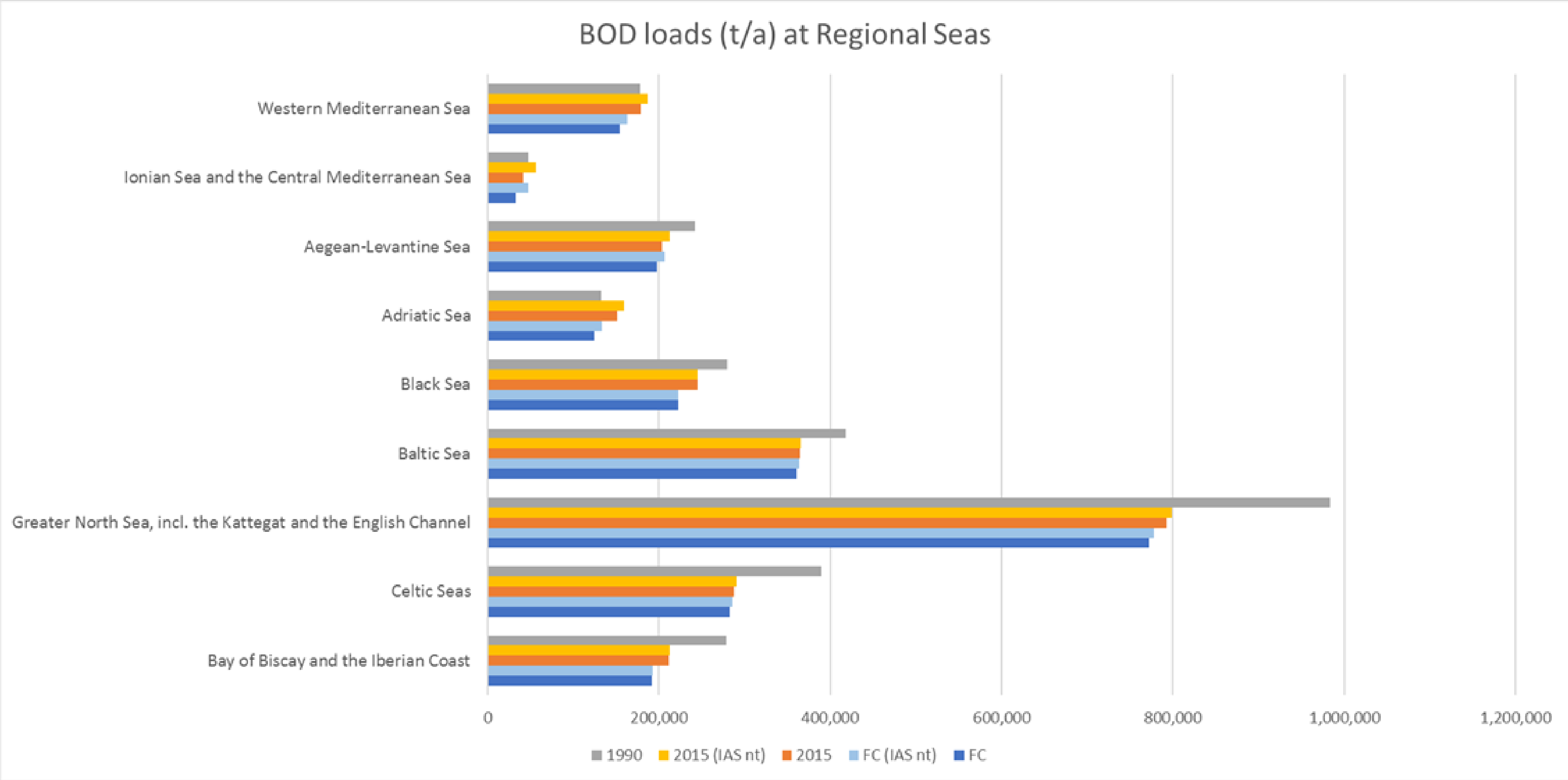
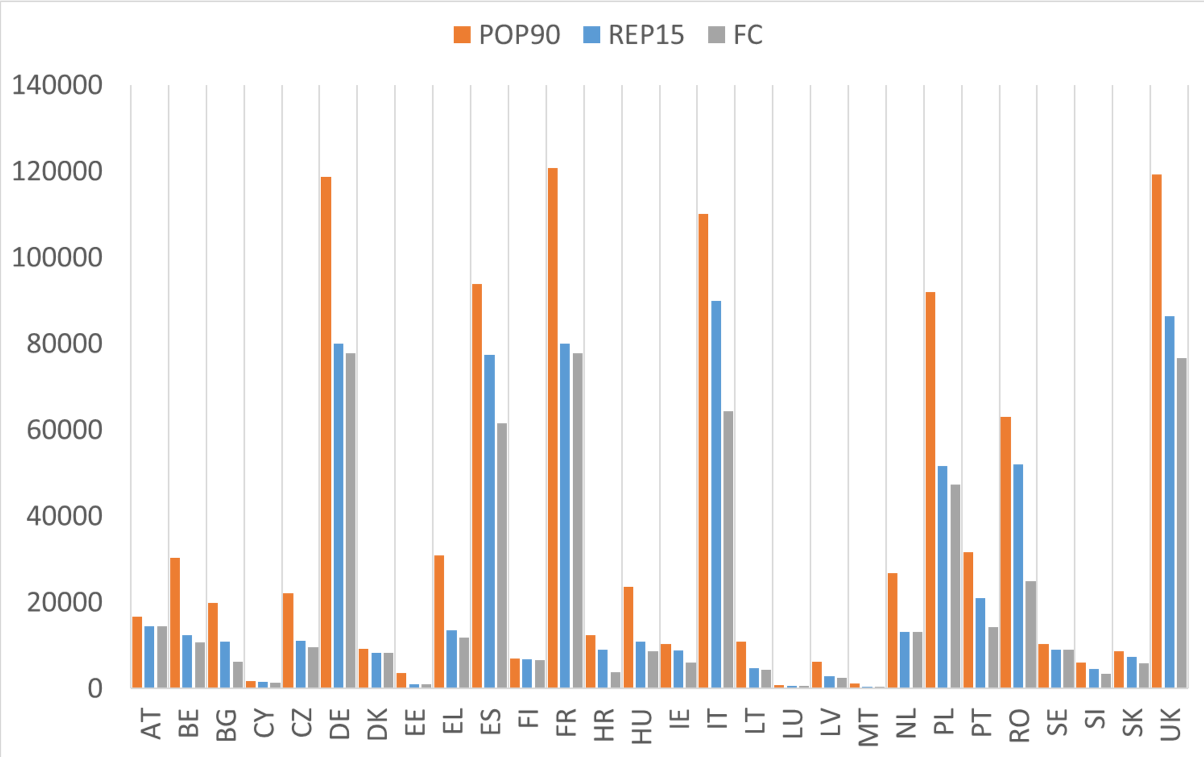



*Figure 15– loads of $BOD_5$ to the European regional seas under the different scenarios*[99]




[99] In the graph, we present loads computed under 2015 and full compliance conditions, both for IAS equivalent to primary treatment (denoted as "IAS nt") , and for IAS equivalent to the level of treatment of the WWTP of the corresponding agglomeration.

*Figure 16 – Loads from urban wastewaters under different scenarios (POP90= estimated loads for 1990 based on population and connection rates; REP15= loads reported by Member States in 2015; FC= full compliance) in tonnes N, P per year discharged to the stream network. Above: N; below: P. Details on the loads under the different scenarios are provided in Annex II*



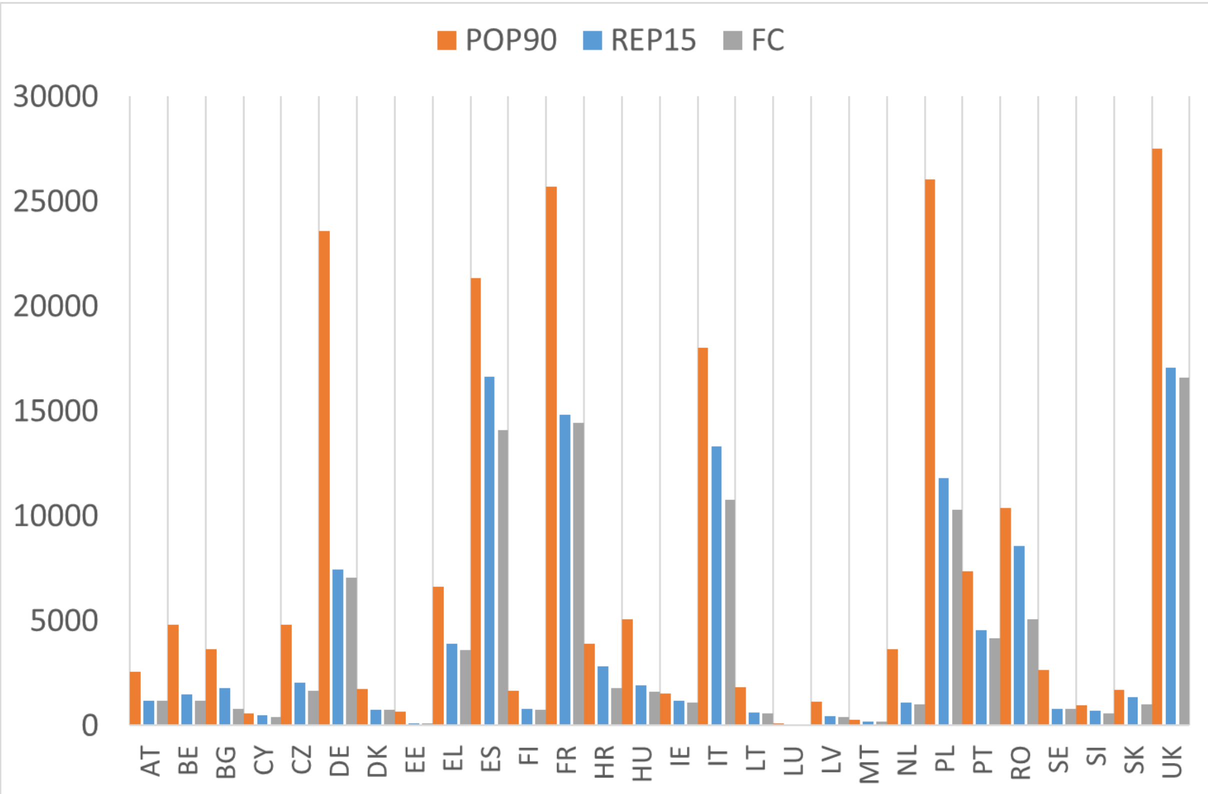
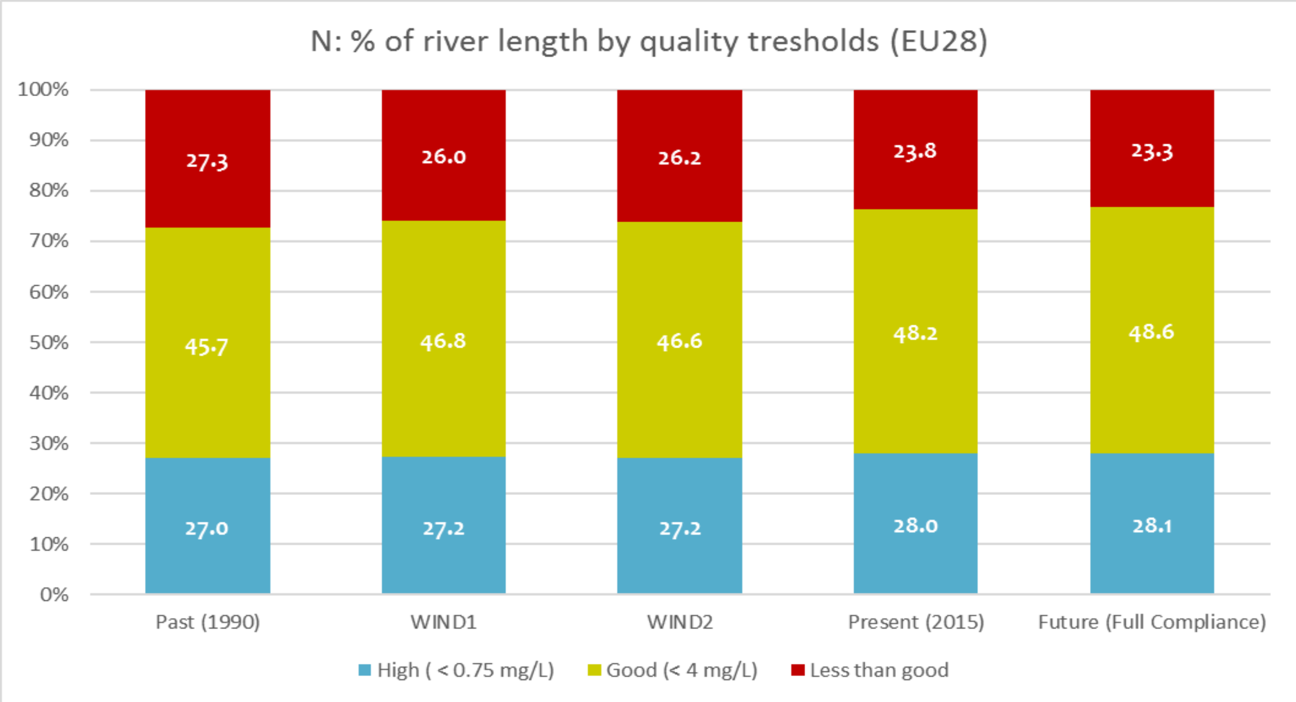



*Figure 17 – conditions of the European stream network with respect to nutrient quality standards under the different scenarios*



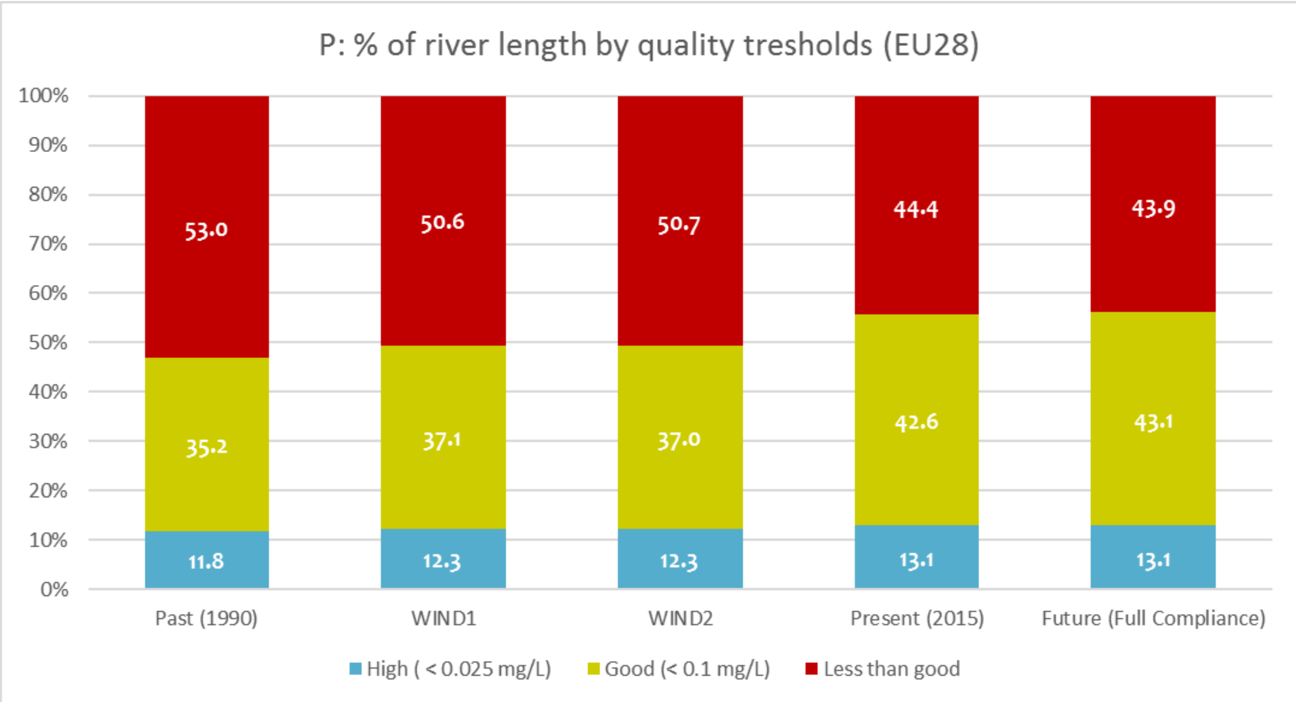



*Figure 18 – percent change of river length, by country, not meeting good ecological status standards for N, P, under WIND scenario compared to baseline (present) scenario.*

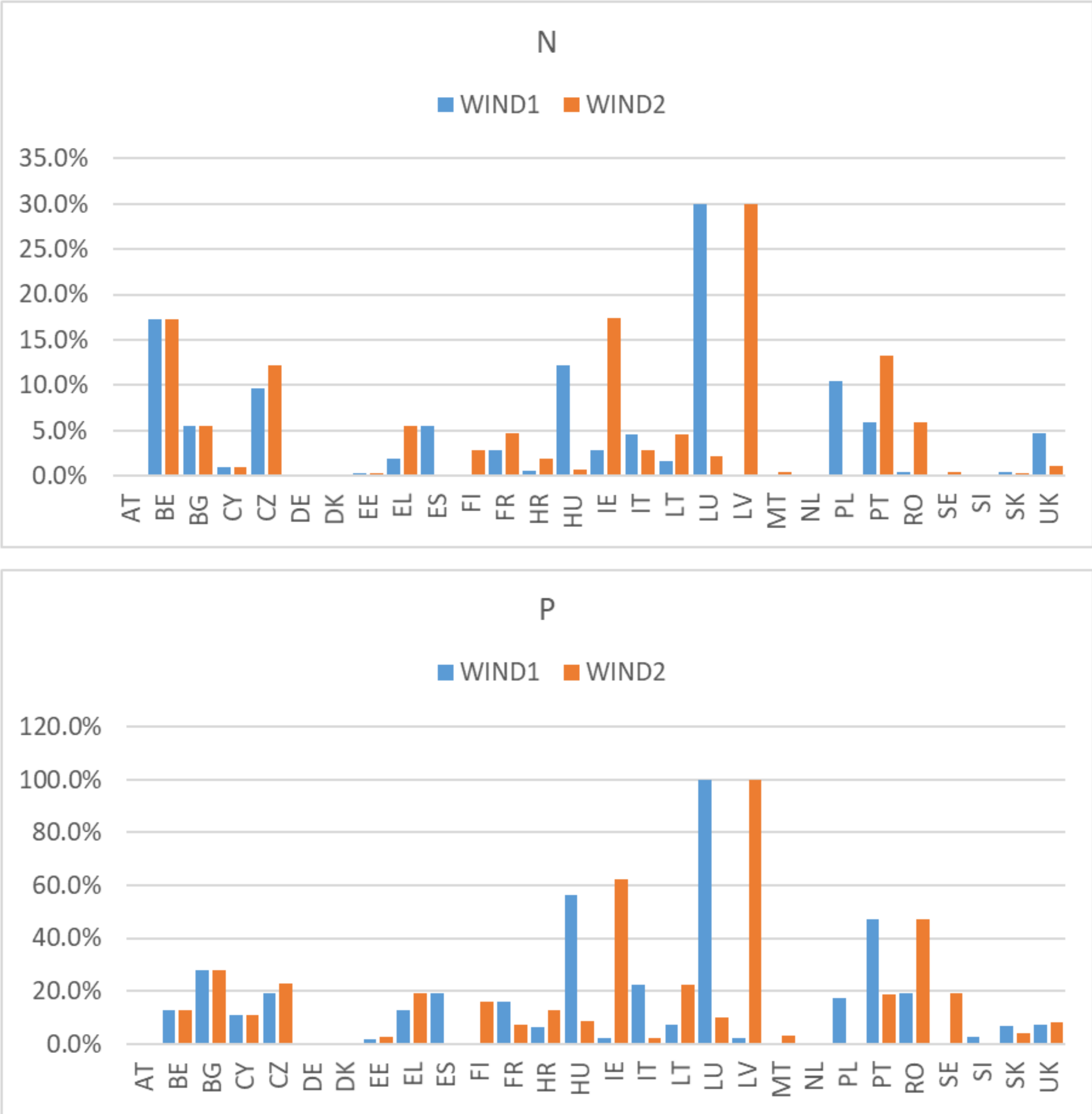


*Figure 19 – length of the European stream network below N limits for the DWD (11.3 mg/L) and precautionary limits for children (6 mg/L)*

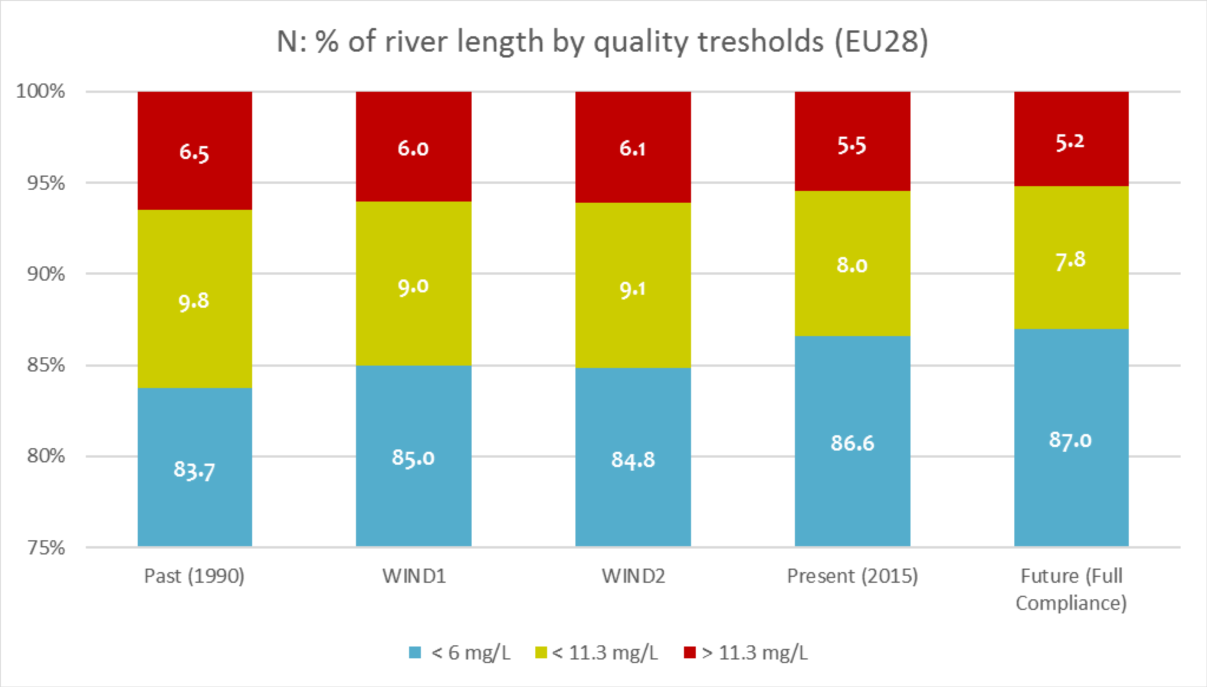



*Figure 20 – percent change of river length, by country, not meeting drinking water standards for N, under WIND scenario compared to baseline (present) scenario.*

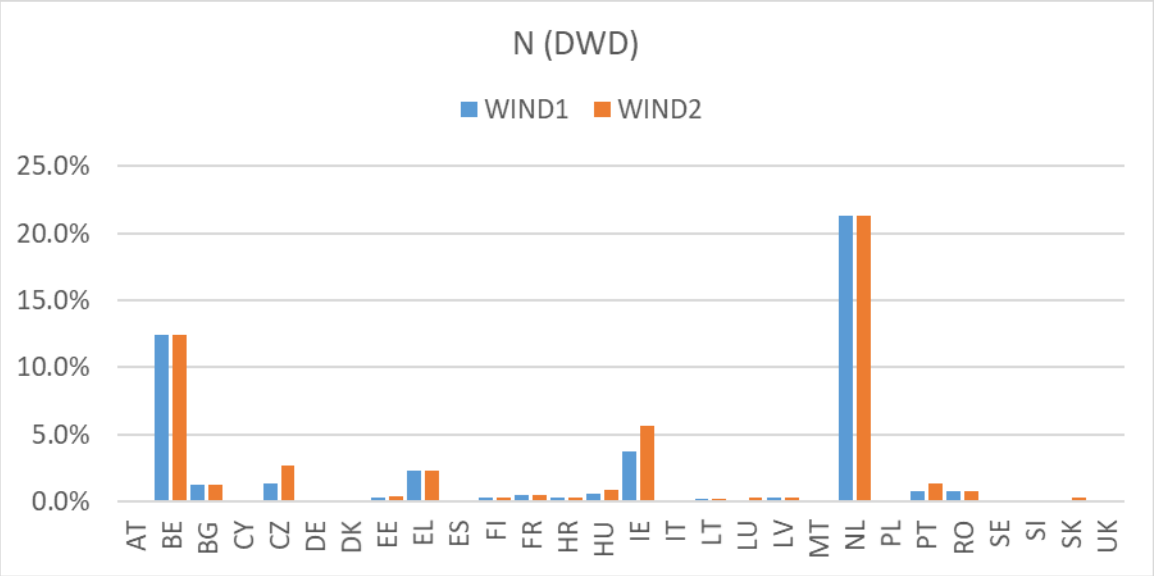
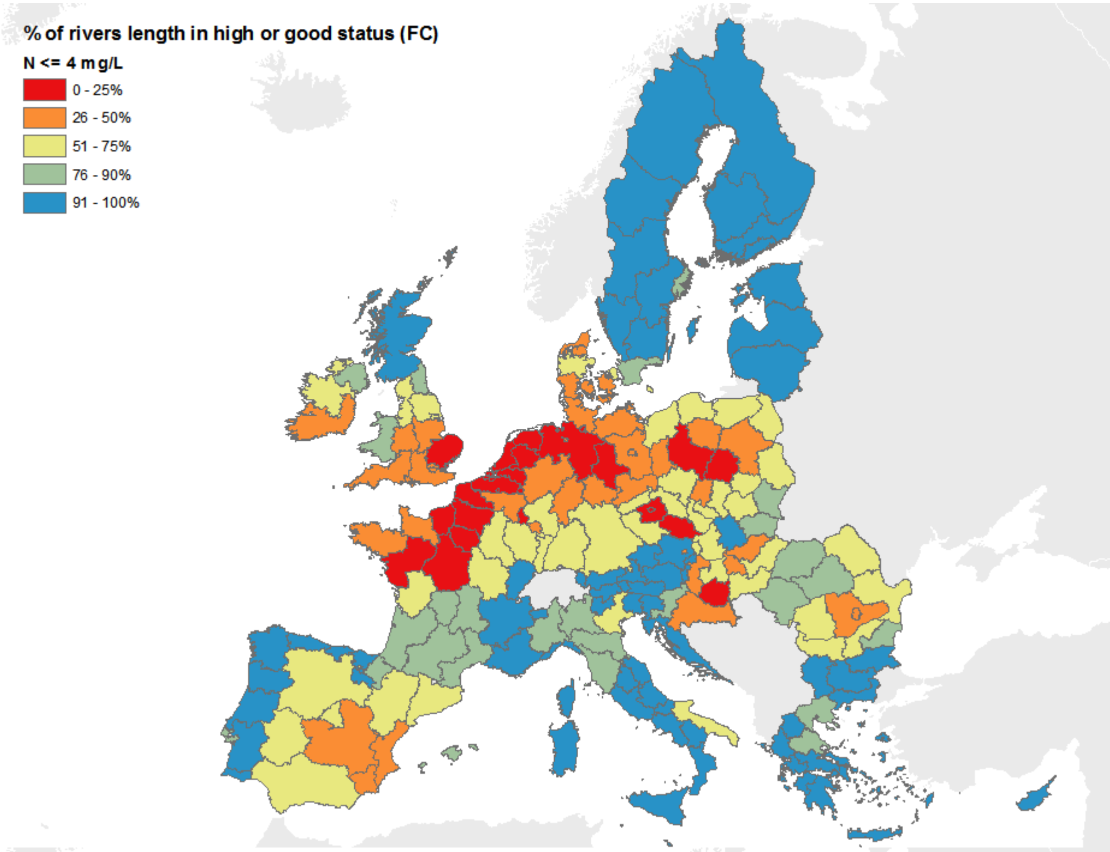


*Figure 21 - percentage of the stream network below an indicative total N threshold for good ecological status (4 mg/L), aggregated at NUTS2 level (NUTS1 for DE, UK, NL, BE). Full compliance (FC), Baseline as reported by Member Sates (2015).*



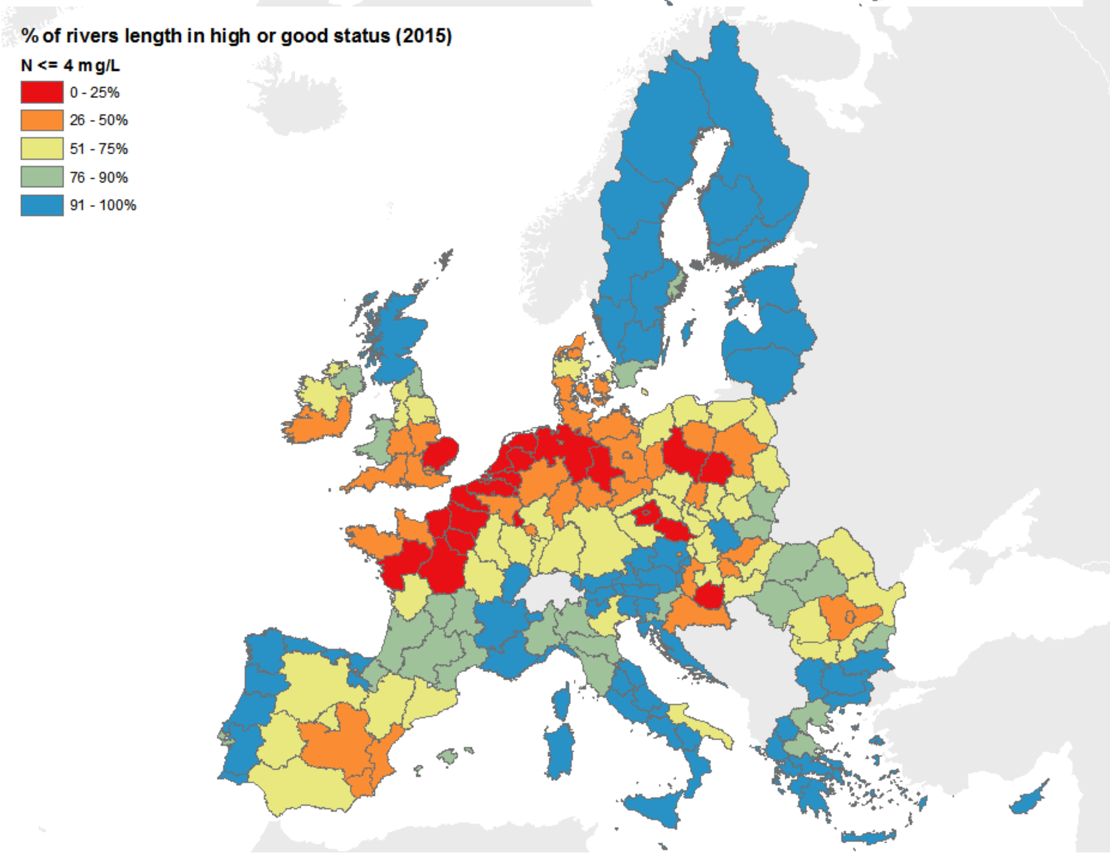
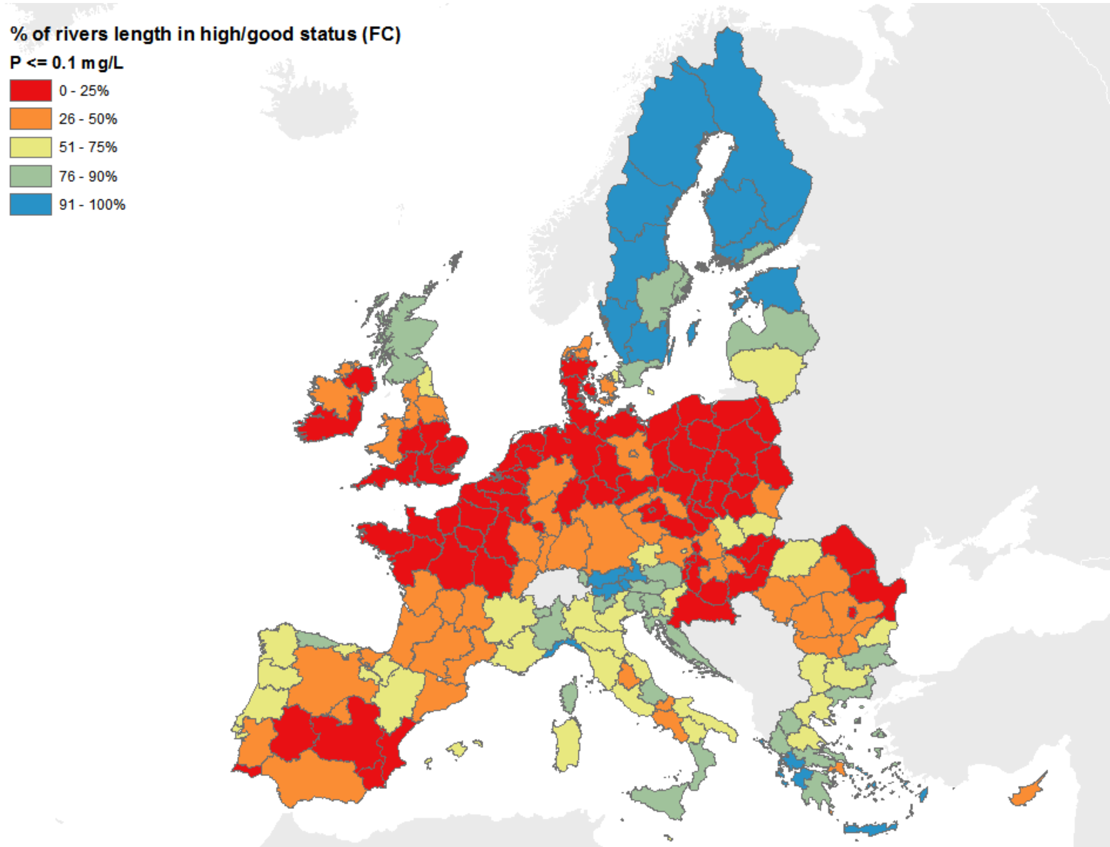



*Figure 22 - percentage of the stream network below an indicative total P threshold for good ecological status (0.1 mg/L), aggregated at NUTS2 level (NUTS1 for DE, UK, NL, BE). Full compliance (FC), Baseline as reported by Member Sates (2015).*



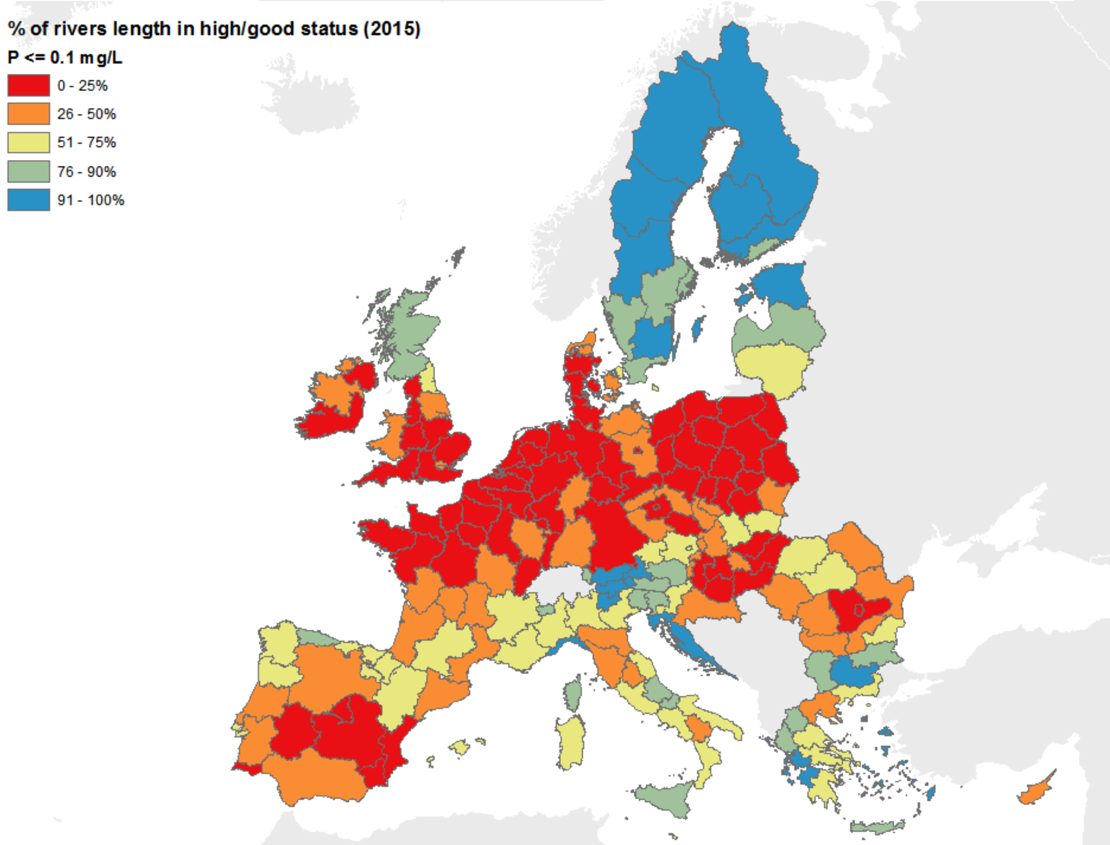
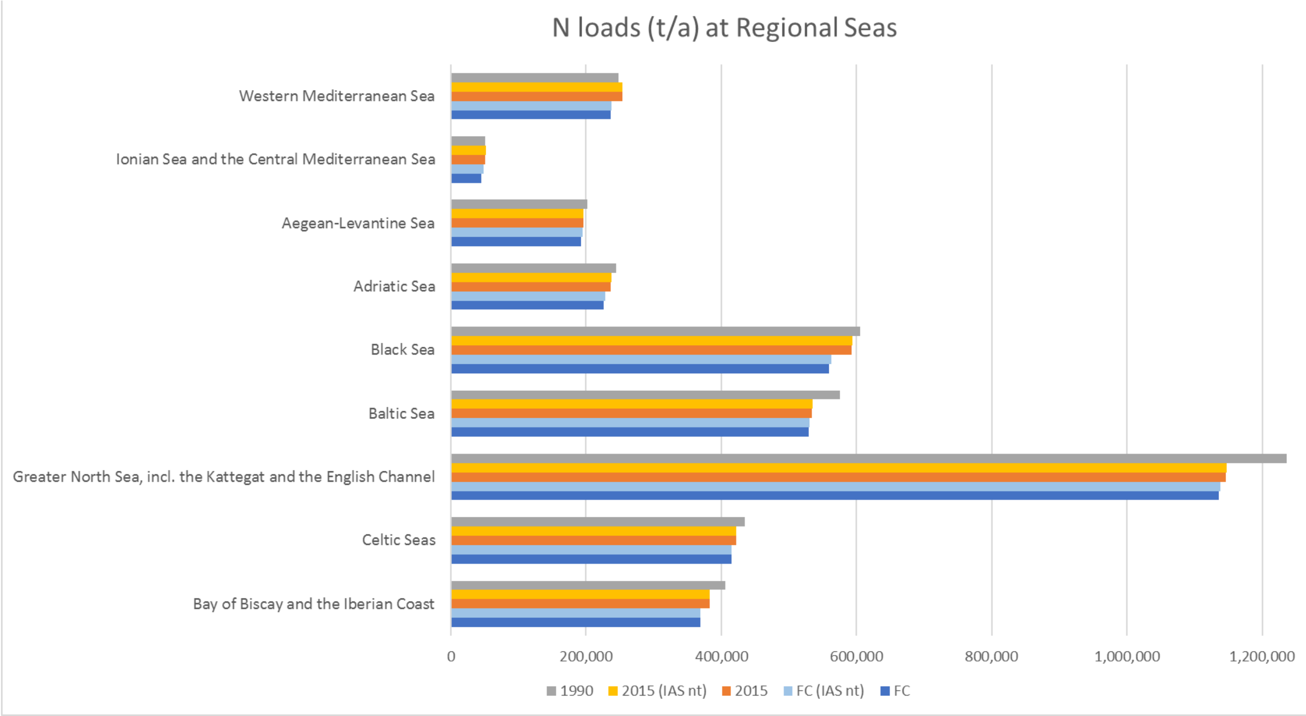
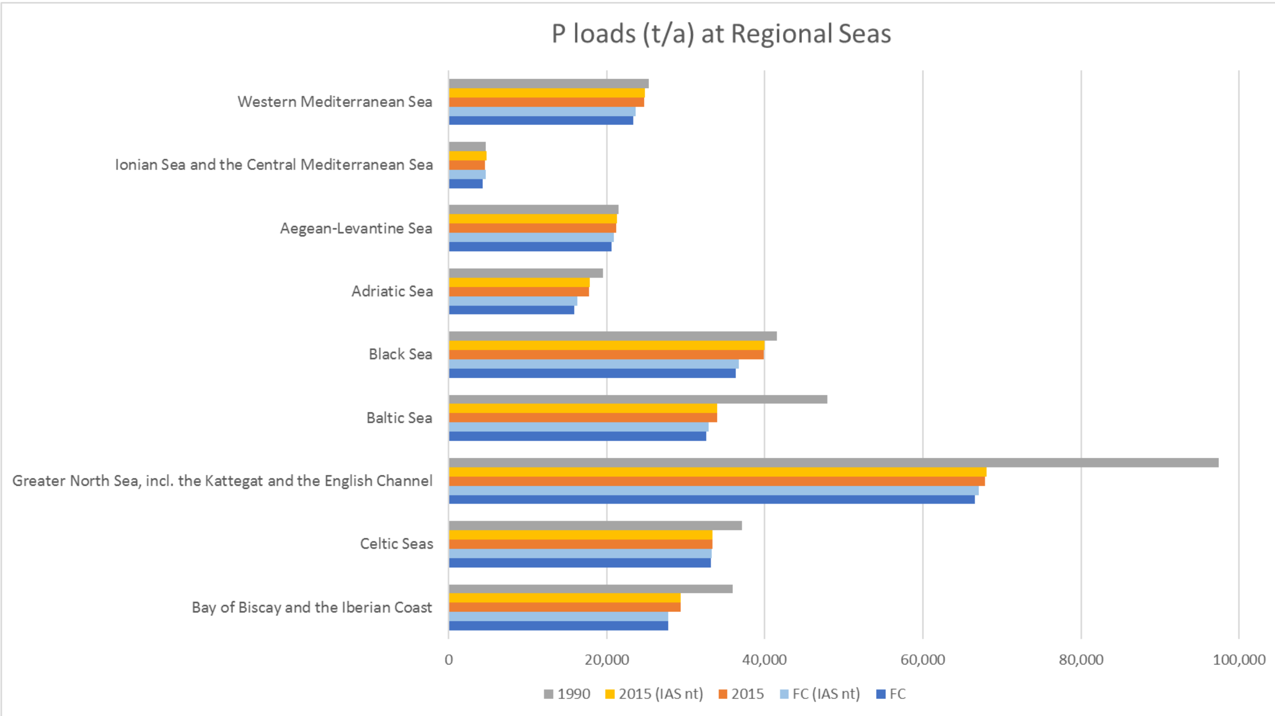



*Figure 23 – loads of nutrients to the European regional seas under the different scenarios*[100]




[100] In the graphs, we present loads computed under 2015 and full compliance conditions, both for IAS equivalent to primary treatment (denoted as "IAS nt") , and for IAS equivalent to the level of treatment of the WWTP of the corresponding agglomeration.

## 3.3 Impact of the level of IAS management

If all agglomerations are at full compliance, IAS represent a potentially significant residual share of the pollutant load discharged to the receiving water bodies (Figure 24). In EL, HR, HU, IE, IT and SK, IAS would represent more than 20% of the $BOD_5$ load discharged to the receiving water bodies under full compliance. For N and P, the incidence would be usually below 10%, and only in the cases of SK and HU slightly above. The impact of IAS on coliform concentrations in freshwater is quite high, and the stream network below good quality threshold concentrations increases significantly in regions with a sizable component of IAS (even above 50%, see Figure 28). On the contrary, the impact on coastal waters is negligible (results not shown here). The impact of IAS in terms of $BOD_5$ is potentially significant, but lower than for coliforms, with certain regions increasing the length of the stream network below good status thresholds by up to more than 20% (Figure 25).

For nutrients the impact of IAS is similar to that for BOD5, but smaller, with certain regions increasing the length of the stream network below good status thresholds by up to more than 5% (Figure 26, Figure 27). Although a large share of the total load of nutrients to seas comes from agriculture and other diffuse sources, improvements due to urban wastewater treatment are still sizable, as Figure 23 shows particularly for certain regions.

Under assumptions of scenario "IAS+", countries where some level of management of IAS is enforced are expected to show a lesser impact on water quality compared to the IAS scenario. In particular, AT, CY, DE, EE, UK are expected to have IAS equivalent to the agglomeration's WWTP, while CZ is expected to have IAS equivalent to the agglomeration's WWTP in 50% of the cases (see §2.3.5). Under this circumstance, the incidence shown in

Figure 24 would reduce to 0 (or by 50% in the case of CZ). In terms of water quality, under assumptions of IAS+ those countries where IAS effectiveness is ensured would see impacts offset as can be grasped from a comparison of scenarios in Figure 25 (for BOD) Figure 26 (for N), Figure 27 (for P) and Figure 28 (for coliforms).

*Figure 24 – Potential load assuming IAS correspond to a primary treatment, as a percentage of the total load under full compliance, for BOD, N and P from wastewater.*

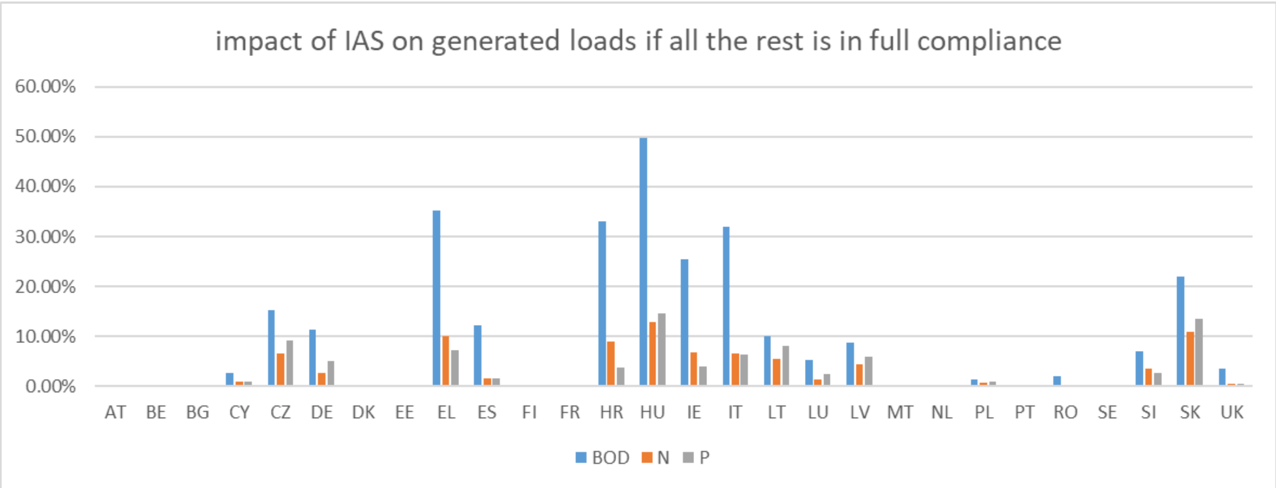
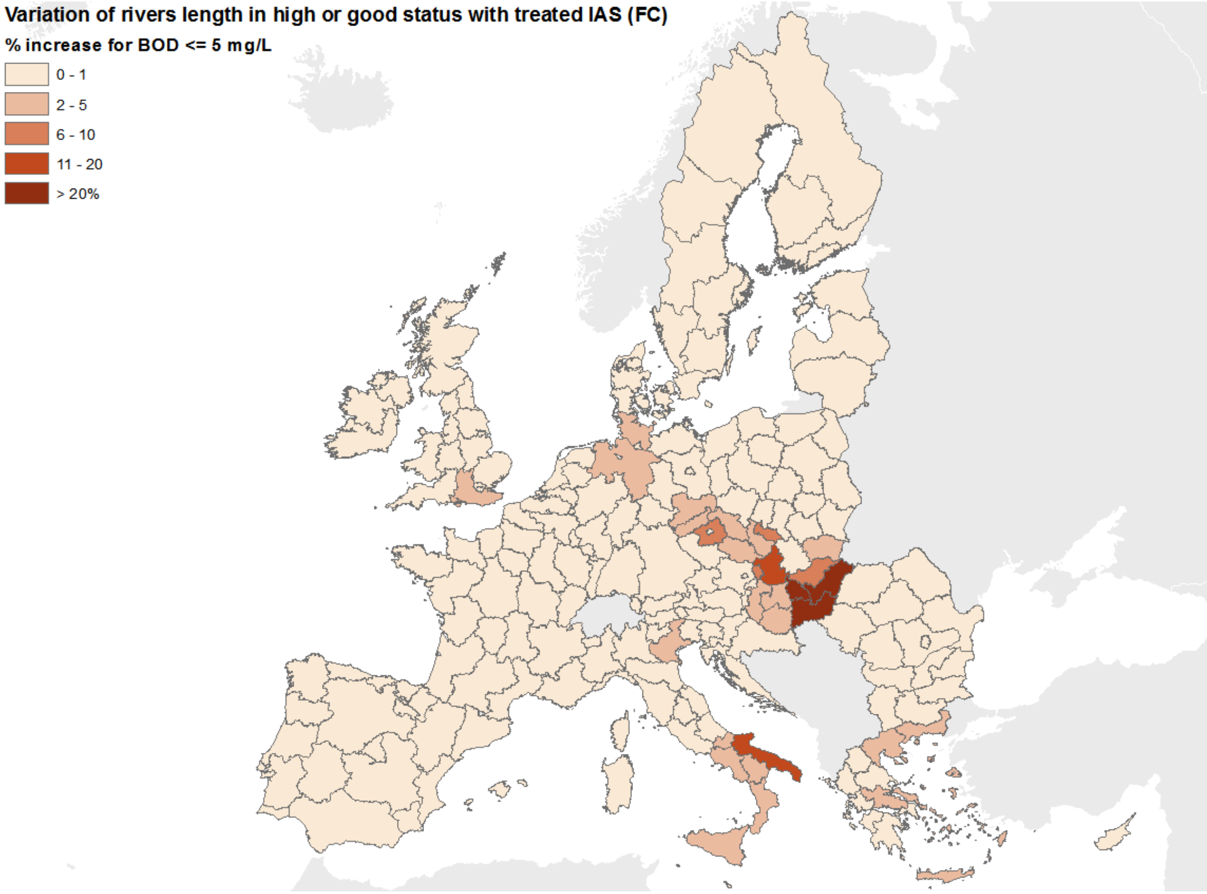


*Figure 25 BOD: %Variation of hydrographic network in good quality under IAS (above) and IAS+(below), by NUTS2 region (NUTS1 for DE, UK, NL, BE)*



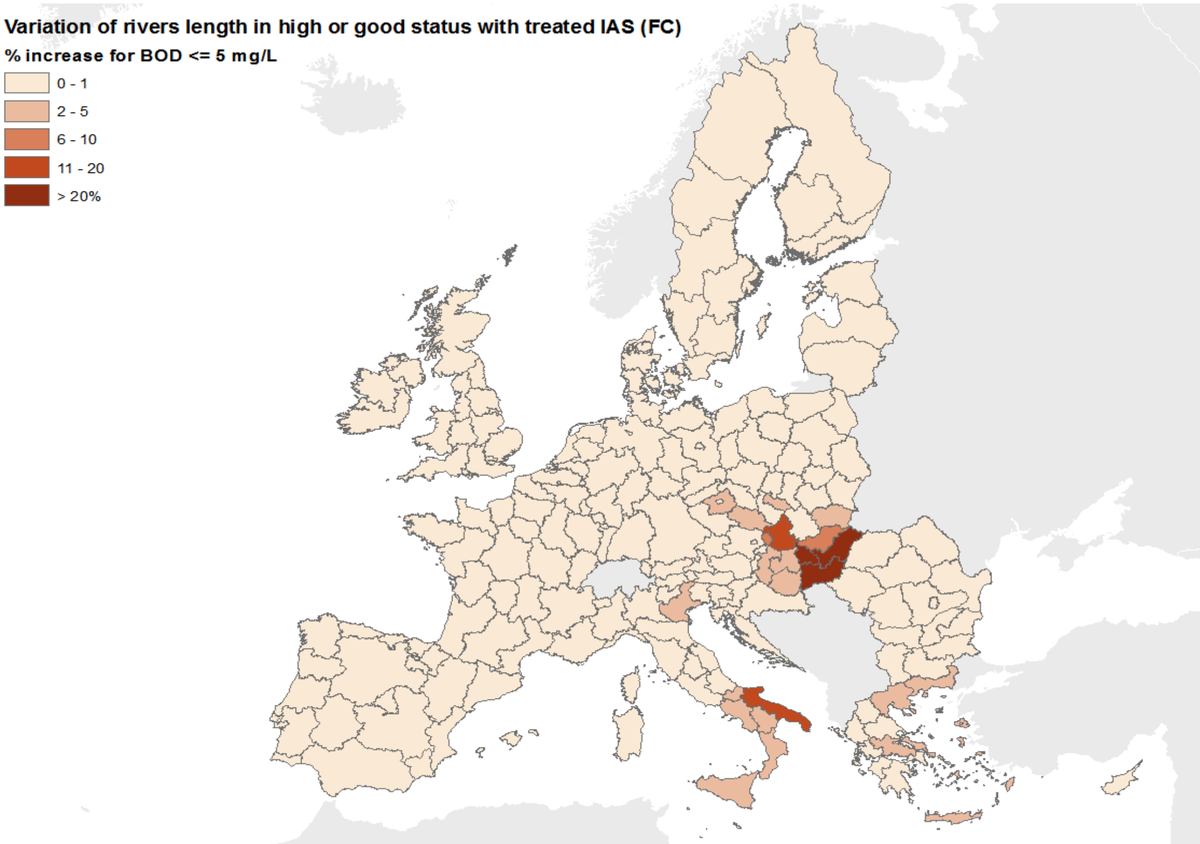
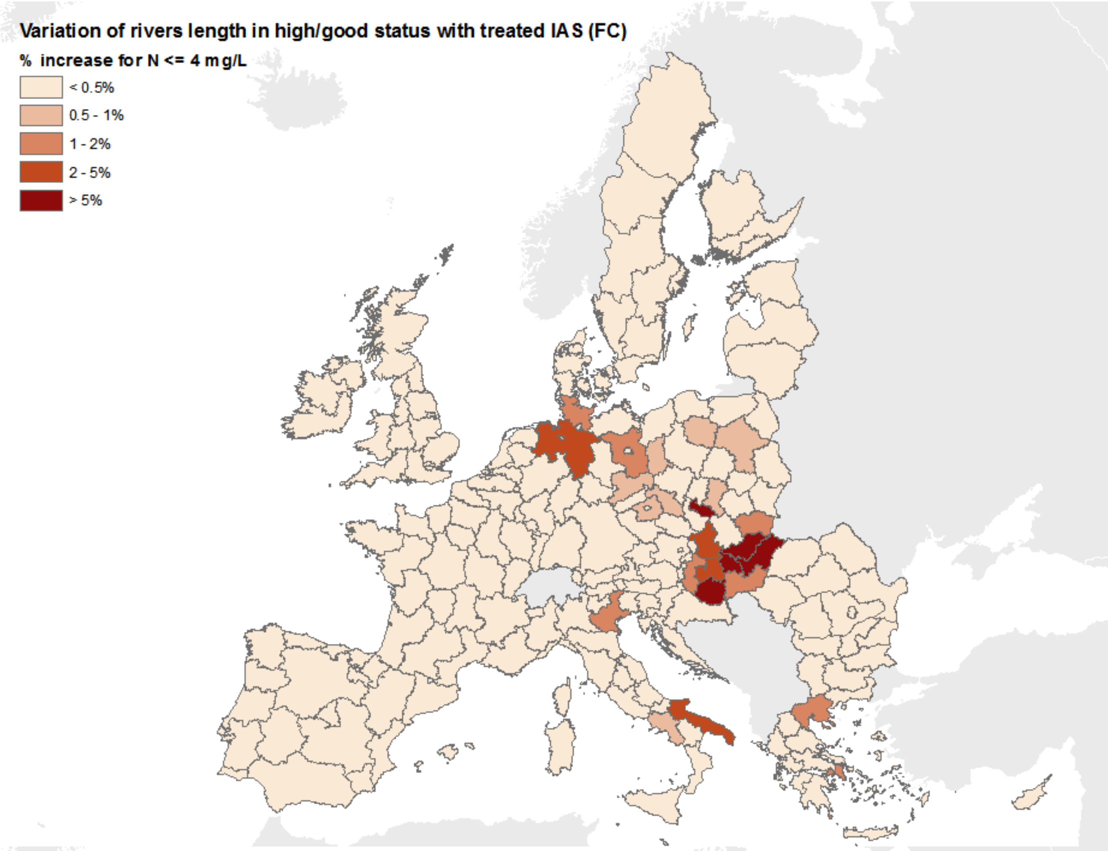



*Figure 26 N: %Variation of hydrographic network in good quality under IAS (above) and IAS+(below), by NUTS2 region (NUTS1 for DE, UK, NL, BE)*



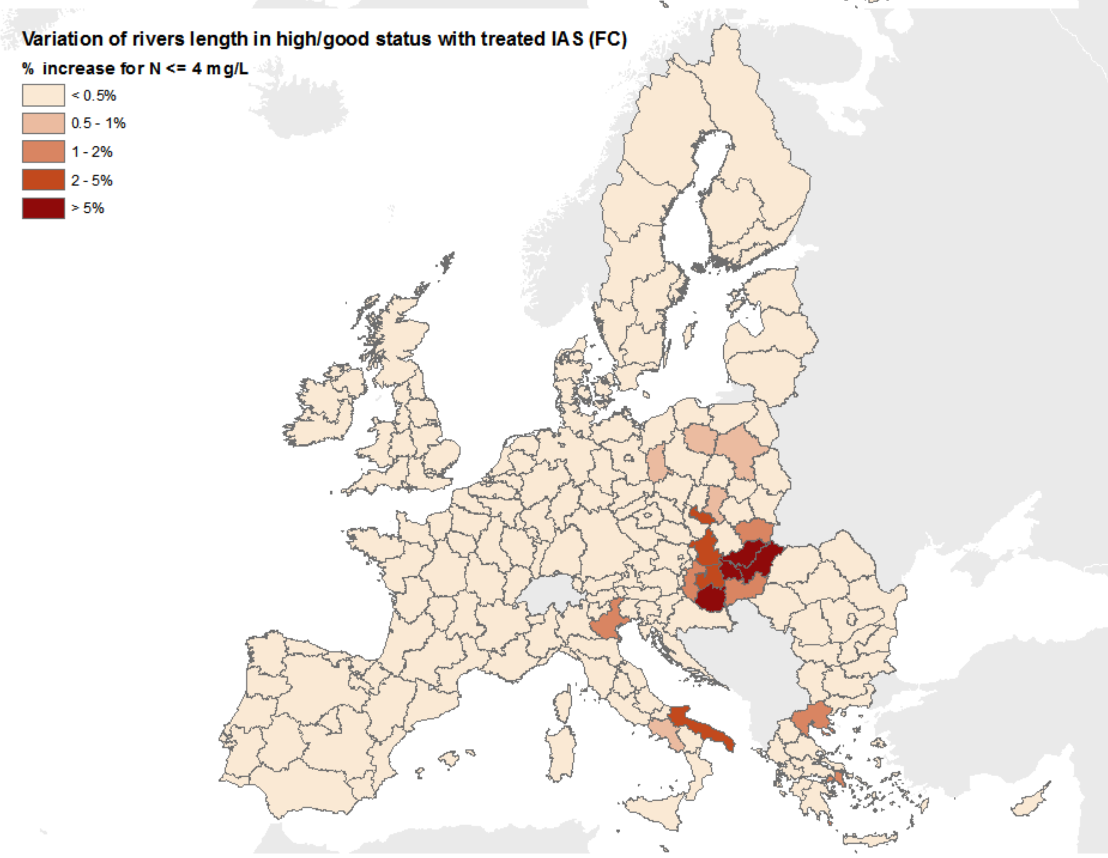
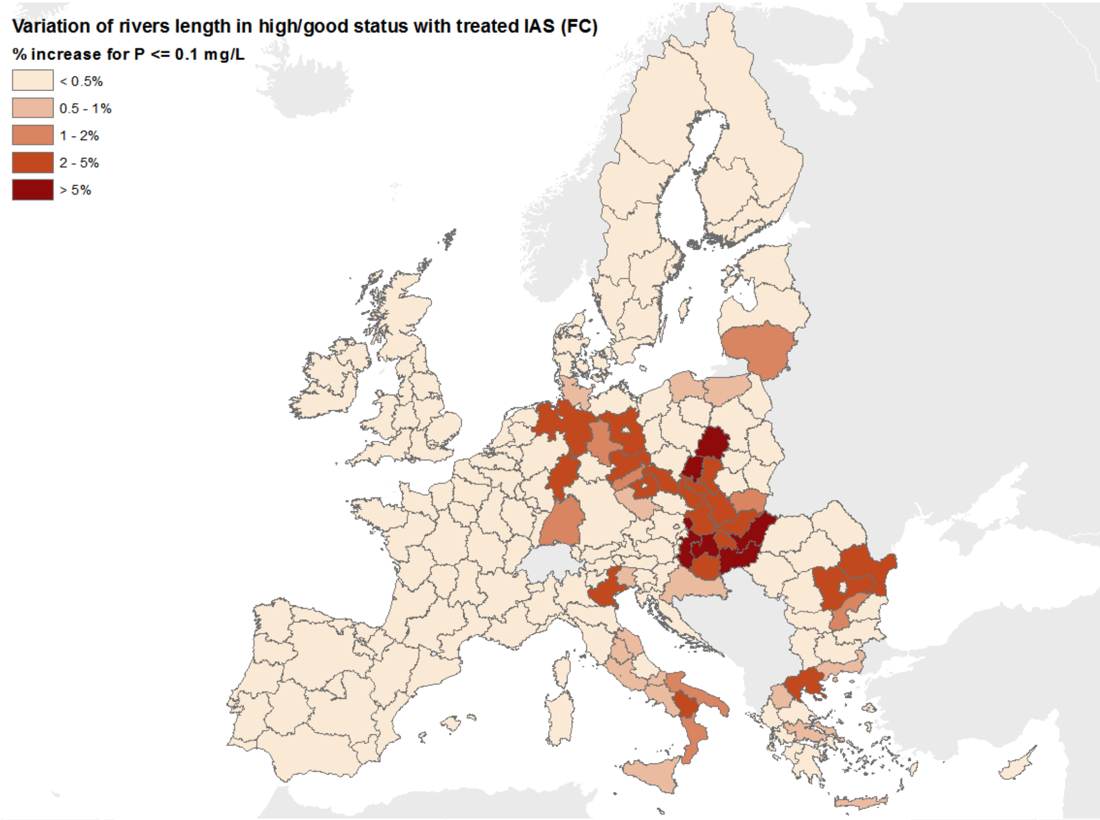



*Figure 27 P: %Variation of hydrographic network in good quality under IAS (above) and IAS+(below), by NUTS2 region (NUTS1 for DE, UK, NL, BE)*



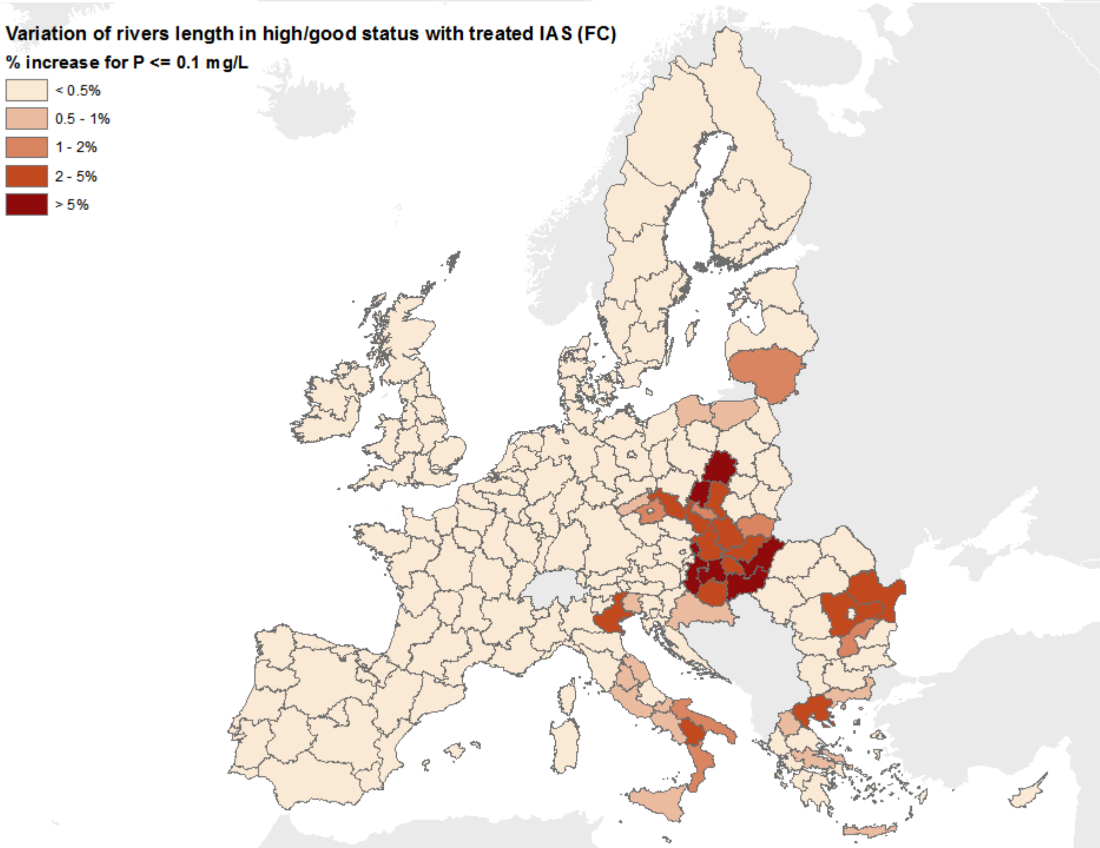
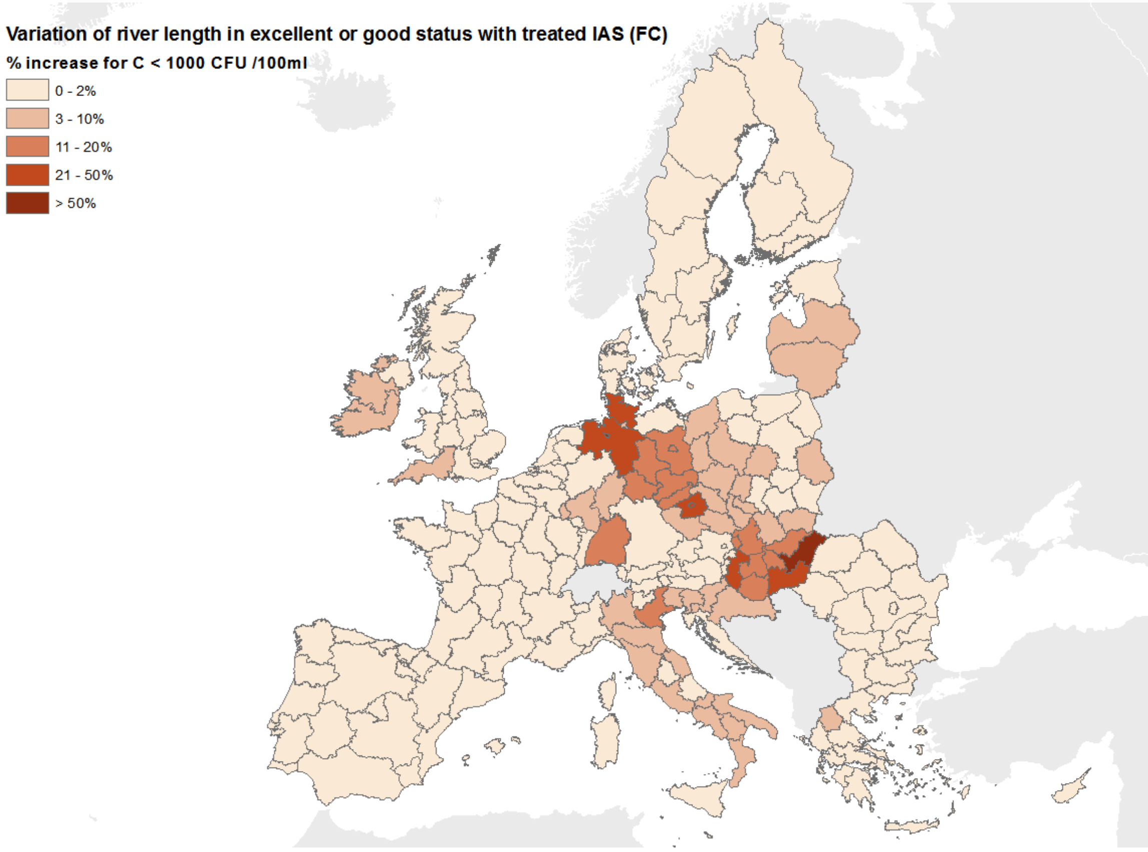
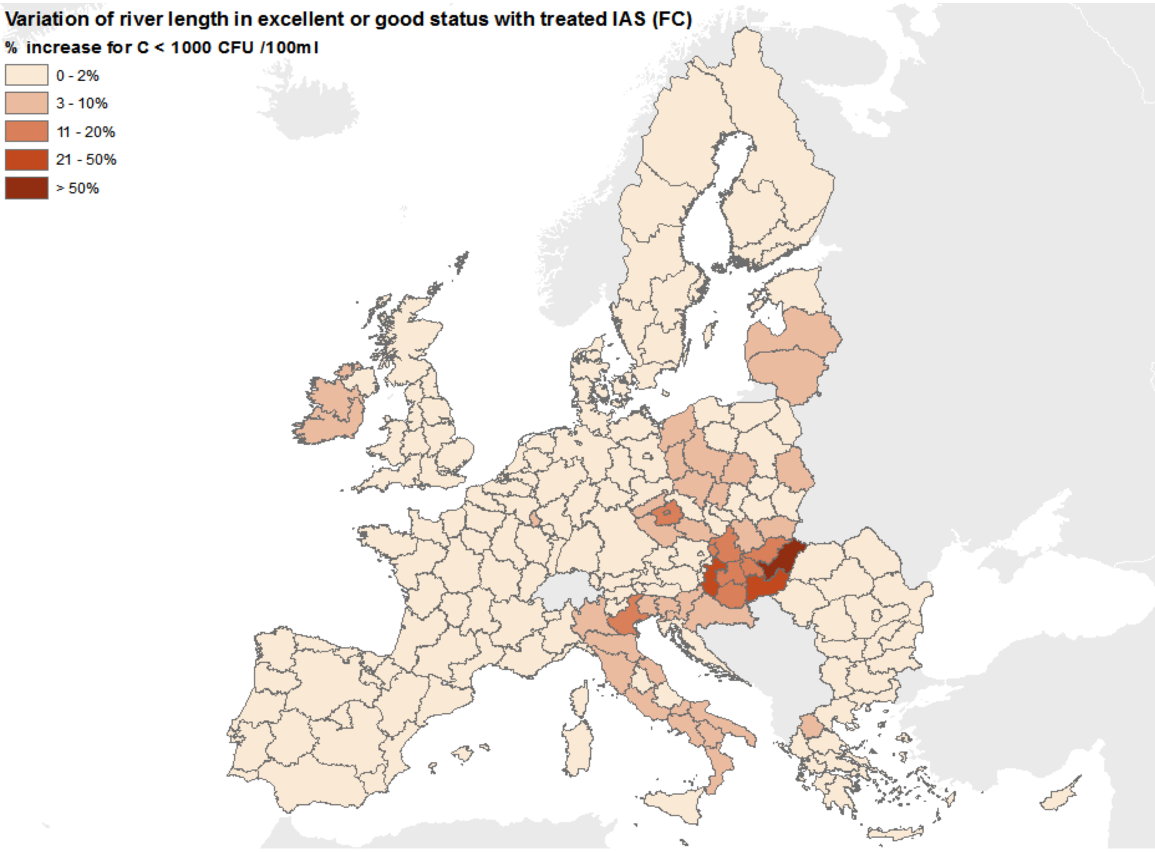



*Figure 28 coliforms: %Variation of hydrographic network in good quality under IAS (above) and IAS+(below), by NUTS2 region (NUTS1 for DE, UK, NL, BE)*



## 3.4 Chemicals

We have modelled the 12 metachemicals in Table 13, with the assumptions on removal efficiencies through primary sludge and in a secondary bioreactor shown in Figure 2, and correcting removal for tertiary treatment using Equation 7. Due to the limitations inherent to the assumptions made in the analysis (particularly about a uniform emission factor across the EU), we present the results aggregated for the EU and not in spatial form as for the other assessment endpoints. The graphs from Figure 29 to Figure 40 show, for the different metachemicals, the change in the percent of the European stream network (by length) below different thresholds of metachemical concentration. As emission rates are not known, we can only evaluate the change in the river length below conventional thresholds. We have conventionally set as thresholds the median and the 75th percentile of metachemical concentrations in the pre-directive scenario. If we assume that, in the pre-directive scenario, at least 50% of the sub-basins was in good conditions in terms of chemical pollution, and 75% in acceptable conditions[101], these thresholds may be regarded as conventional quality standards. However, it should be noted that such thresholds retain a meaning only when comparing scenarios, and not in absolute terms.

Having made these clarifications, we can examine the results for the 12 metachemicals, having in mind the substances that may correspond to each metachemical.

Table 10 associates all chemicals of concern for this study to one metachemical, based on the evidence shown above. Metachemicals c#1 and c#13 have limited practical interest as they correspond to a contradictory behavior in the WWTP and in rivers, and will not be considered any longer in this report. For metachemicals c#4 and c#14 we did not find corresponding chemicals, in the list considered here. The other chemicals were all associated to one metachemical. For those substances essentially unaffected by the wastewater treatment plant (Table 10) apparently the directive can be said to have brought no benefit.

The different metachemicals, hence substances of concern, have different environmental and WWTP fate and, consequently, react differently to the implementation of the UWWTD. However, by inspecting the graphs from Figure 29 to Figure 40, one can appreciate that all metachemicals show a similar change in the length of the stream network below the two thresholds. About 30% of the stream network is typically below the median threshold, and about 55% is below the $75^{th}$ percentile threshold in the pre-directive scenario. These percentages increase to more than 55% and around 80%, respectively, in the baseline scenario, and slightly higher in the full compliance scenario. This pattern is very similar for all metachemicals considered in this study, in spite of the variability of their fate.

This outcome is somehow unexpected, and supports the hypothesis that the effect of the UWWTD has been significant on all chemicals undergoing some form of retention in WWTPs (either in sludges, or through removal). As the pattern of improvement is rather constant, one may argue that the effects of wastewater treatment in terms of chemical pollution are relatively independent of the molecule, provided it actually undergoes some retention in the WWTP.

For substances corresponding to metachemicals c#4, c#5, c#9, c#10 and c#15, however, while the UWWTD may have helped reducing the loads of these chemicals to the stream network, it may have also caused a possible significant loading to soils through the reuse of sludge in agriculture, which may represent in part a transfer of, instead of a real solution to, the problem.

[101] Percentiles are computed on the population of sub-basins included in the model; each sub-basin may have a different length of the corresponding river segment, hence the % of length below a given percentile is not the corresponding percentage.

*Figure 29 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#2, e.g. 2-Ethylhexyl-4-methylphenol (EHMC)*

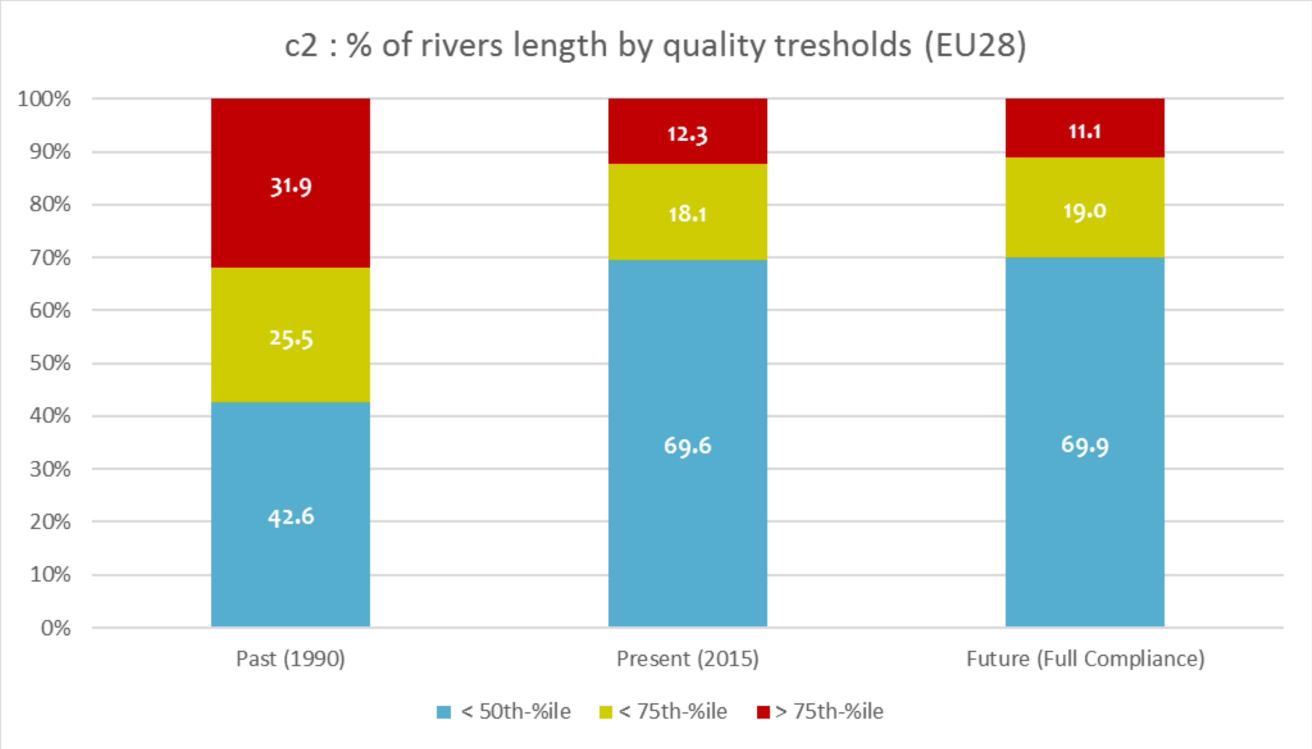



*Figure 30 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#3, e.g. estrone (E1), estradiol (E2), Bisphenol A*

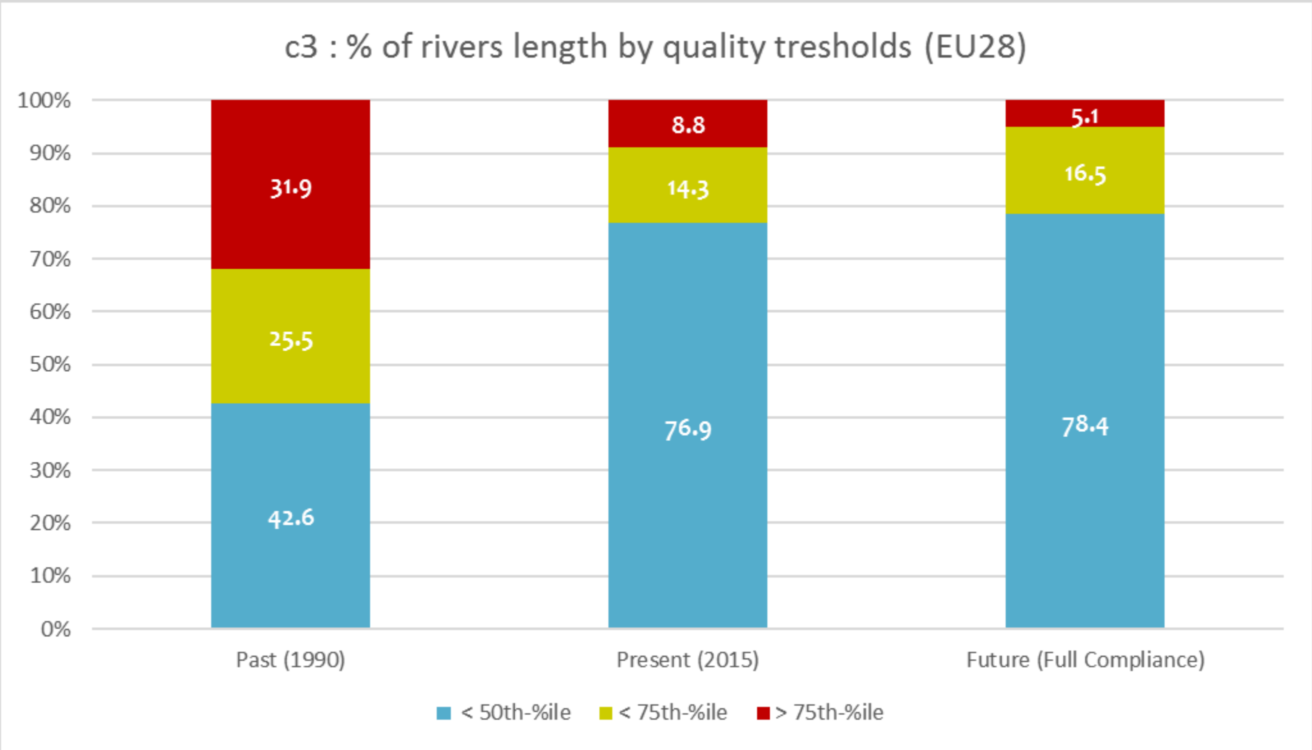


*Figure 31 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#4. No example chemicals considered in this study.*

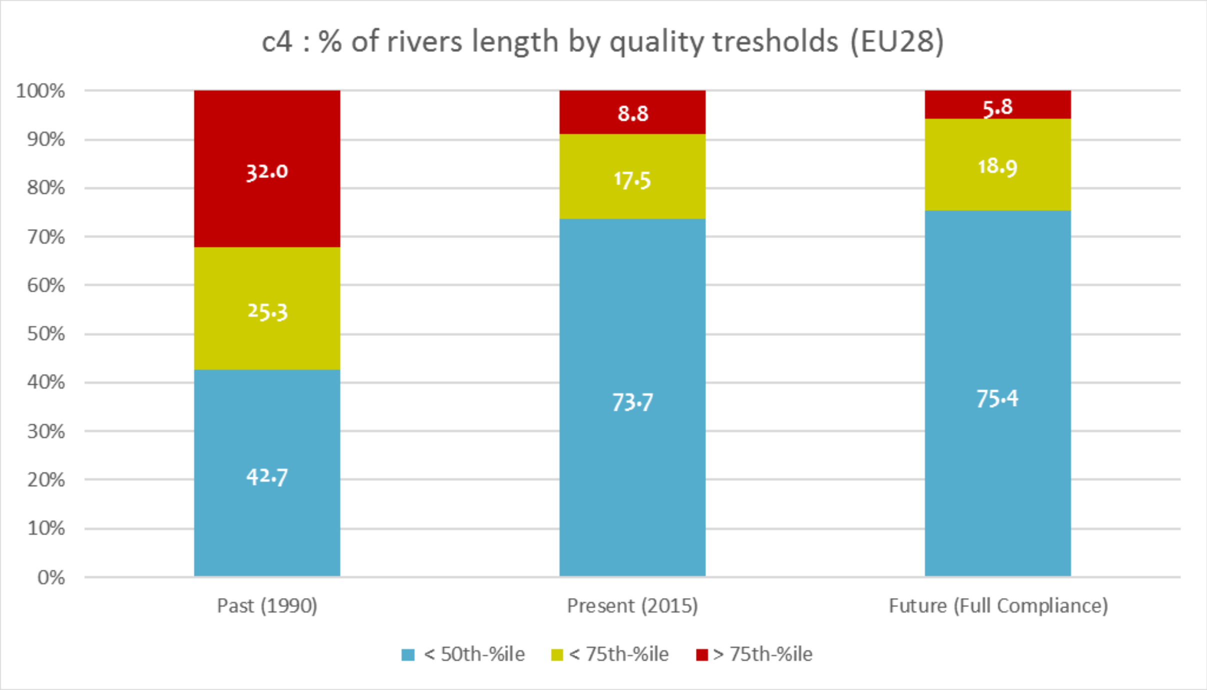



*Figure 32 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#5, e.g. Triclosan, Ciprofloxacin*

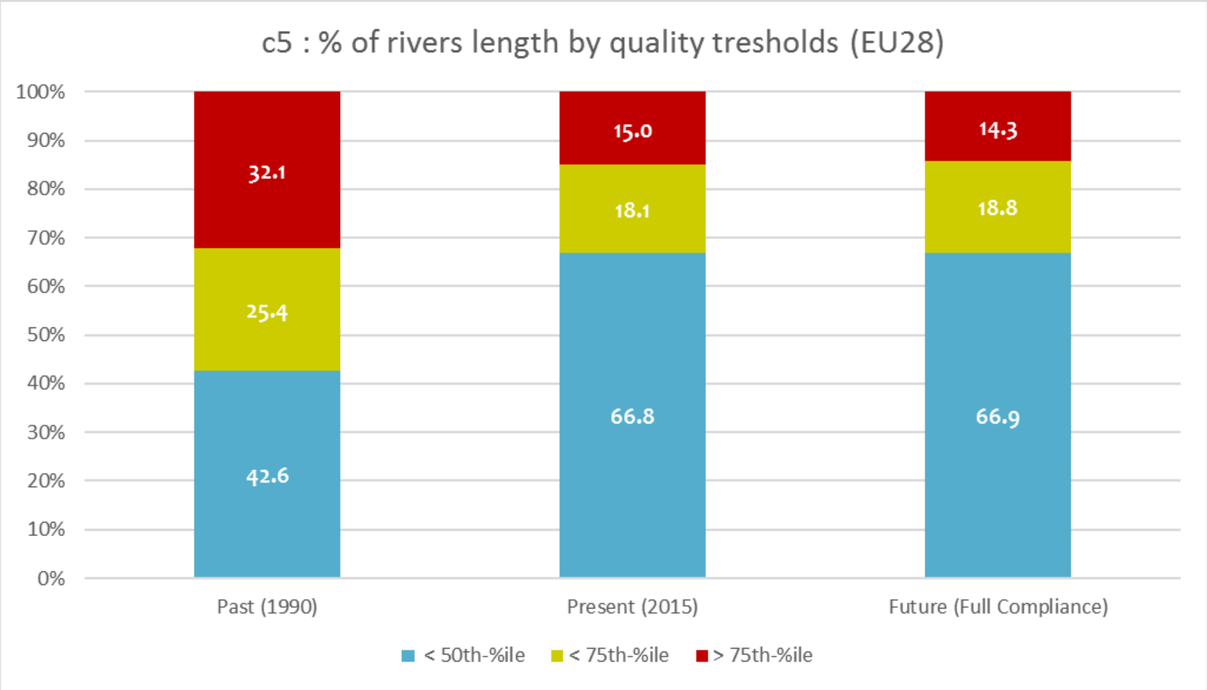


*Figure 33 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#6, e.g. Diclofenac*

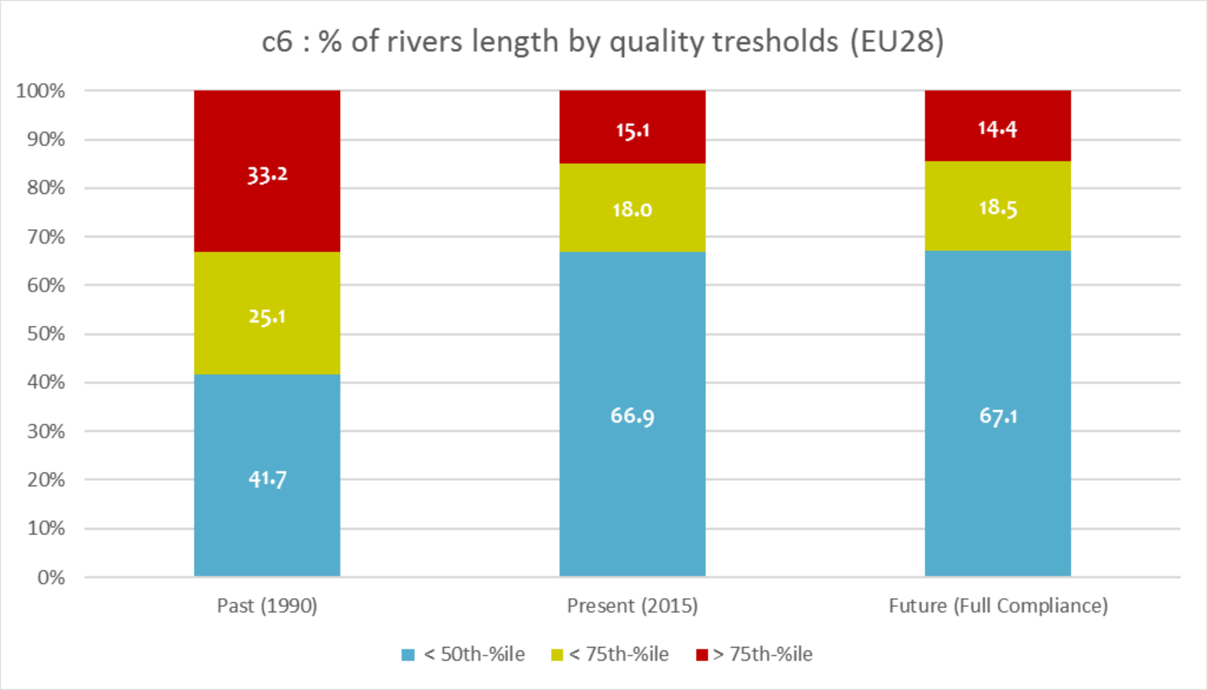



*Figure 34 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#7, e.g. OP, Acrylamide, ethinylestradiol (EE2), Ibuprofen*

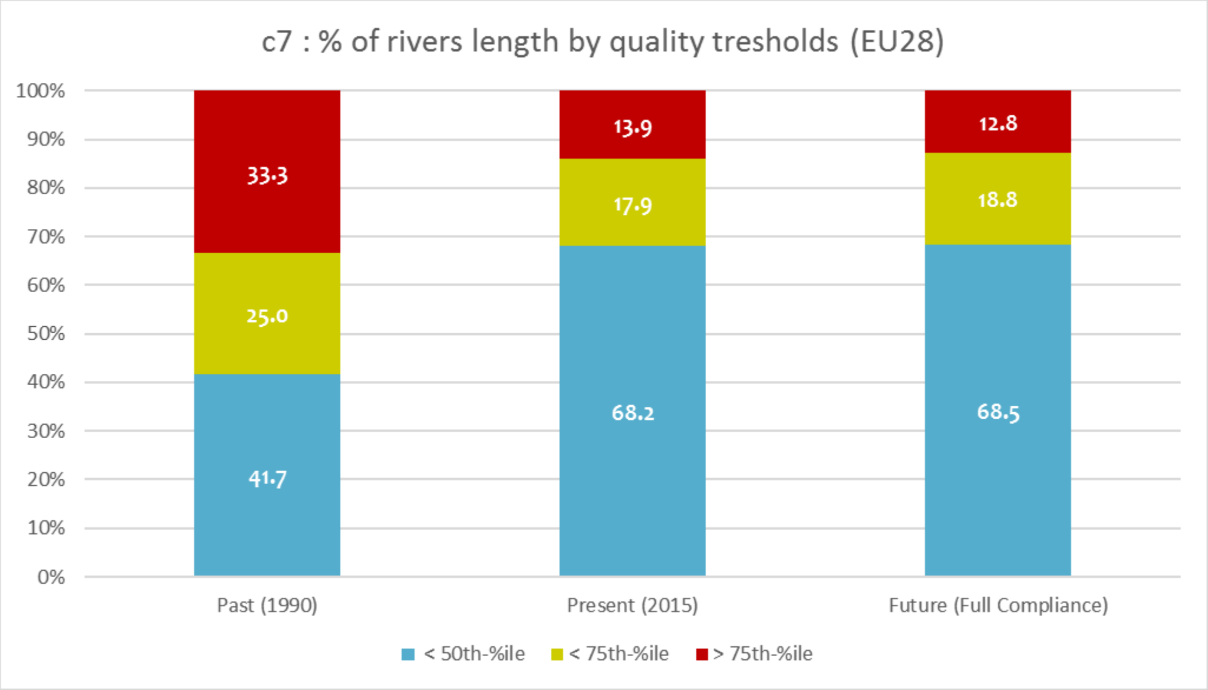


*Figure 35 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#8, e.g. D4, benzene, trichlorobenzenes, chloroform, dichloromethane, 1,2-dichloroethane*

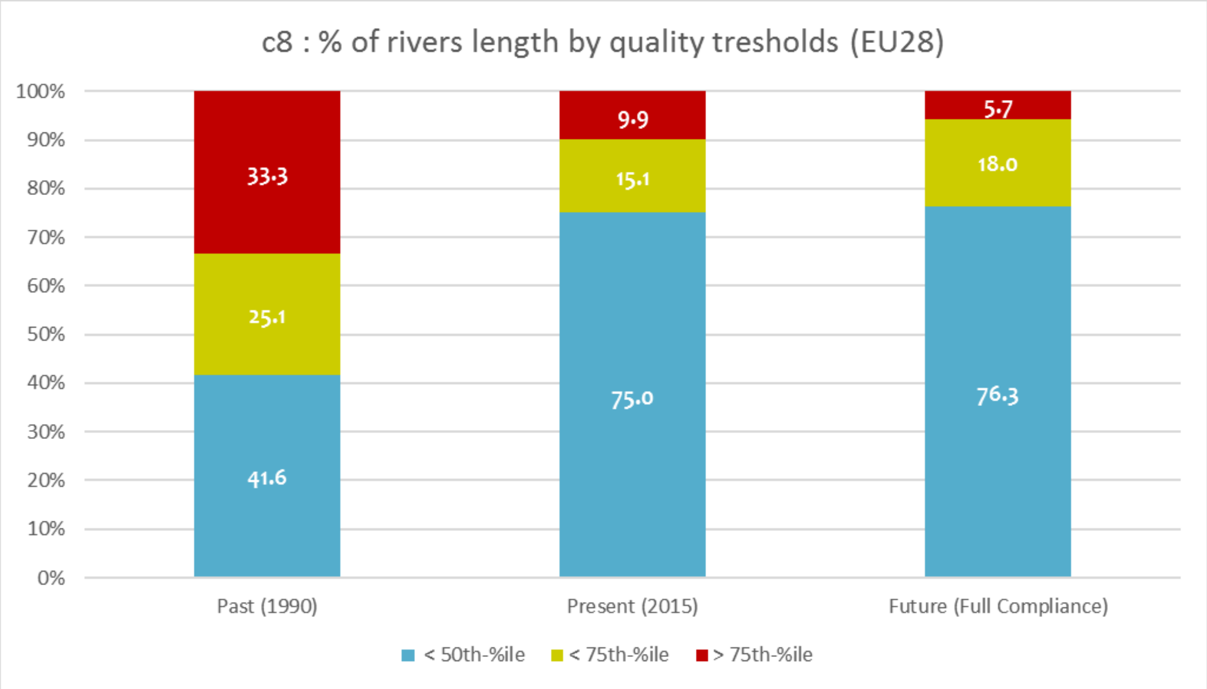



*Figure 36 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#9, e.g. Nonylphenols*

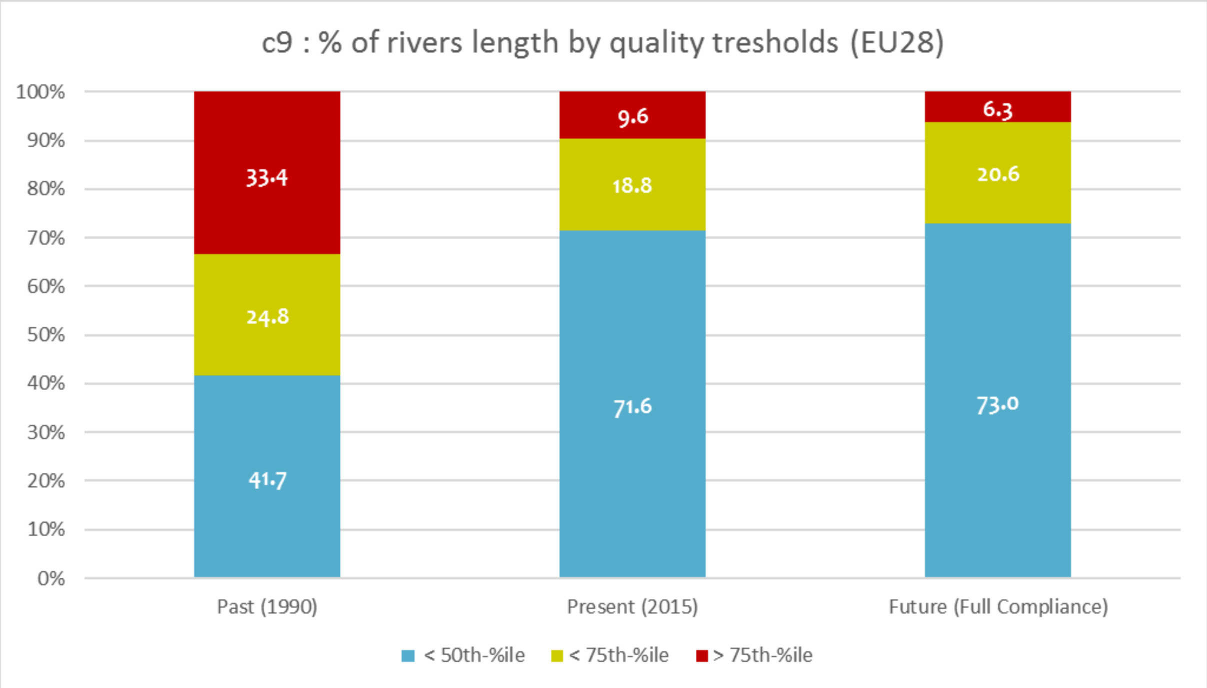


*Figure 37 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#10, e.g.tributyltin cation, brominated diphenylethers, anthracene, fluoranthene, naphthalene, benzo(a)pyrene.*

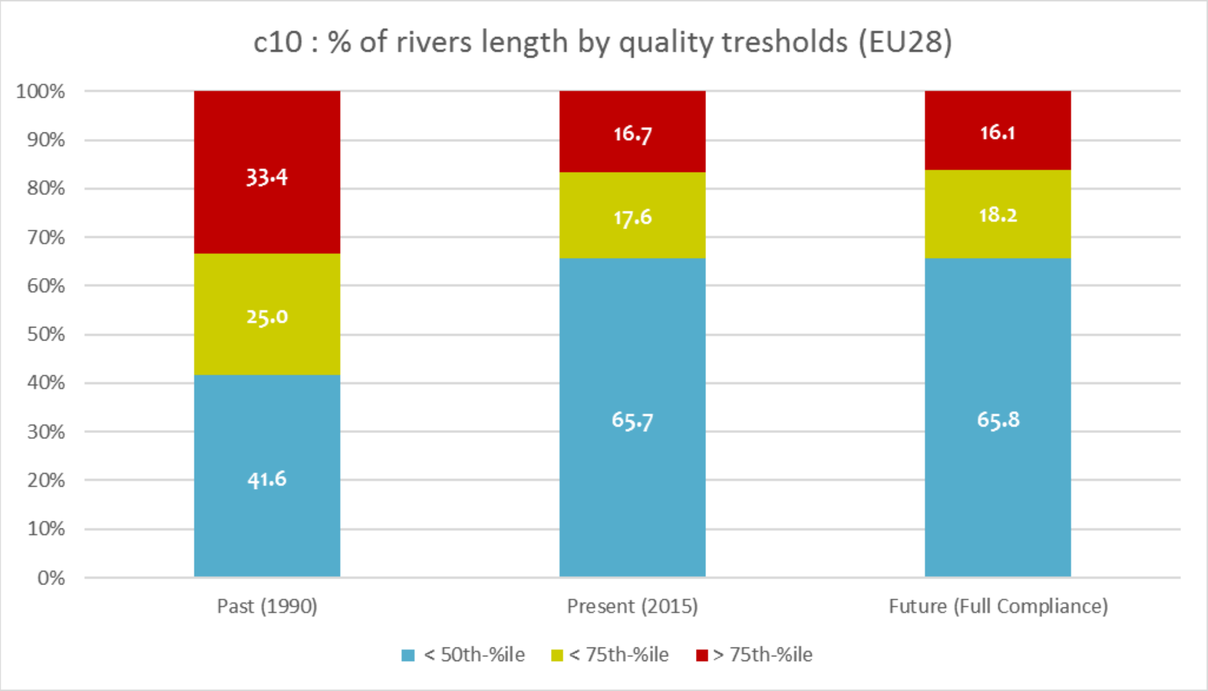



*Figure 38 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#11, e.g. Fluoxetine*

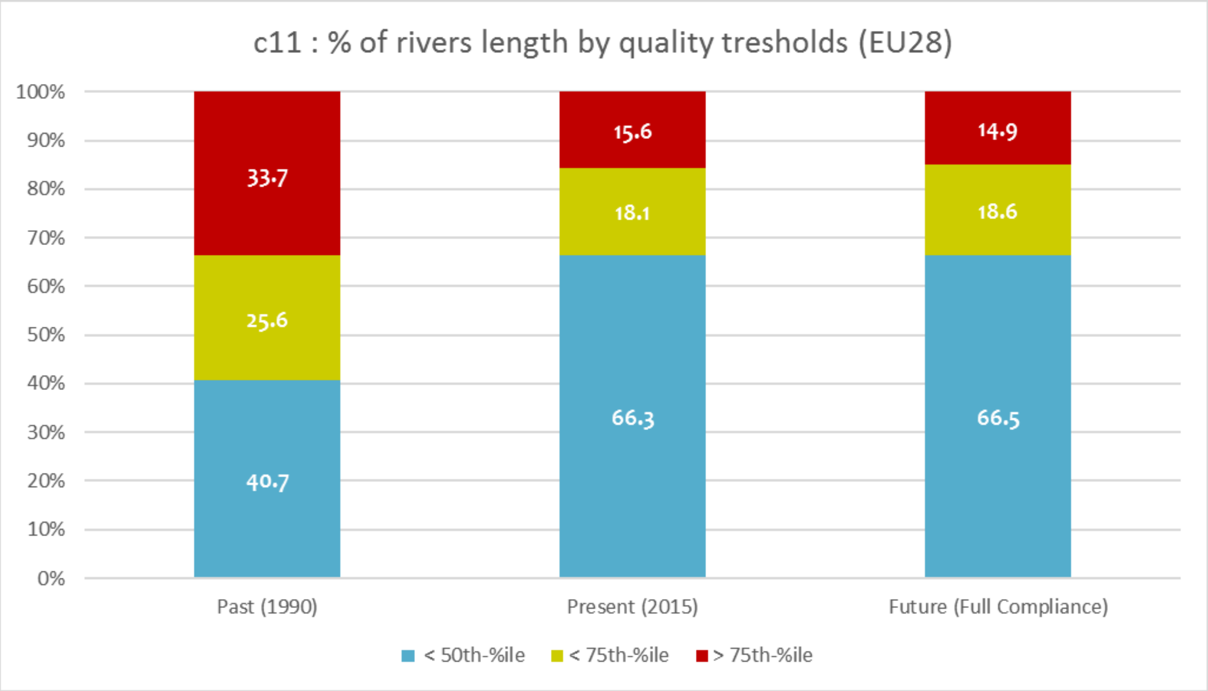


*Figure 39 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#12. No example chemicals considered in this study.*

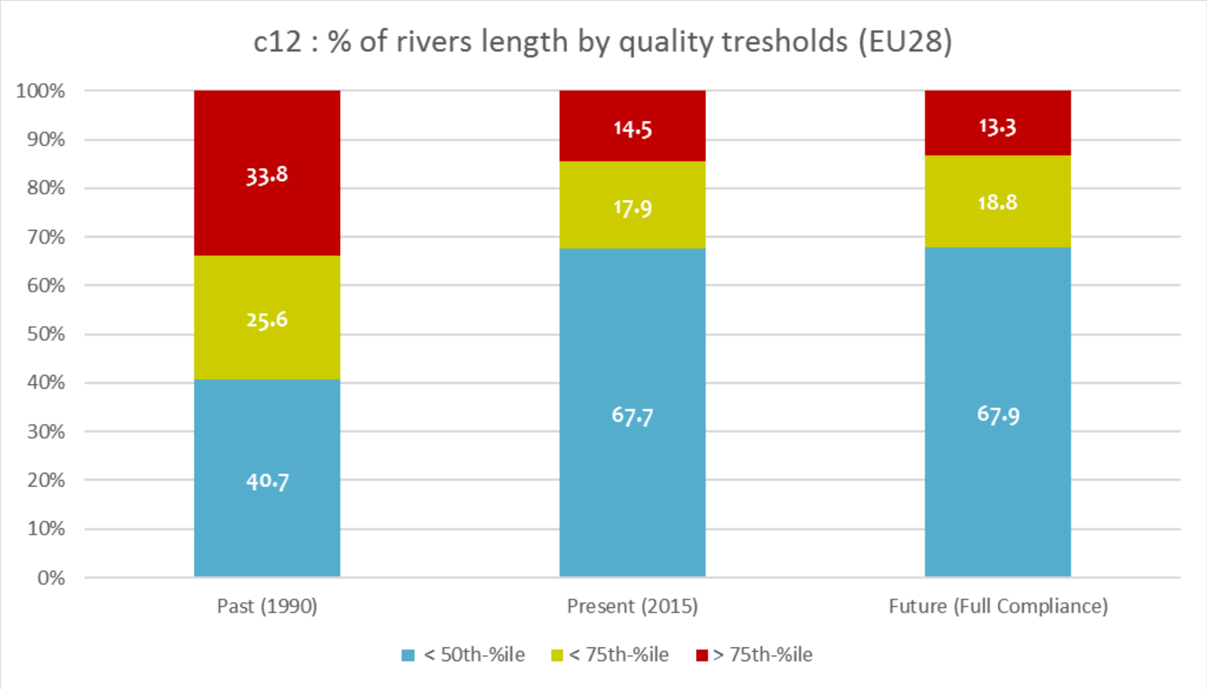
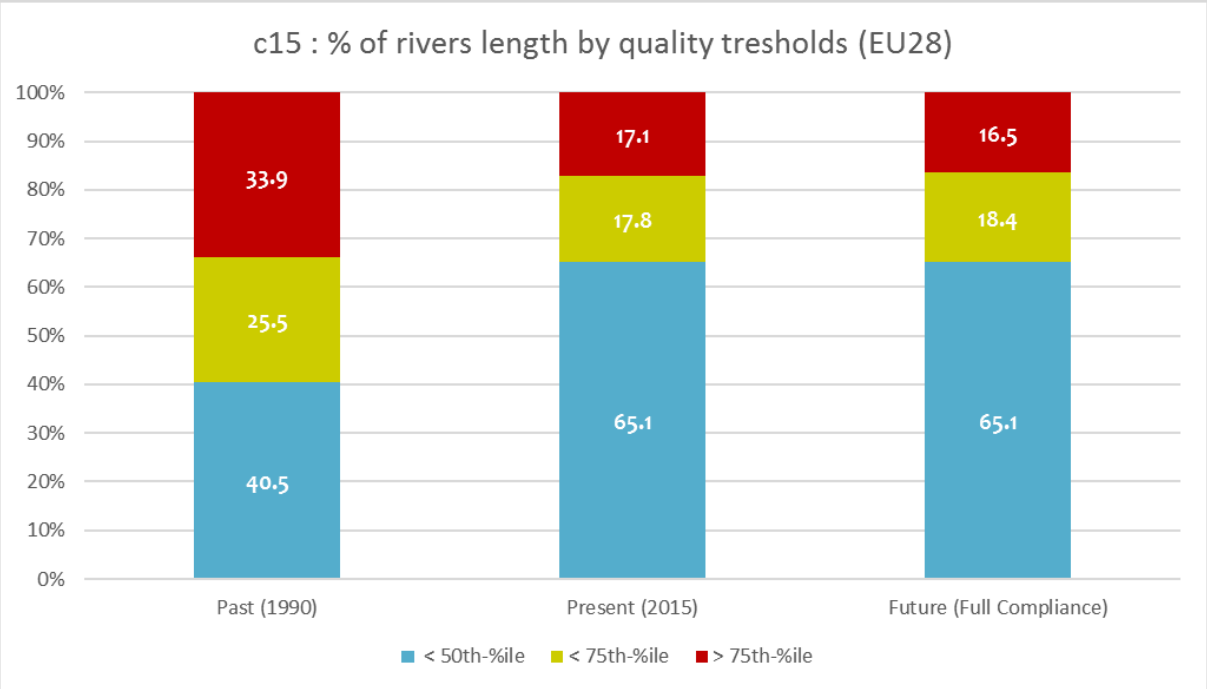



*Figure 40 – reduction of pollution in the European stream network with the implementation of the UWWTD: metachemical c#15, e.g. metals, sertraline, C10-13 chloroalkanes, HBCDD*




The patterns of loads to the European regional seas for the different metachemicals (shown from Figure 41 to Figure 52) [102] appear more complex, but confirm the general trend of pollution in the stream network.

[102] In the graphs, we present loads computed under 2015 and full compliance conditions, both for IAS equivalent to primary treatment (denoted as "IAS nt") , and for IAS equivalent to the level of treatment of the WWTP of the corresponding agglomeration.

It should be noted that, under a pre-directive scenario, a larger part of the population was not connected to sewage networks, hence the loads conveyed to the stream network and seas may have increased under current and full compliance scenarios. This may have occurred to the advantage of reducing local pollution anyway.

*Figure 41 – loads (PE) under different scenarios of metachemical c#2, e.g. 2-Ethylhexyl-4-methylphenol (EHMC)*

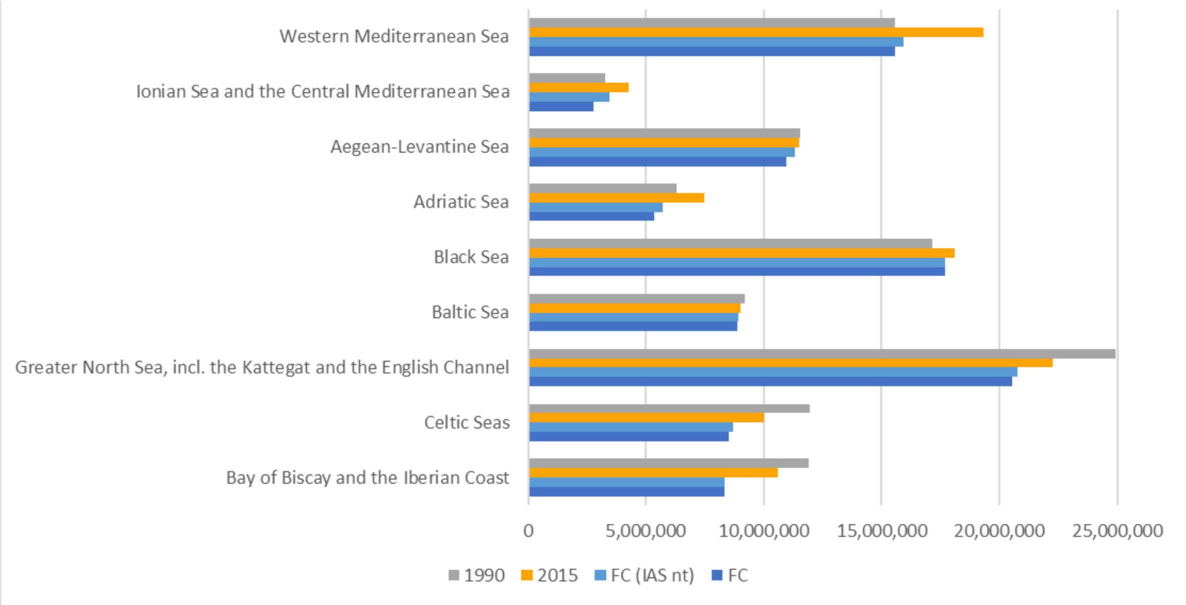



*Figure 42– loads (PE) under different scenarios of metachemical c#3, e.g. estrone (E1), estradiol (E2), Bisphenol A*

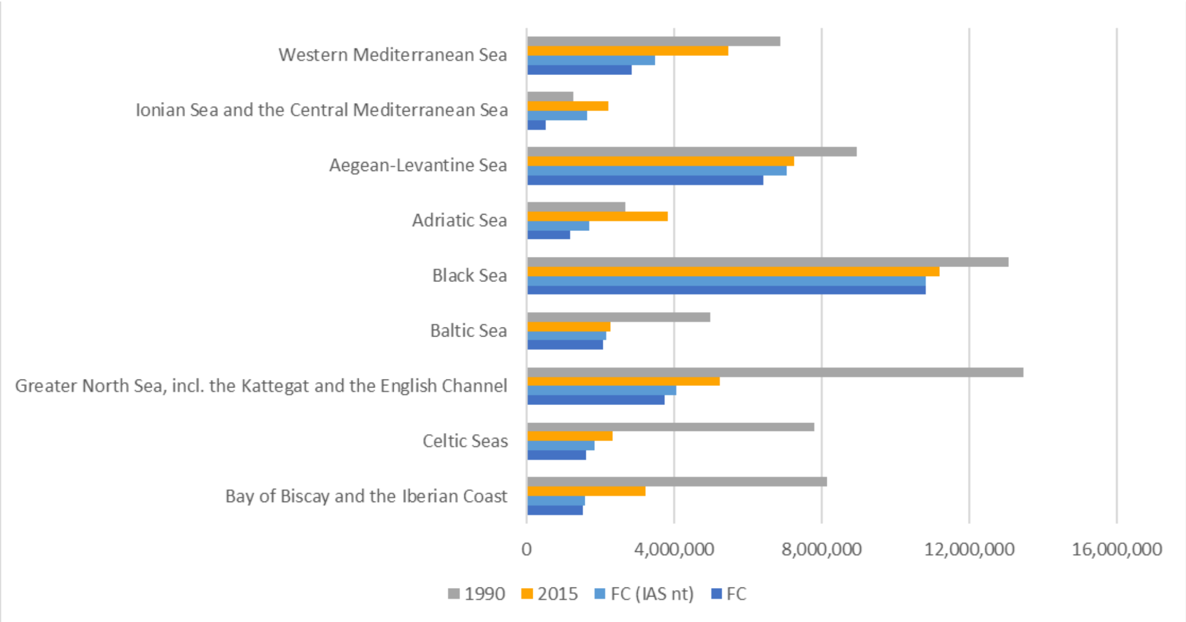


Figure 43– loads (PE) under different scenarios of metachemical c#4. No example chemicals considered in this study.

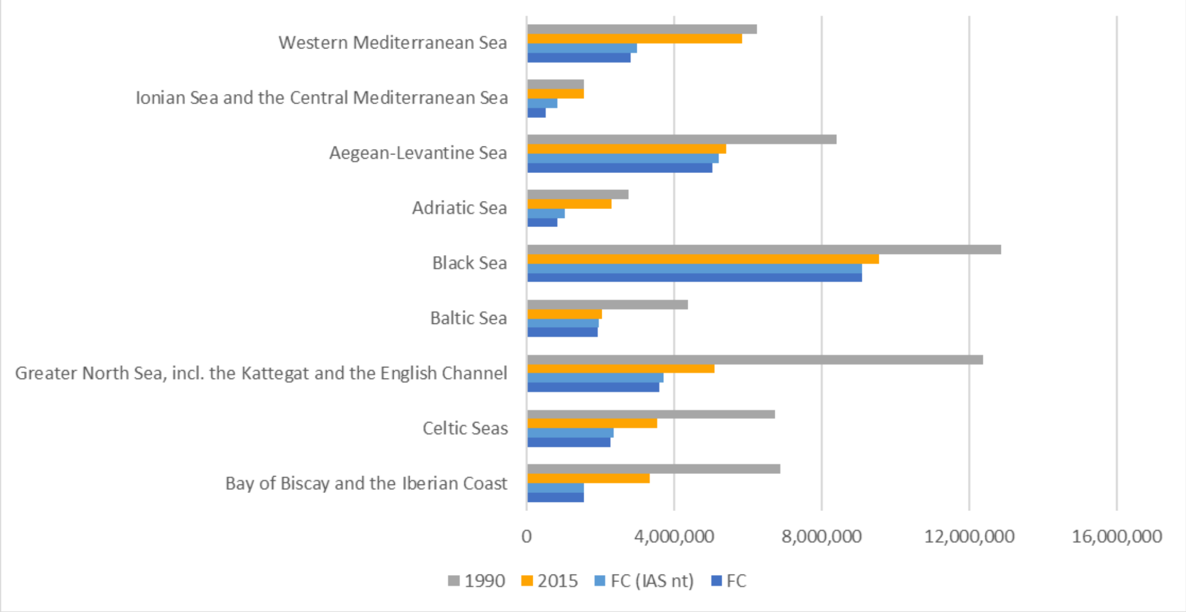



Figure 44– loads (PE) under different scenarios of metachemical c#5, e.g. Triclosan, Ciprofloxacin

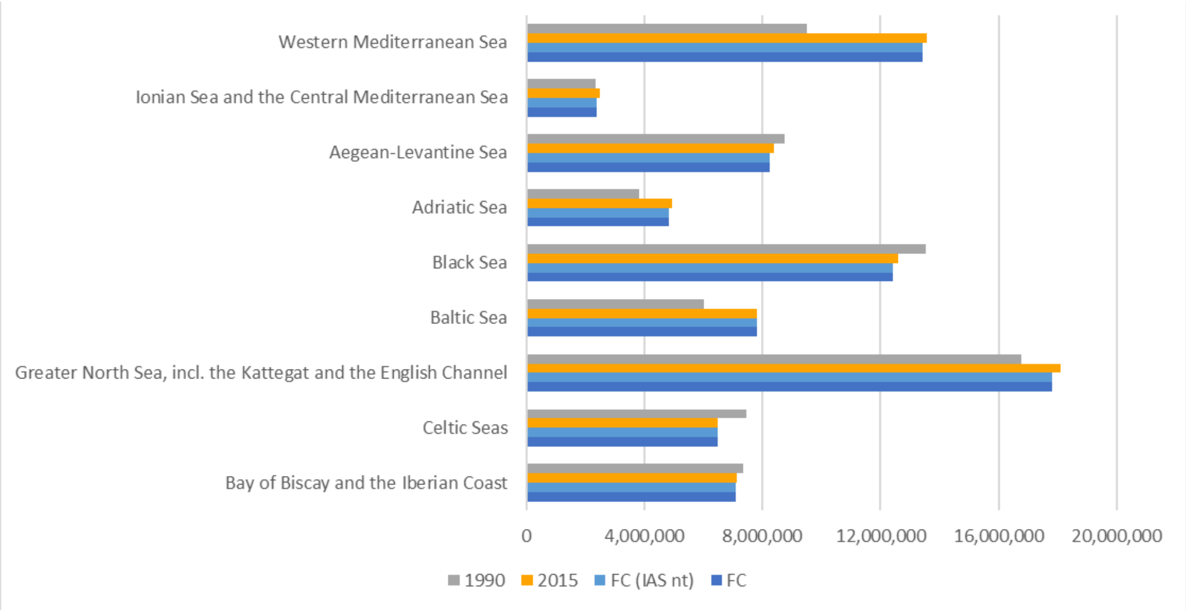



Figure 45– loads (PE) under different scenarios of metachemical c#6, e.g. Diclofenac

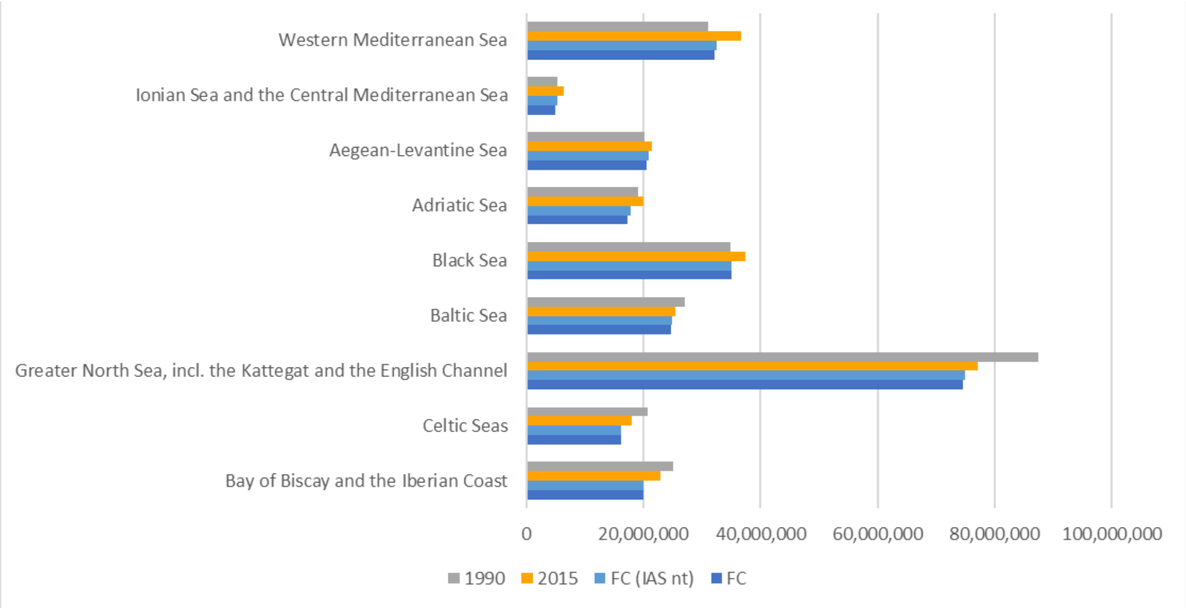


*Figure 46– loads (PE) under different scenarios of metachemical #7, e.g. OP, Acrylamide, ethinylestradiol (EE2), Ibuprofen*

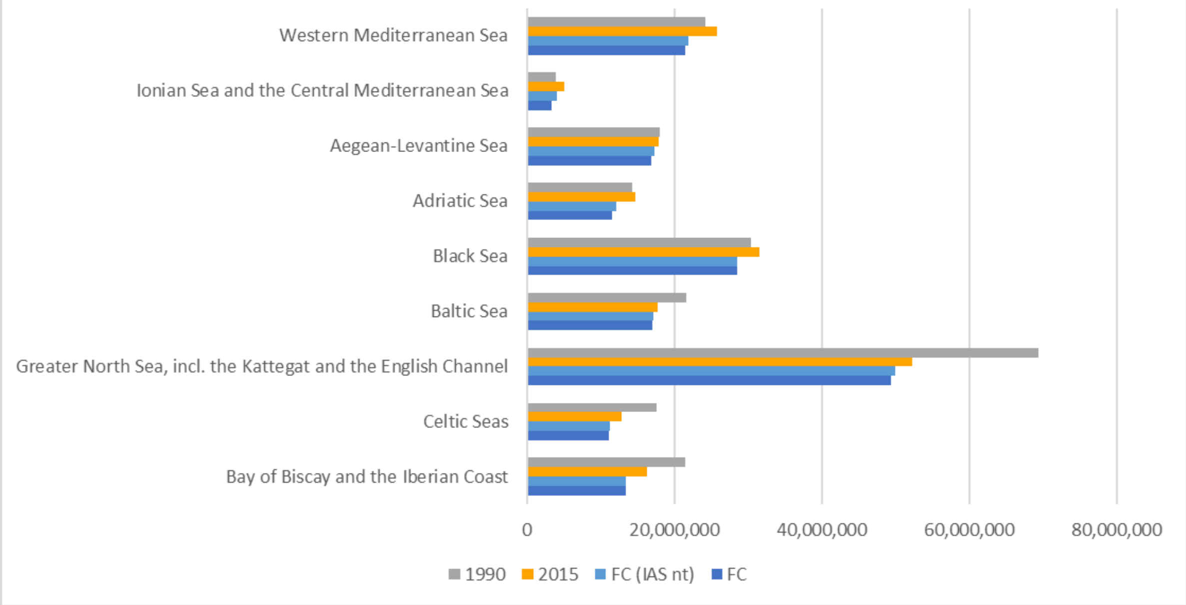



*Figure 47– loads (PE) under different scenarios of metachemical #8, e.g. D4, benzene, trichlorobenzenes, chloroform, dichloromethane, 1,2-dichloroethane*

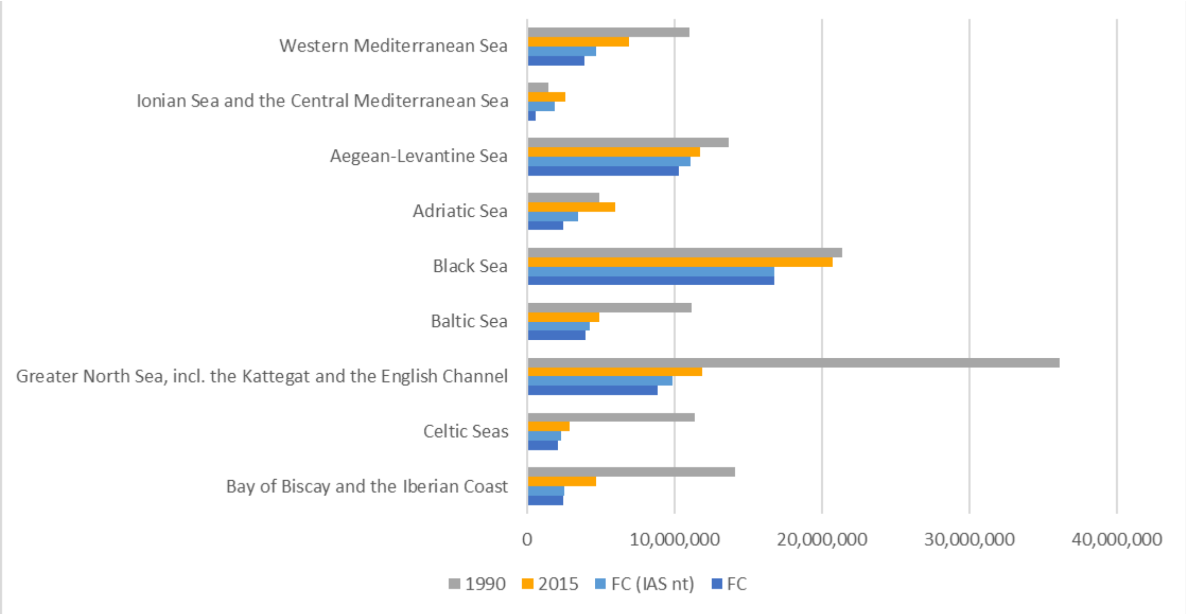



*Figure 48– loads (PE) under different scenarios of metachemical #9, e.g. Nonylphenols*

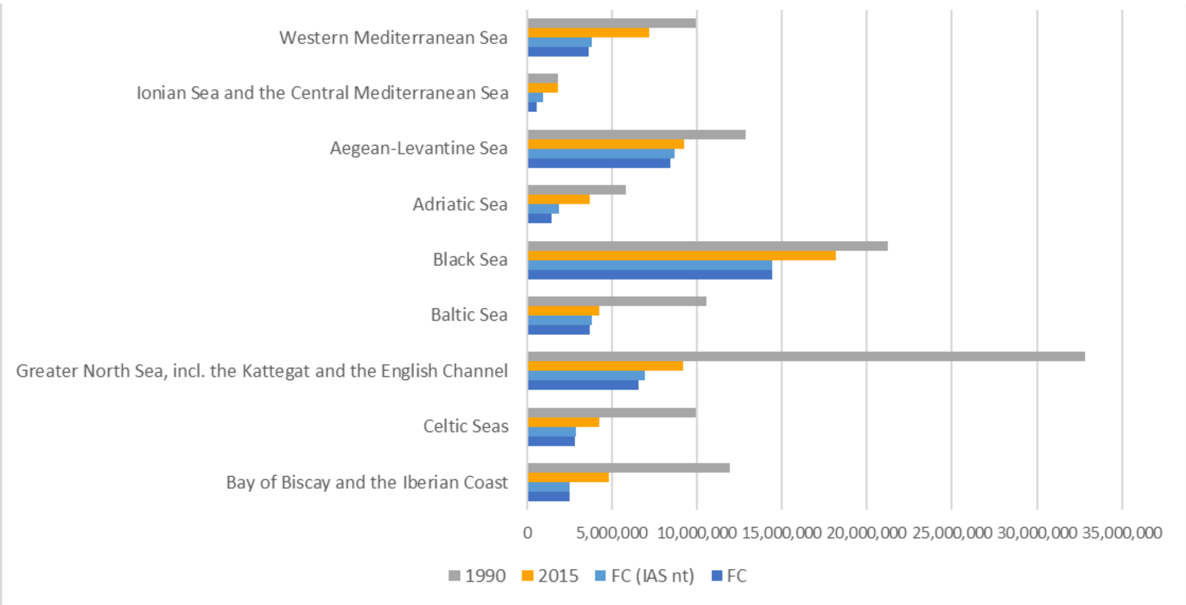


*Figure 49– loads (PE) under different scenarios of metachemical c#10, e.g.tributyltin cation, brominated diphenylethers, anthracene, fluoranthene, naphthalene, benzo(a)pyrene.*

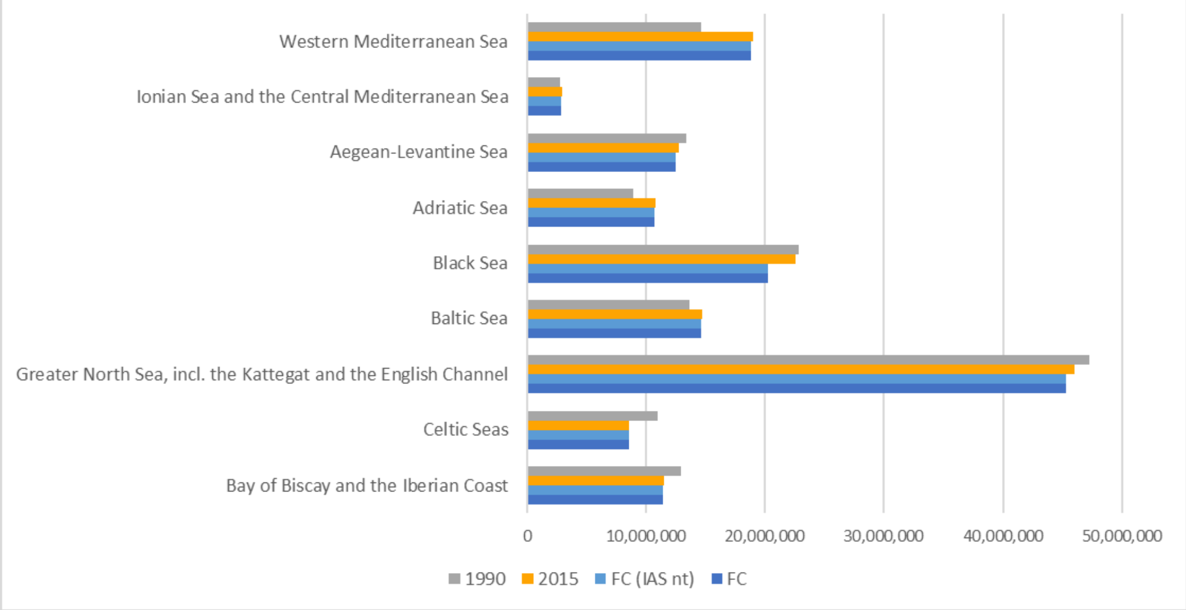



*Figure 50– loads (PE) under different scenarios of metachemical c#11, e.g. Fluoxetine*

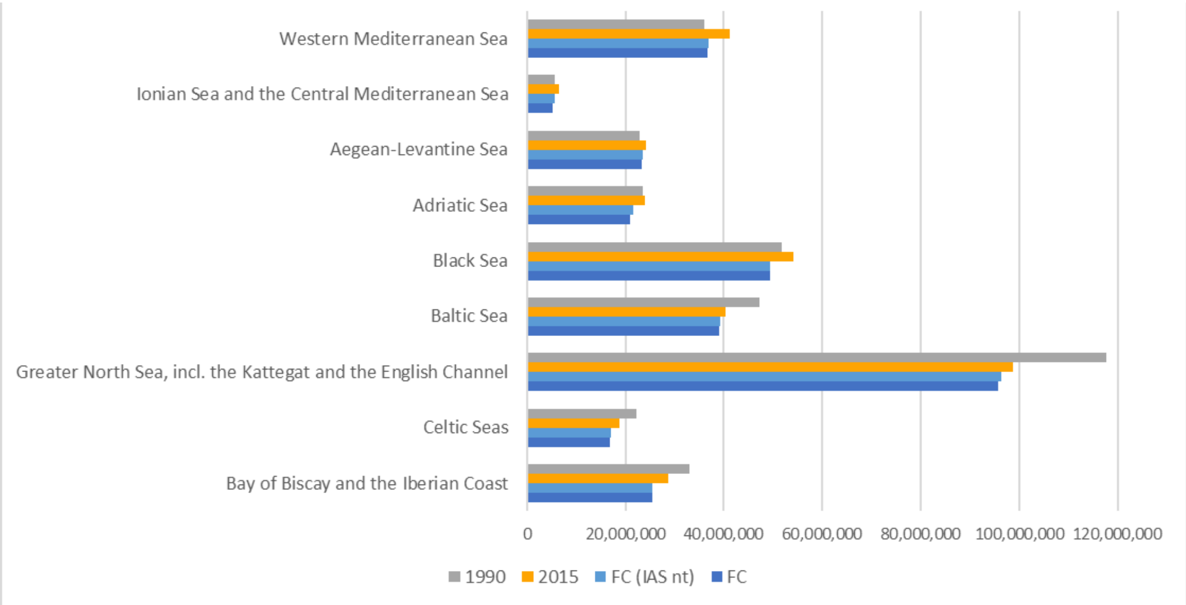



*Figure 51– loads (PE) under different scenarios of metachemical c#12. No example chemicals considered in this study.*

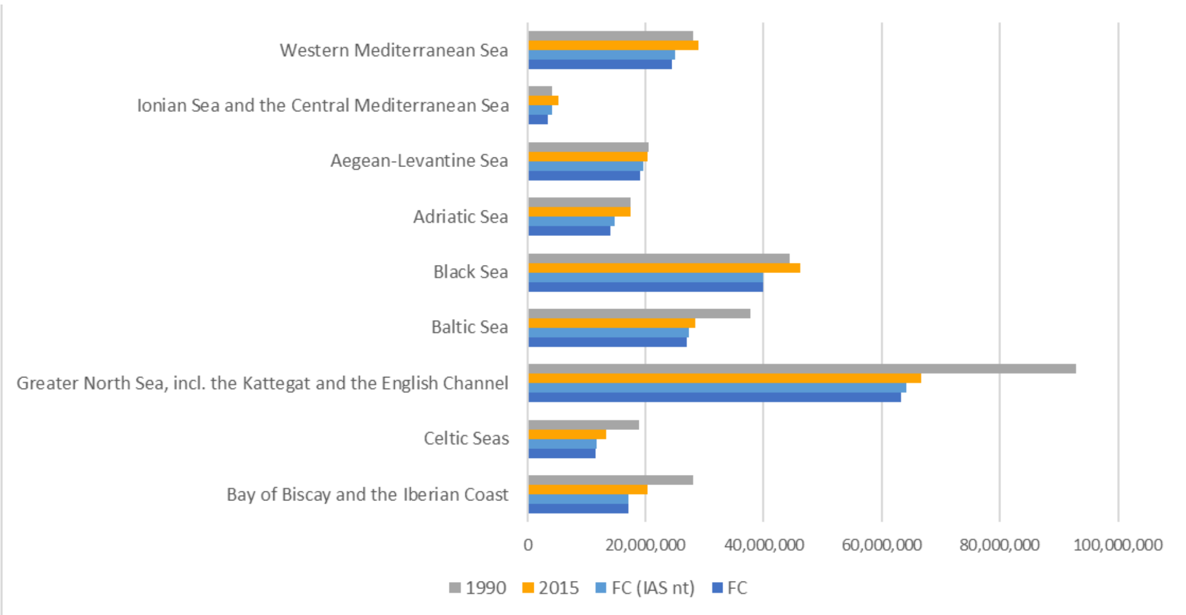
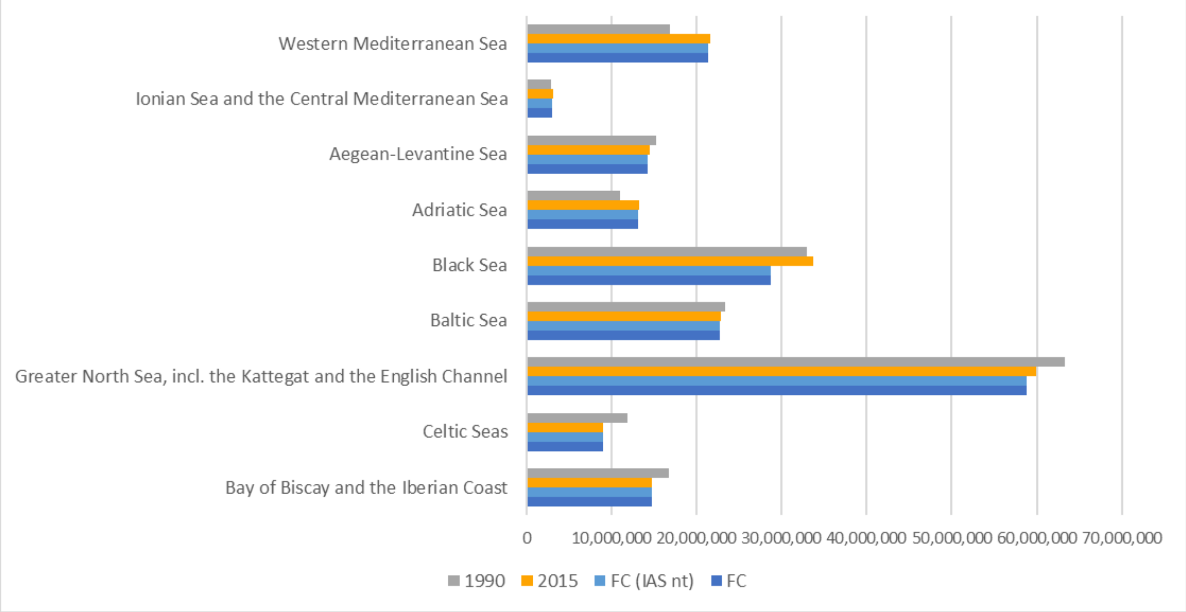


*Figure 52– loads (PE) under different scenarios of metachemical c#15, e.g. metals, sertraline, C10-13 chloroalkanes, HBCDD, DEHP*




## 3.5 Transfer of chemical pollution to soils via sludge

It should be stressed that, for chemicals significantly retained in sludge, wastewater treatment may cause a partial shift of pollution from water to soils, in case sludge is used as such, or after composting, on soils themselves. The percentage of a chemical entering the wastewater treatment plant which remains in the sludge depends on the properties of the chemical. The modelling of removal shown for selected chemicals in Table 11 highlights that many molecules may end up in sludge by a significant percentage (Figure 53). Examples of such substances include pharmaceuticals Ciprofloxacin and Sertraline, antibacterial Triclosan, metals, and Nonylphenols (NP).

The chemicals present in the sludge end up in soils depending on the mode of disposal or reuse of the sludge itself, which is highly variable in the EU[103]. Some countries, such as NL and DE, make no agricultural application of sludge while, for many of the others, sludge reuse on soils is prevalent. Assuming no transboundary transfer of sludge occurs in the EU, Figure 54 shows a computed potential load of contaminants to agricultural soils via sludge application, considering the amount of sludge reported as used in agriculture and other reuses, divided by total crop- and grassland area of the country. It should be stressed that this calculation neglects the fact that sludge is typically applied within a short (e.g. 10 km) distance from WWTPs, hence the average application rate for one country does not necessarily reflect the higher doses on soil near WWTPs and the virtual absence of sludge application on distant soils.

[103] In our illustrative calculations, whose results are presented in Figure 54, we refer to the data provided by Member States and used for the UWWTD's 9th Implementation report: http://ec.europa.eu/environment/water/water-urbanwaste/implementation/implementationreports_en.htm

*Figure 53 – percentage of the pollutant load incoming to a WWTP, which is retained in sludge, based on the assumed properties of chemicals presented in Table 11.*

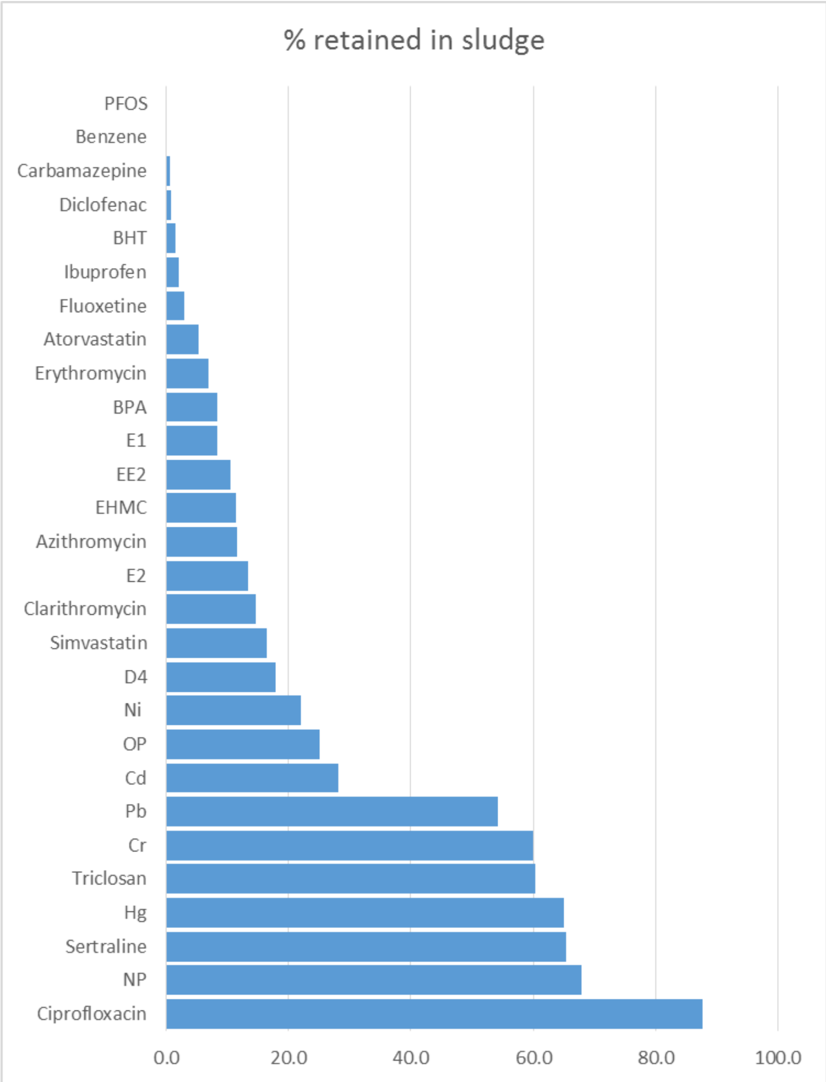
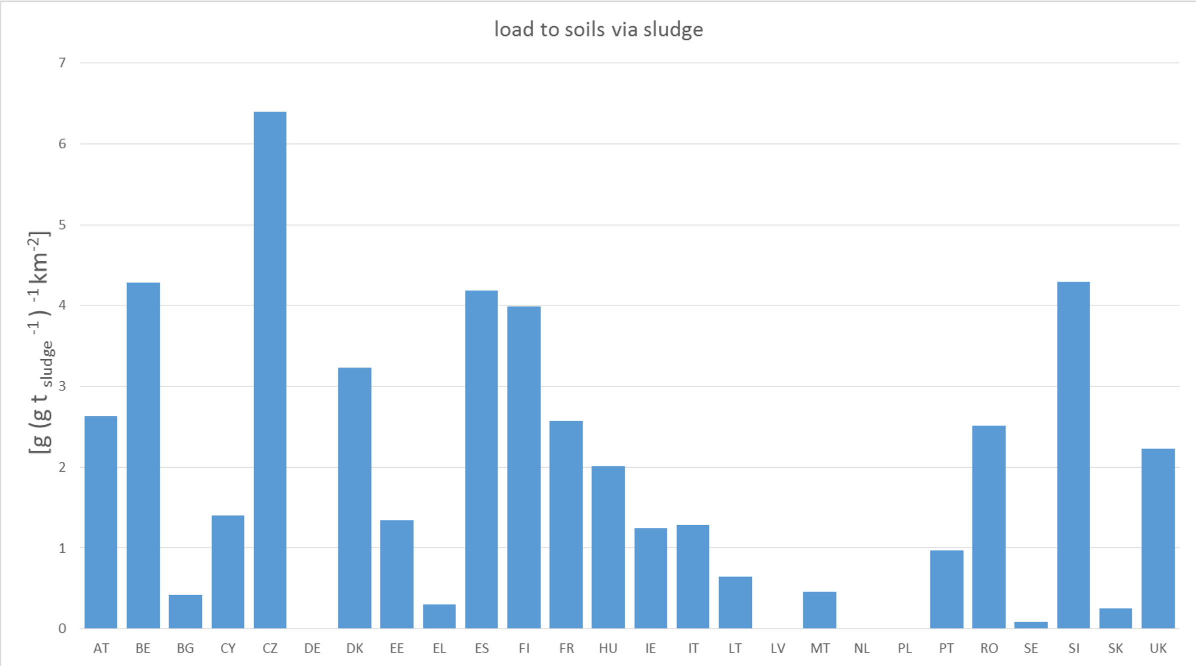


*Figure 54 – potential load of contaminants to soils in the different EU Member States. The numerical values of the y-axis are computed as the tonnes of sludge applied in agriculture in each country, divided by the agricultural area in the country. They can be interpreted as the load of contaminants (g km$^{-2}$y$^{-1}$) for a concentration of contaminants in sludge equal to 1 g t$^{-1}$.*



## 3.6 The potential impact of CSO

Loads from CSO can be evaluated in terms of PE, and compared to the PE multiplied by (1-treatment efficiency) under full compliance (i.e. the PE of treated wastewater which is discharged to the receiving water bodies). A summary of this comparison by country is presented in Figure 55. CSO may account for a load of untreated wastewater corresponding to several tens of thousands to hundreds of thousands PE in the various regions (Figure 56). Under a full compliance scenario, these loads are usually relatively small for certain chemicals and for total N in comparison to loads from treated wastewater, while they may represent more significant proportions (in some cases 100% or more) for coliforms, BOD, to some extent total P, and those chemicals undergoing significant removal in WWTPs (see Figure 57 to Figure 65). These figures indicate CSO may assume a certain relevance in comparison with treated wastewater when the UWWTD will be fully implemented. It should be stressed once more, though, that the present estimate is a first and highly uncertain attempt, likely overestimating loads (even by a large extent) in several parts of the EU.

It should be stressed that the above representation of CSO accounts only for the transfer of wastewater from treatment to direct discharge, and consequently does not include the impact of urban runoff itself. The latter is known to be often laden with significant pollution due to the washout of urban surfaces and drainage network pipes, where contaminants may build up during dry weather between storm events.

The management measures in place in the different EU countries may significantly reduce the loads of pollutants associated to CSO. In the following figures, we show for comparison how the load from CSO can be reduced by measures in place, taking into account the reduction coefficients computed in § 2.4 (see Annex III).

*Figure 55 – loads through CSO as a percentage of the load generated from WWTPs in the different countries*

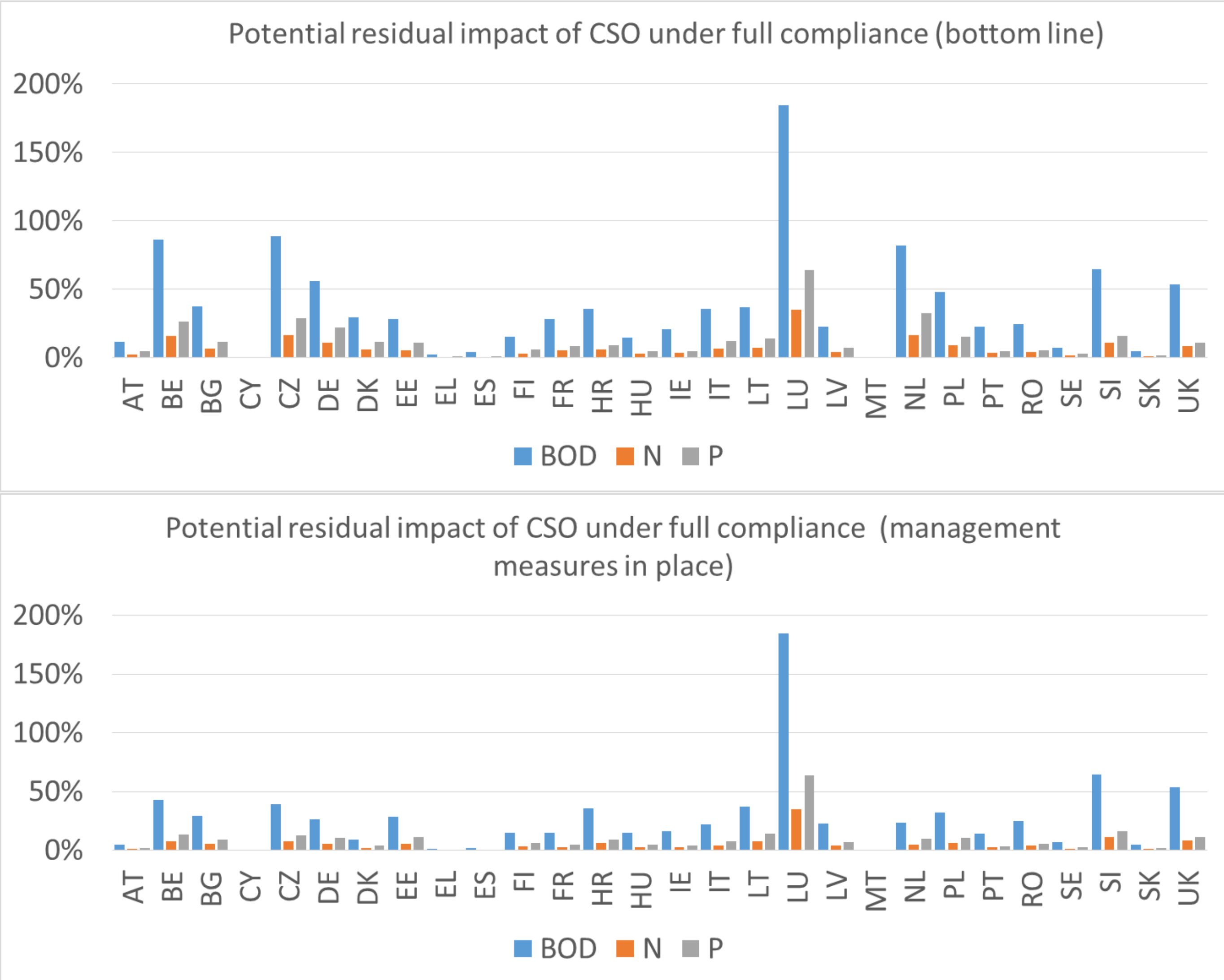


*Figure 56- Potential loads to receiving water bodies: PE of CSO assuming d=4 (below) and with correction factors accounting for management (above)*

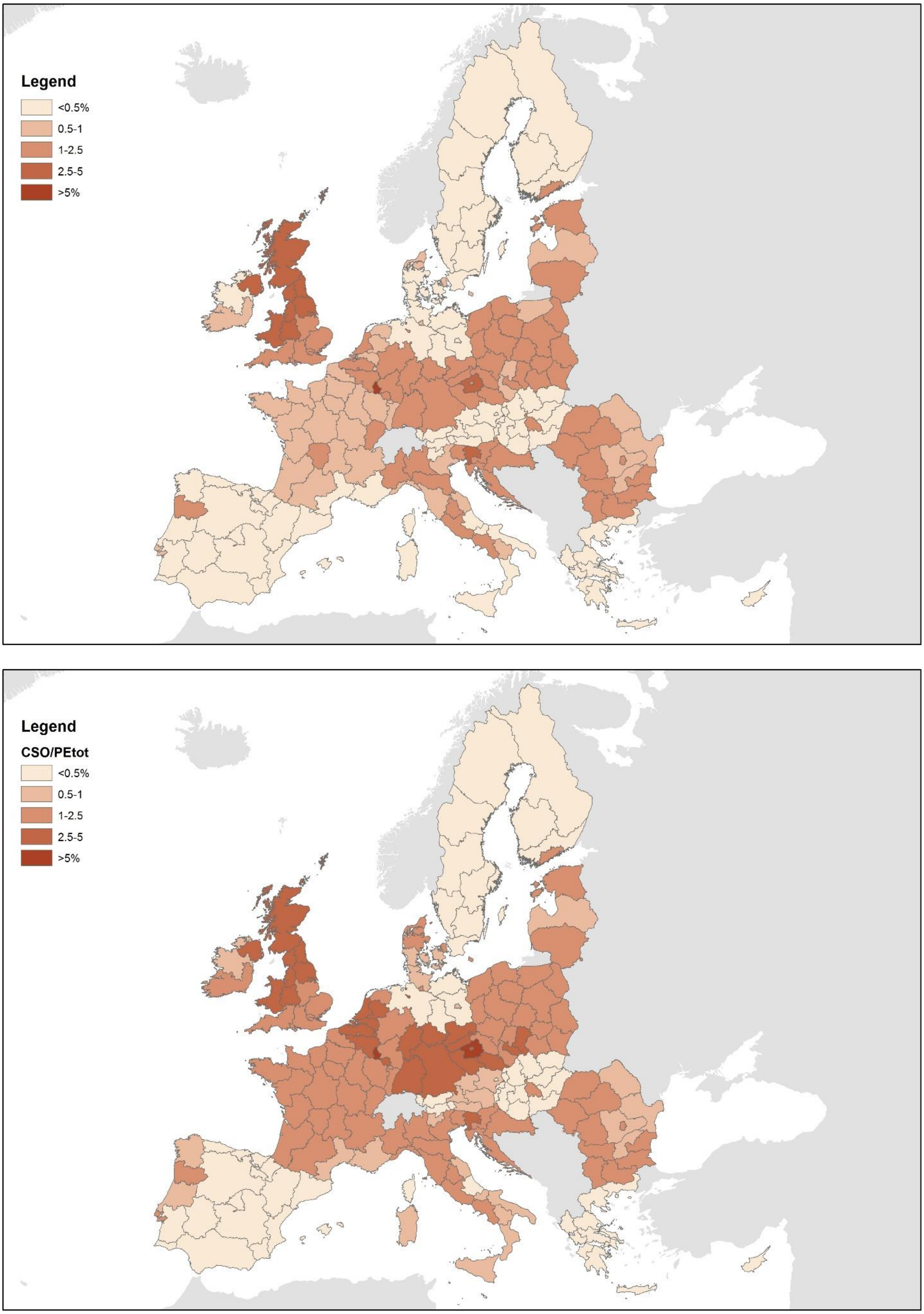
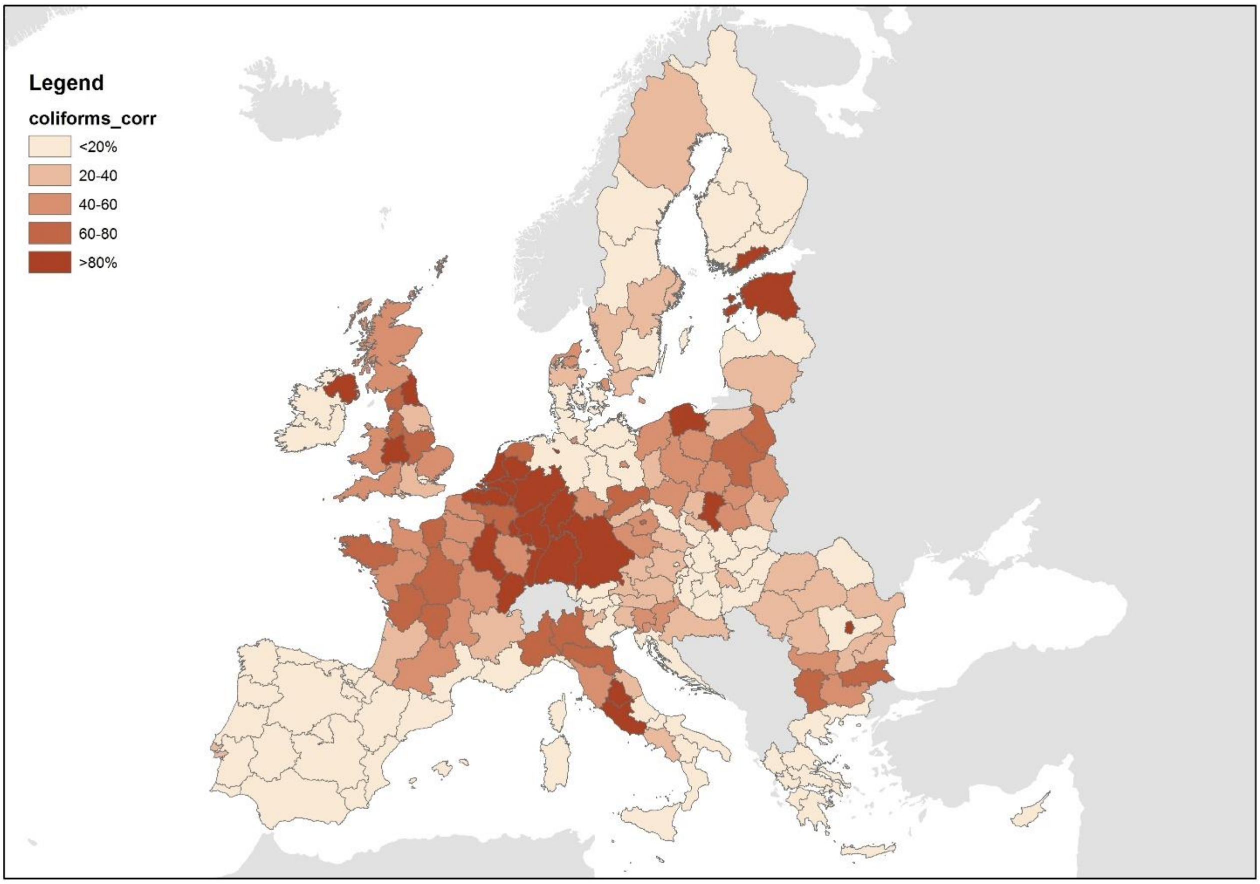


*Figure 57- Potential CSO as a percentage of dry-weather wastewater under full compliance (coliforms) assuming d=4 (below) and with correction factors accounting for management (above)*



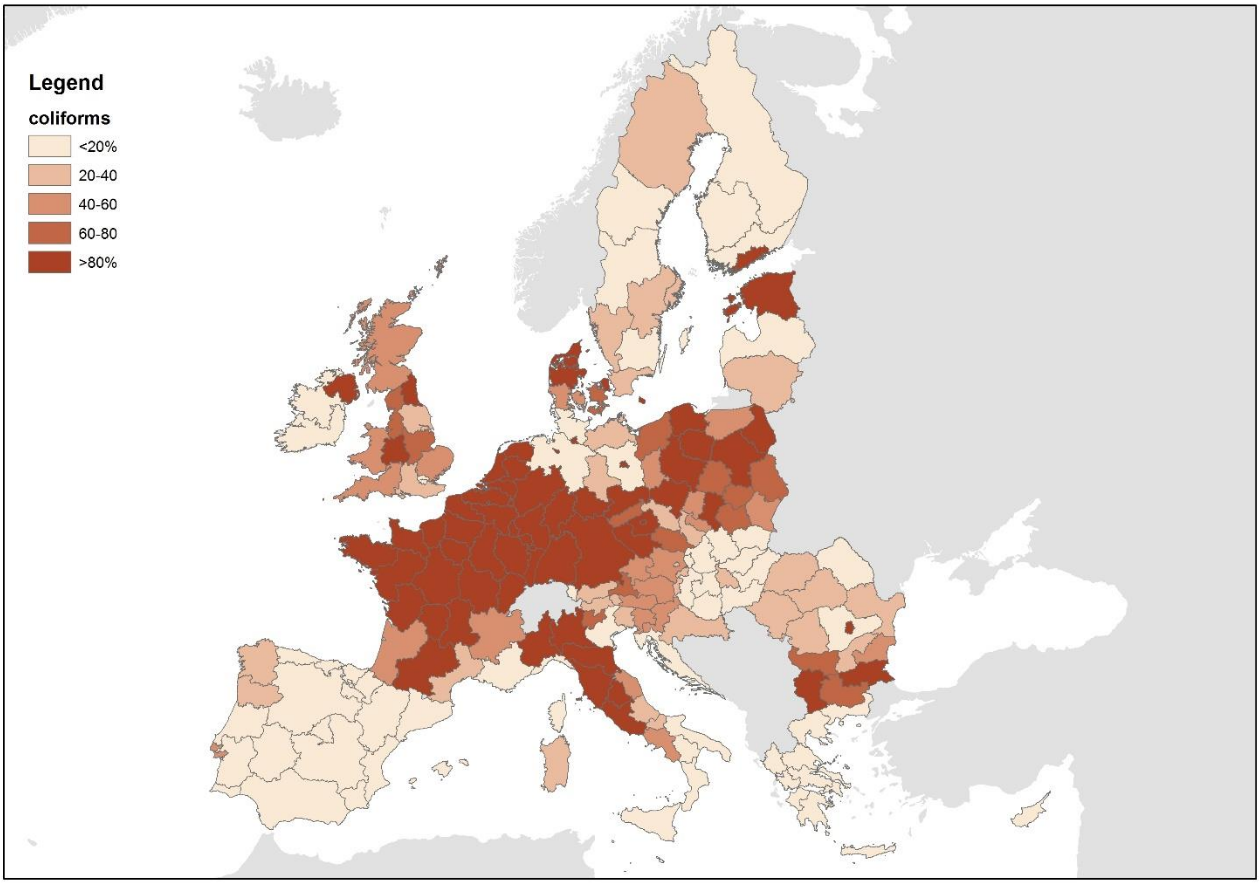
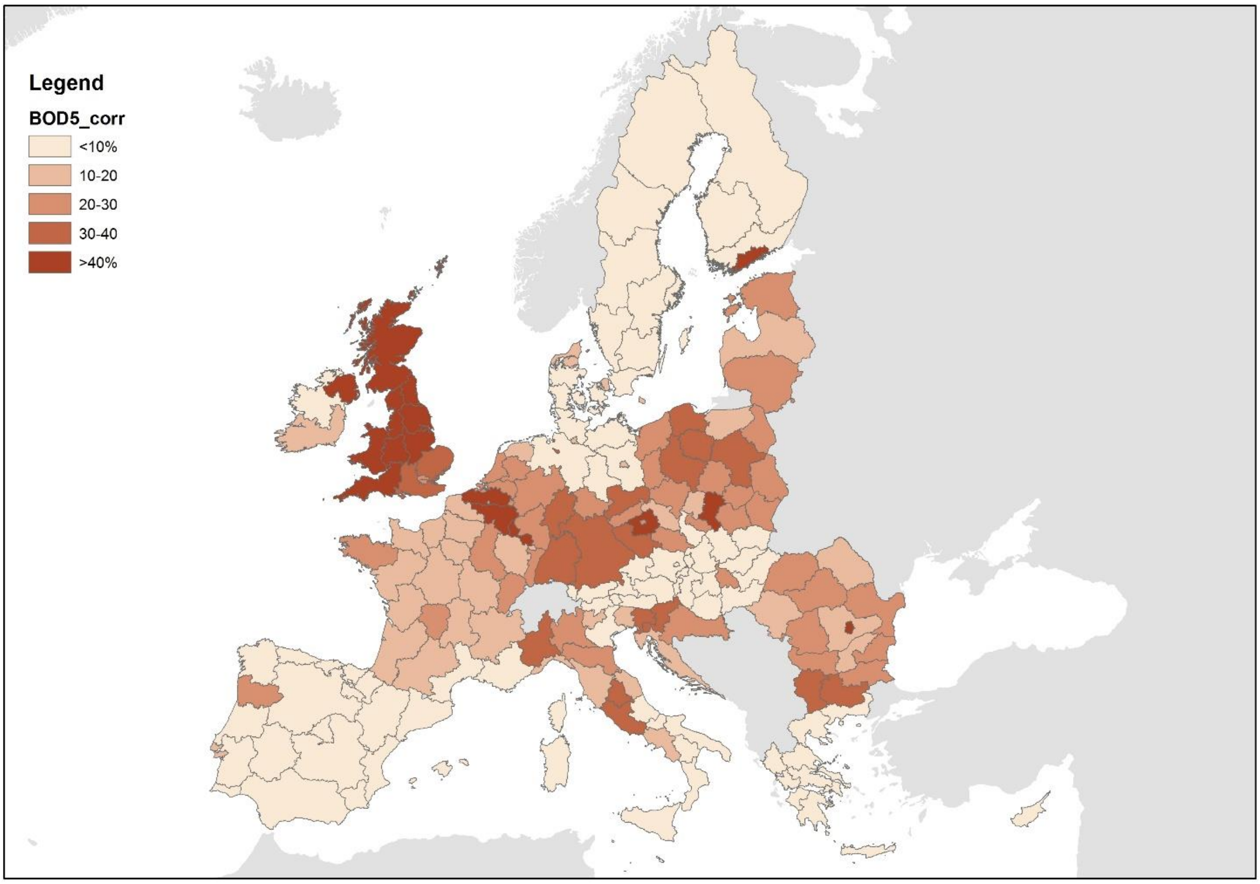



*Figure 58- Potential CSO as a percentage of dry-weather wastewater under full compliance (BOD) assuming d=4 (below) and with correction factors accounting for management (above)*



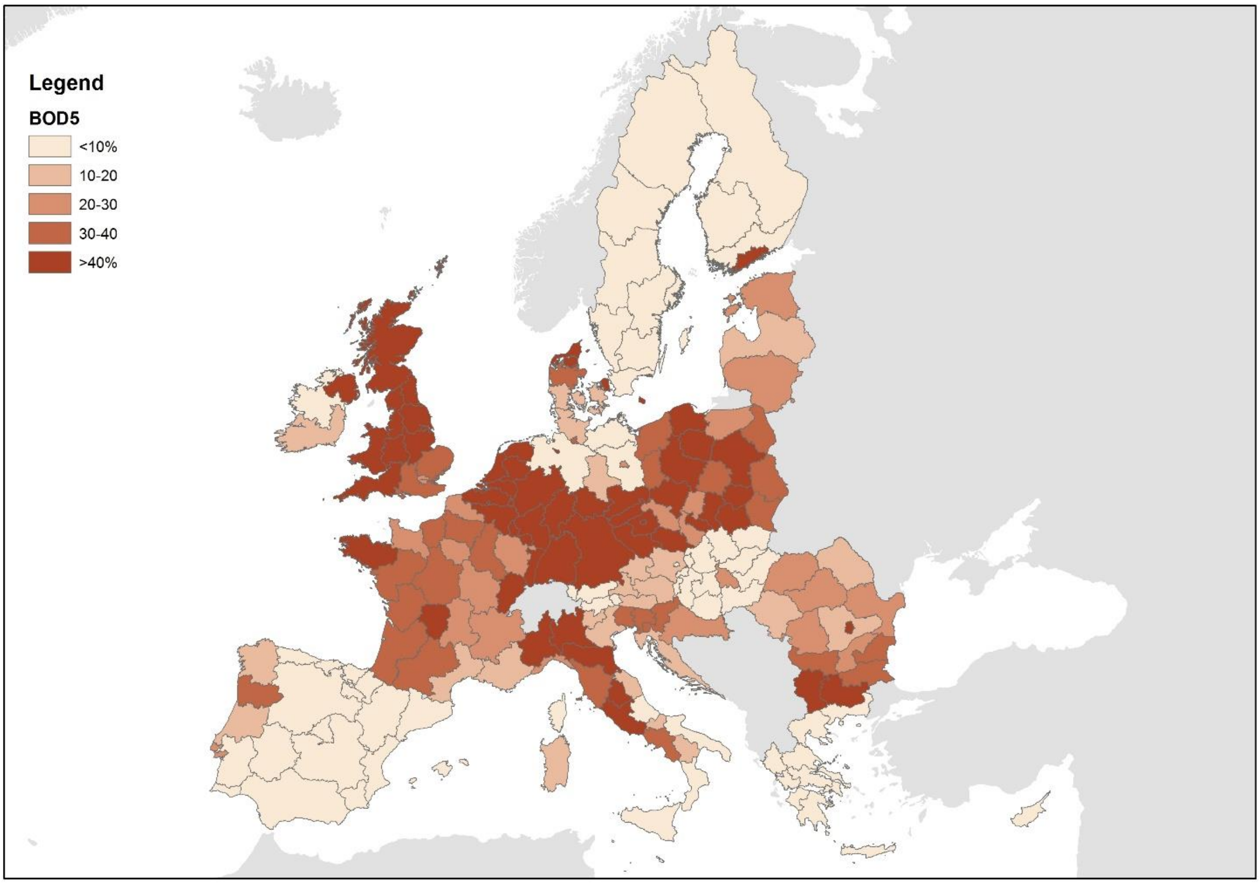
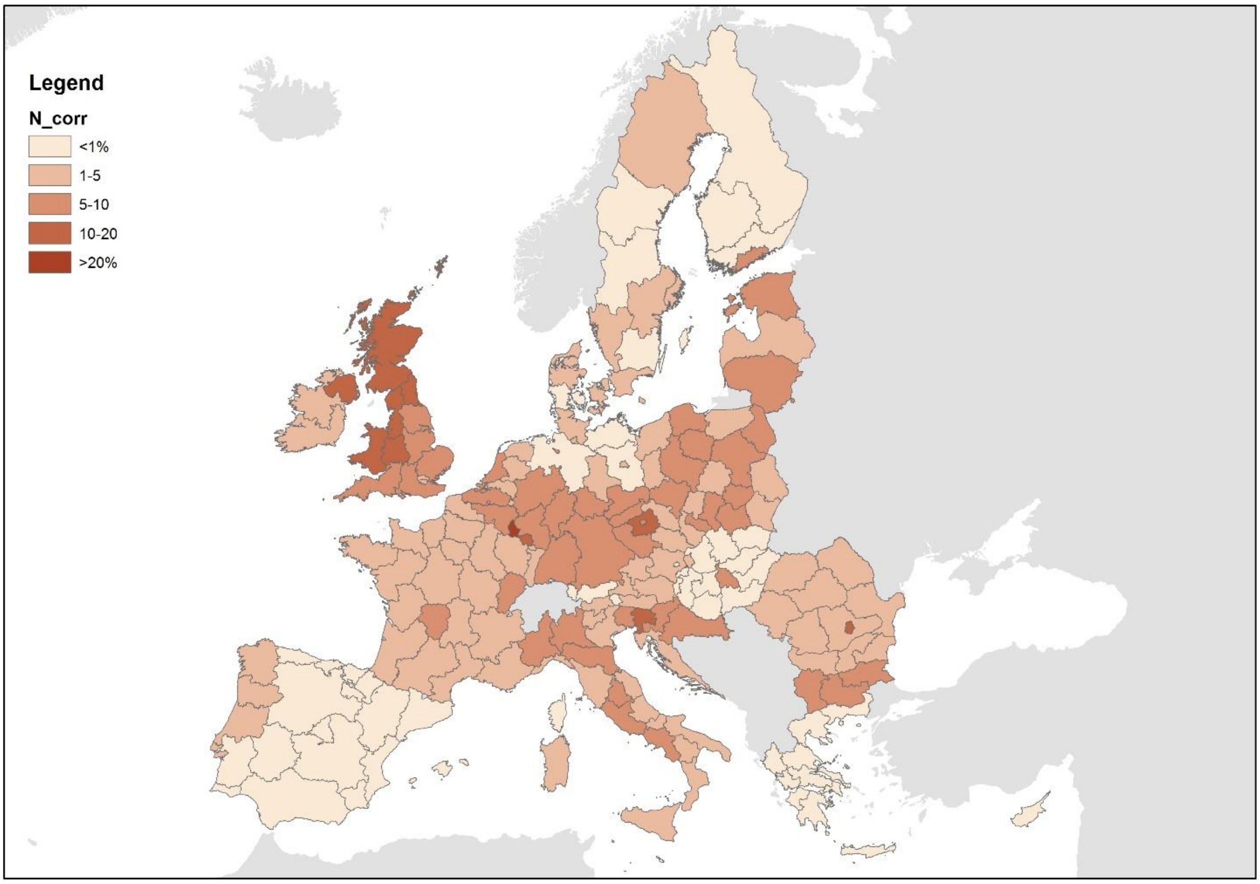



*Figure 59- Potential CSO as a percentage of dry-weather wastewater under full compliance (N) assuming d=4 (below) and with correction factors accounting for management (above)*



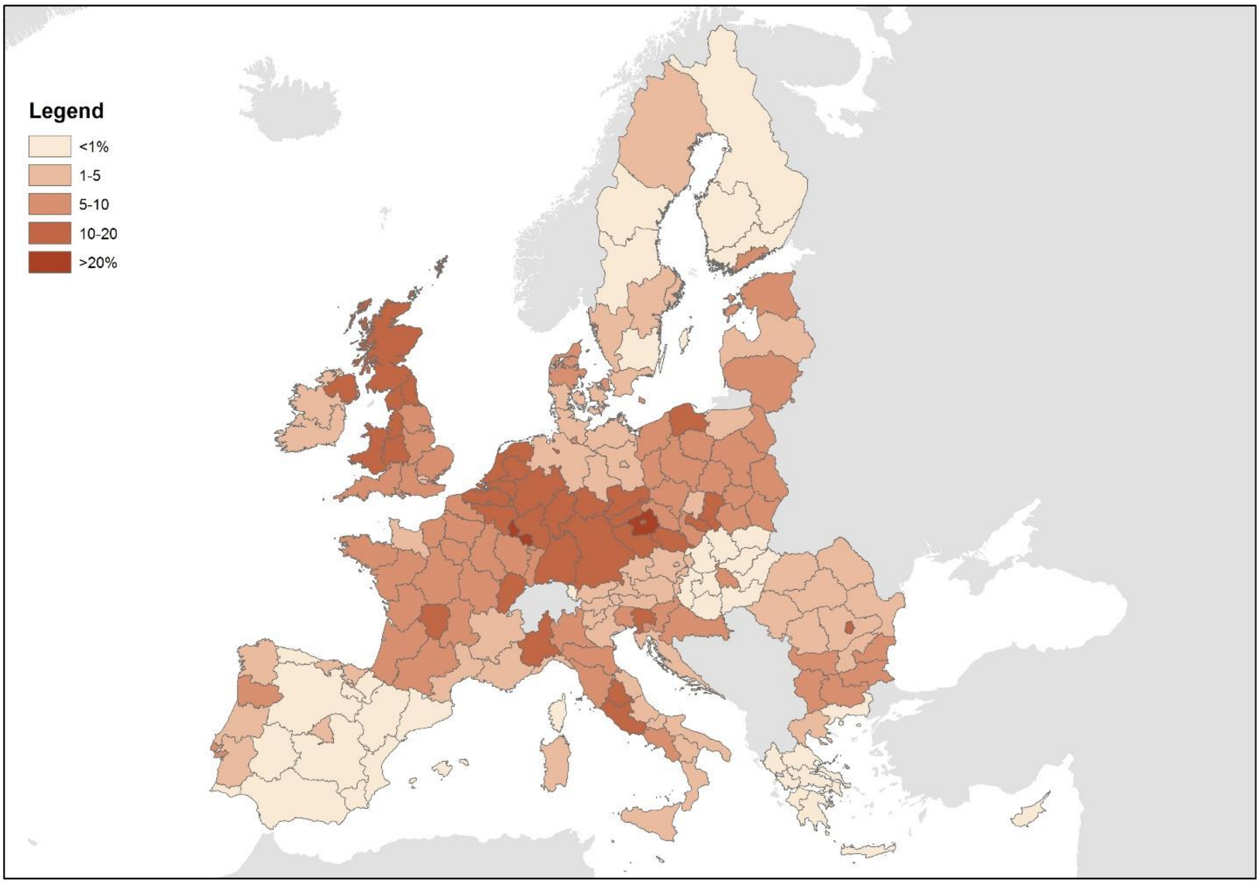
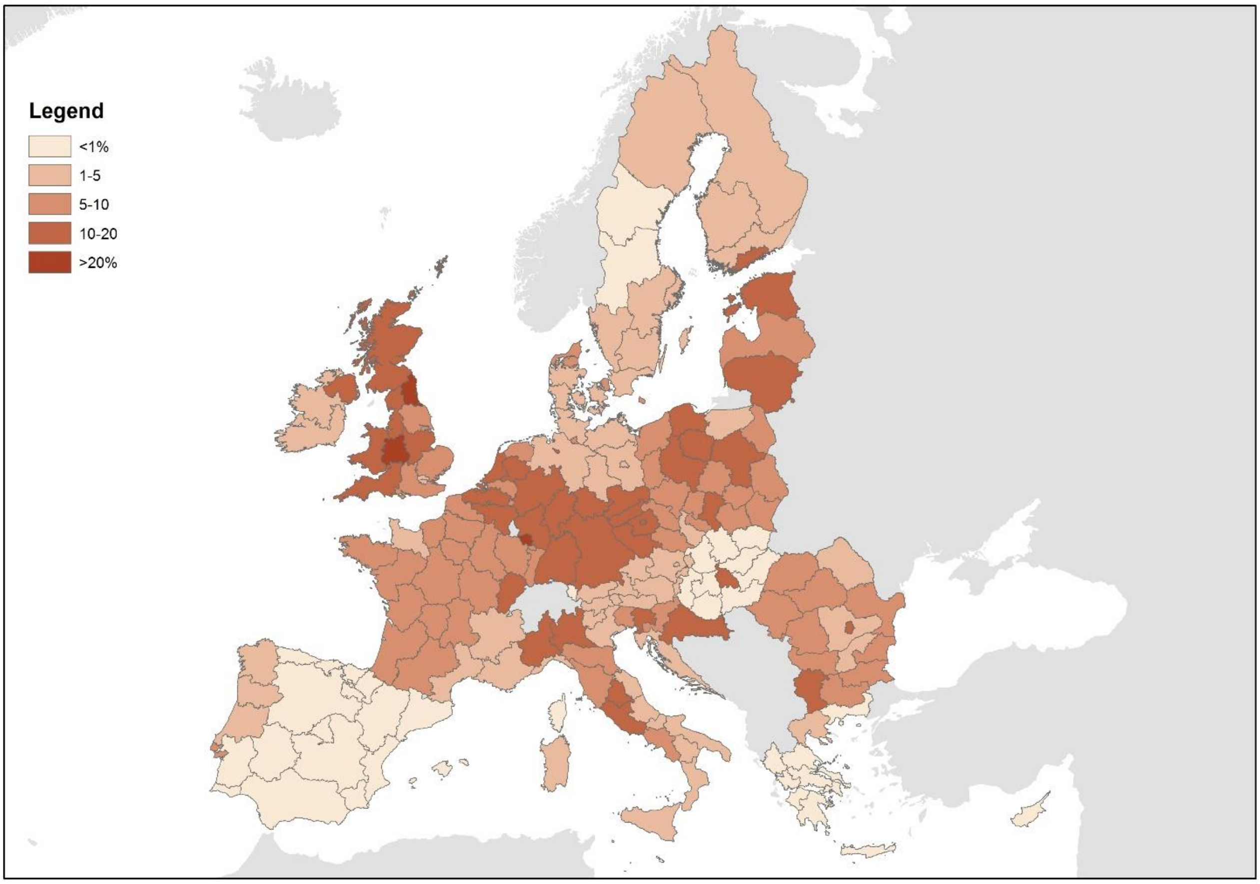



*Figure 60- Potential CSO as a percentage of dry-weather wastewater under full compliance (P) assuming d=4 (below) and with correction factors accounting for management (above)*



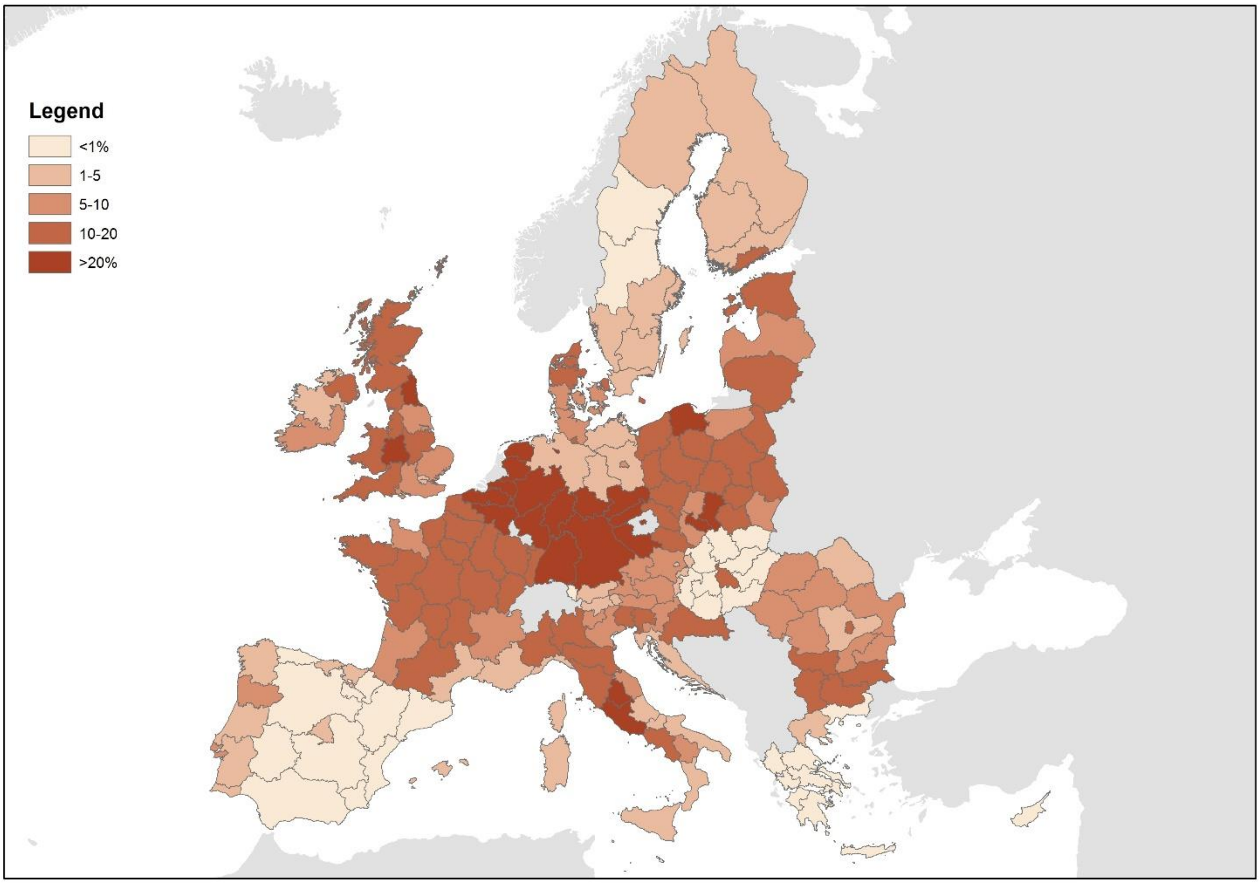
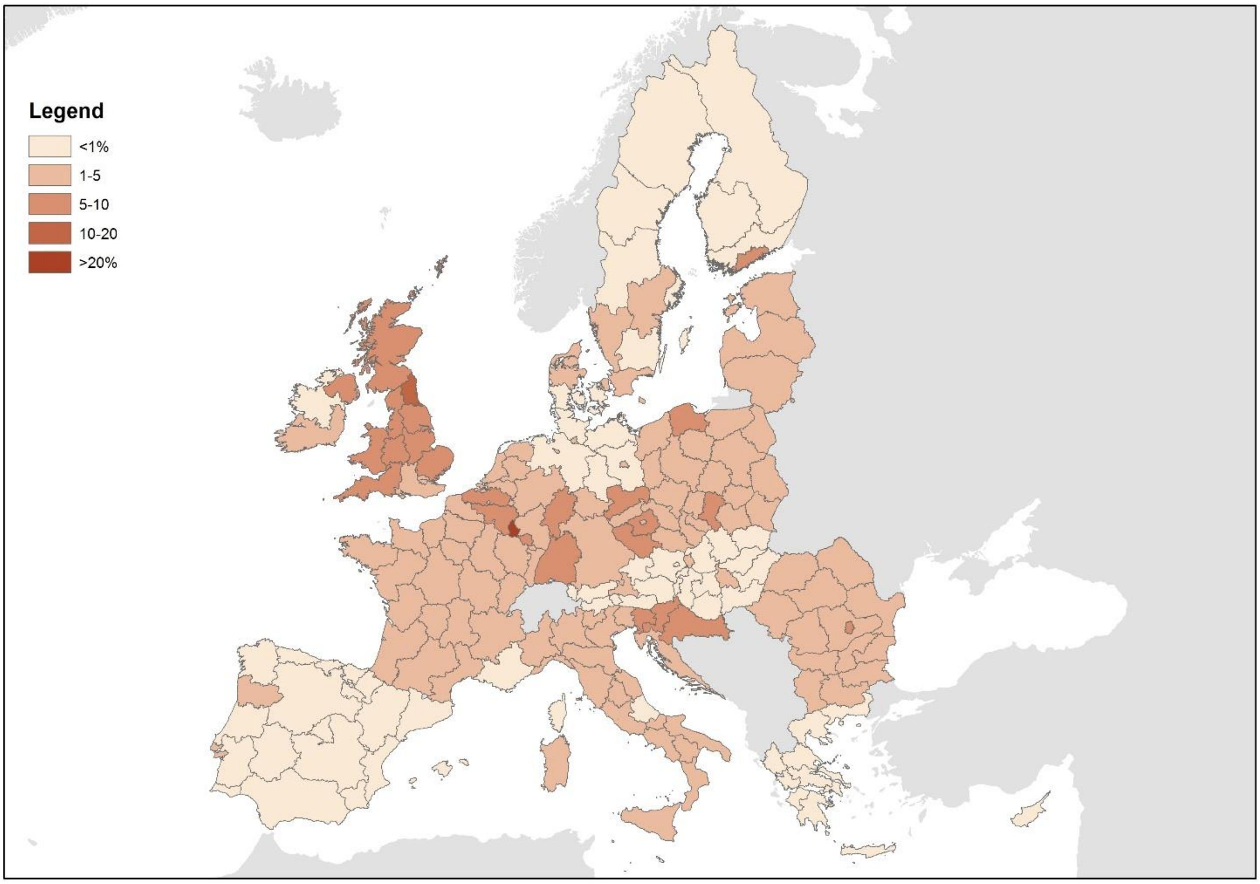



*Figure 61- Potential CSO as a percentage of dry-weather wastewater under full compliance (metach.c#2) assuming d=4 (below) and with correction factors accounting for management (above)*



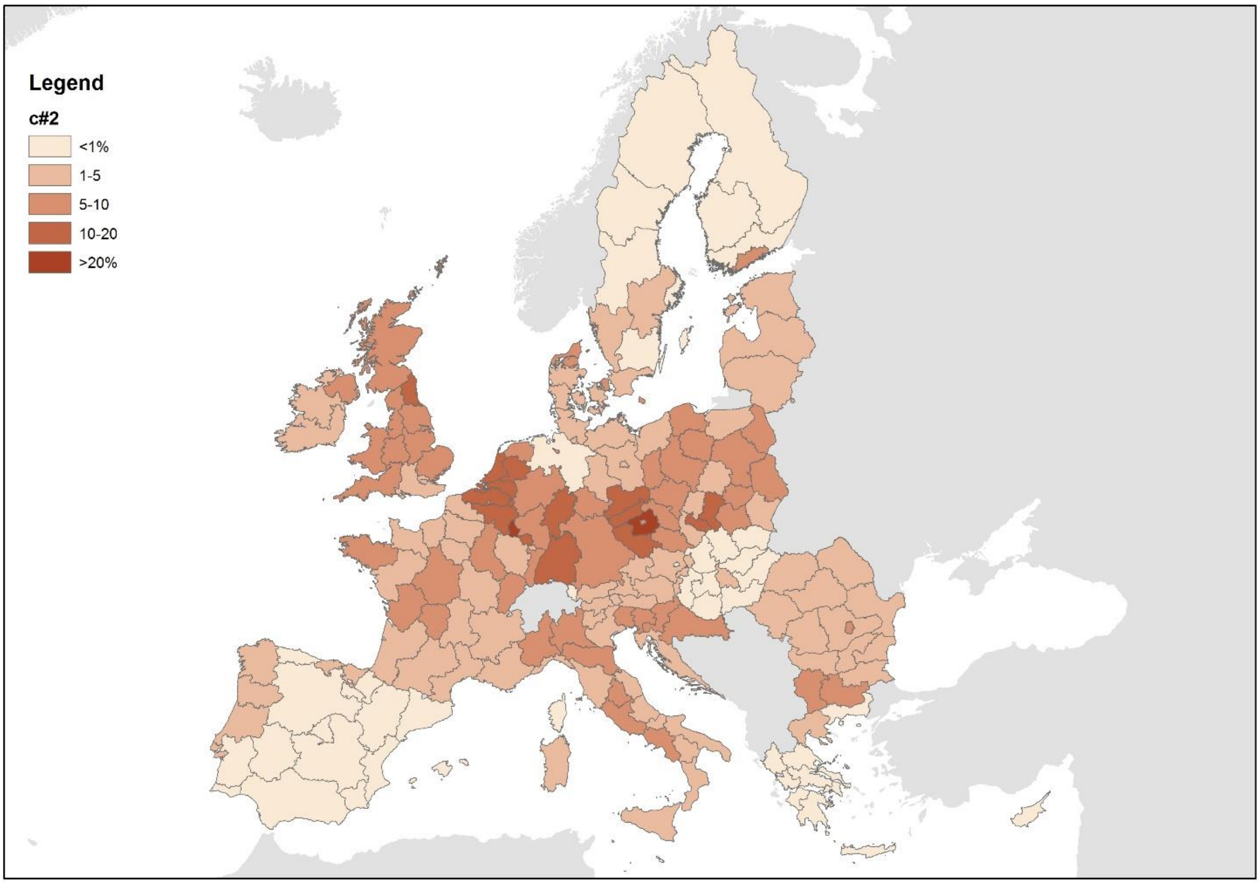
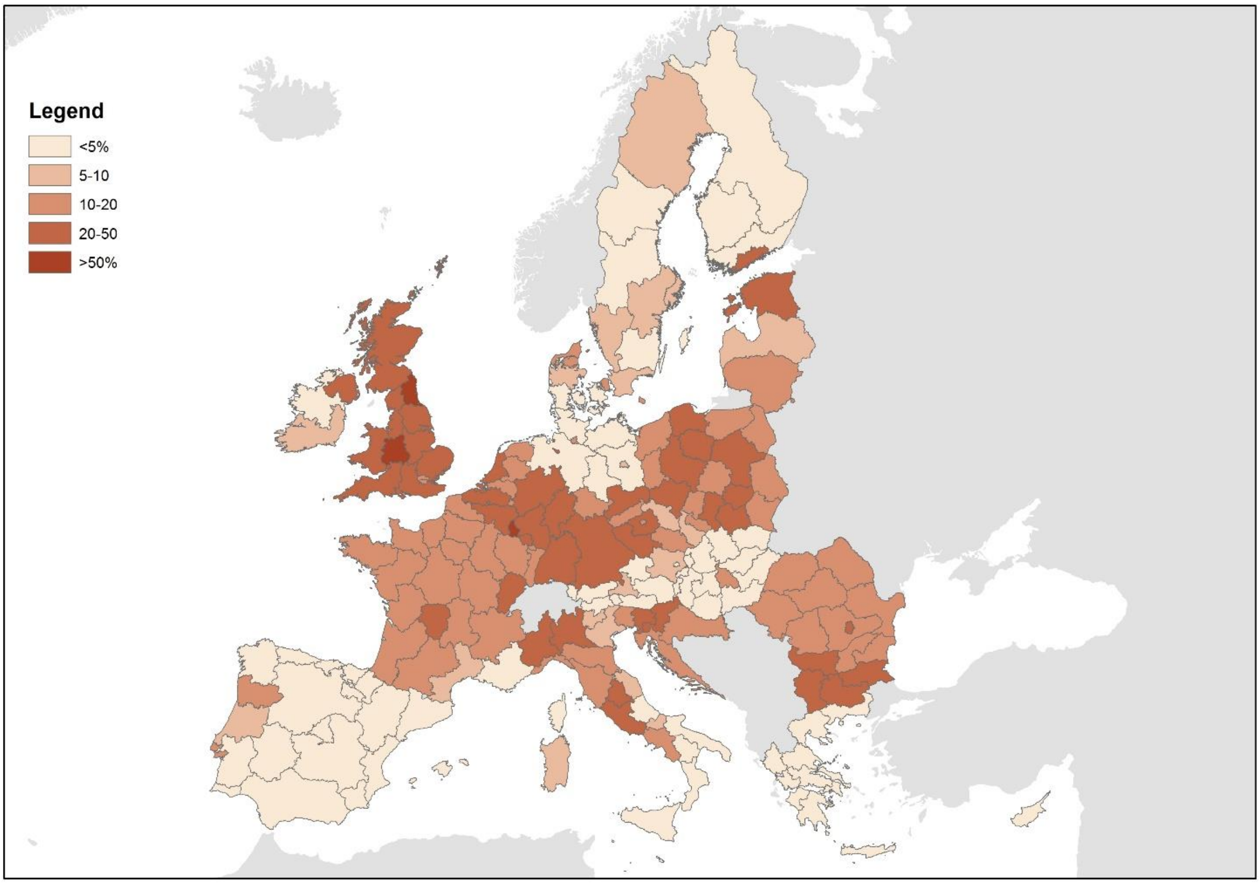



*Figure 62- Potential CSO as a percentage of dry-weather wastewater under full compliance (metach.c#3) assuming d=4 (below) and with correction factors accounting for management (above)*



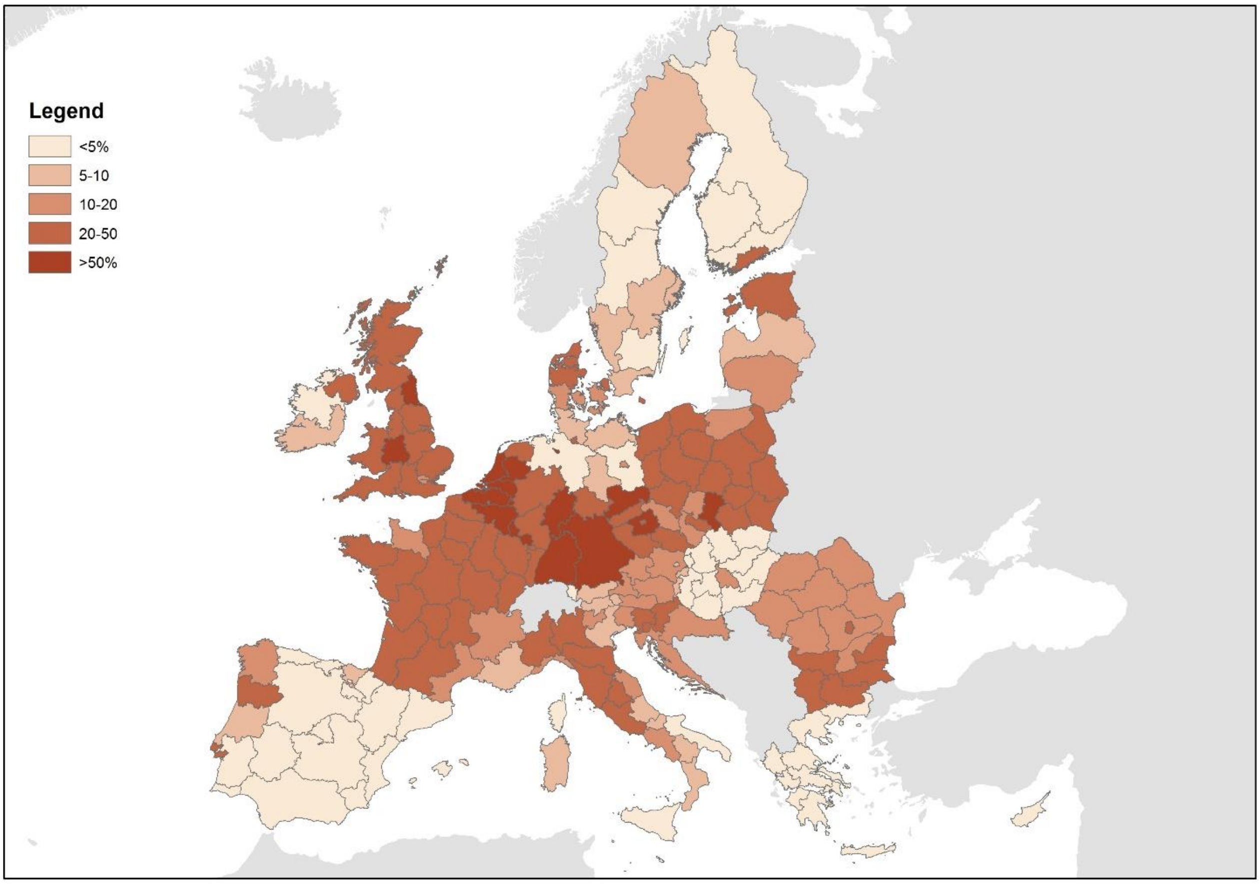
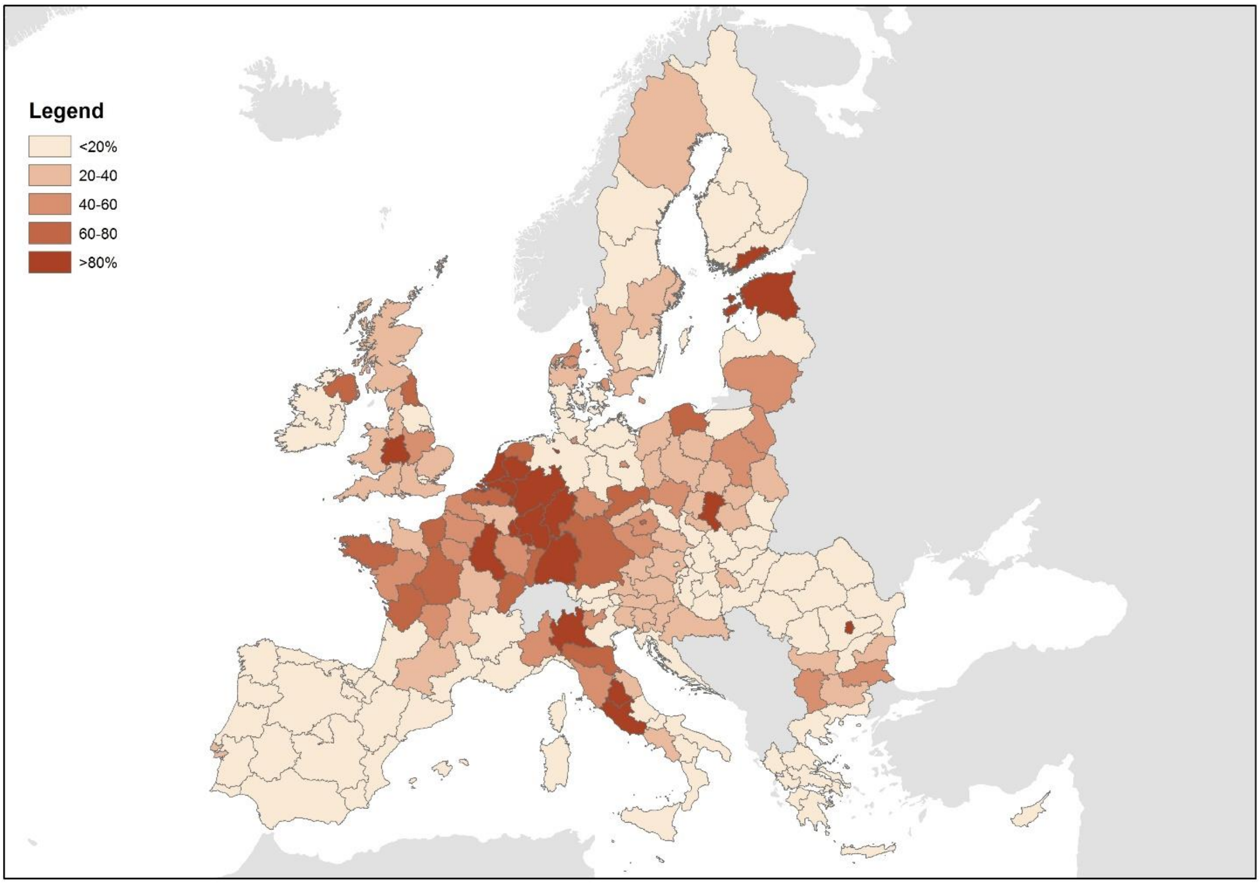



*Figure 63- Potential CSO as a percentage of dry-weather wastewater under full compliance (metach.c#4) assuming d=4 (below) and with correction factors accounting for management (above)*



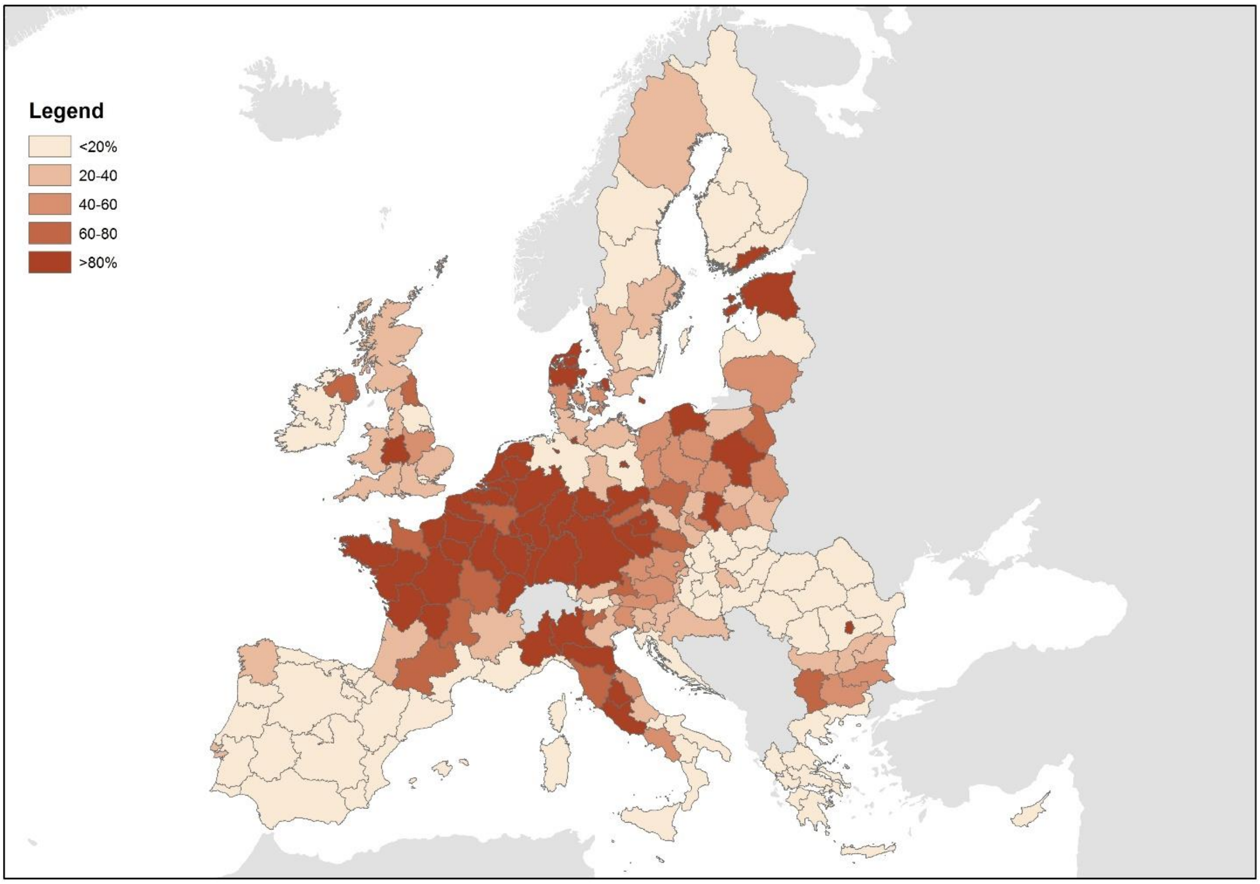
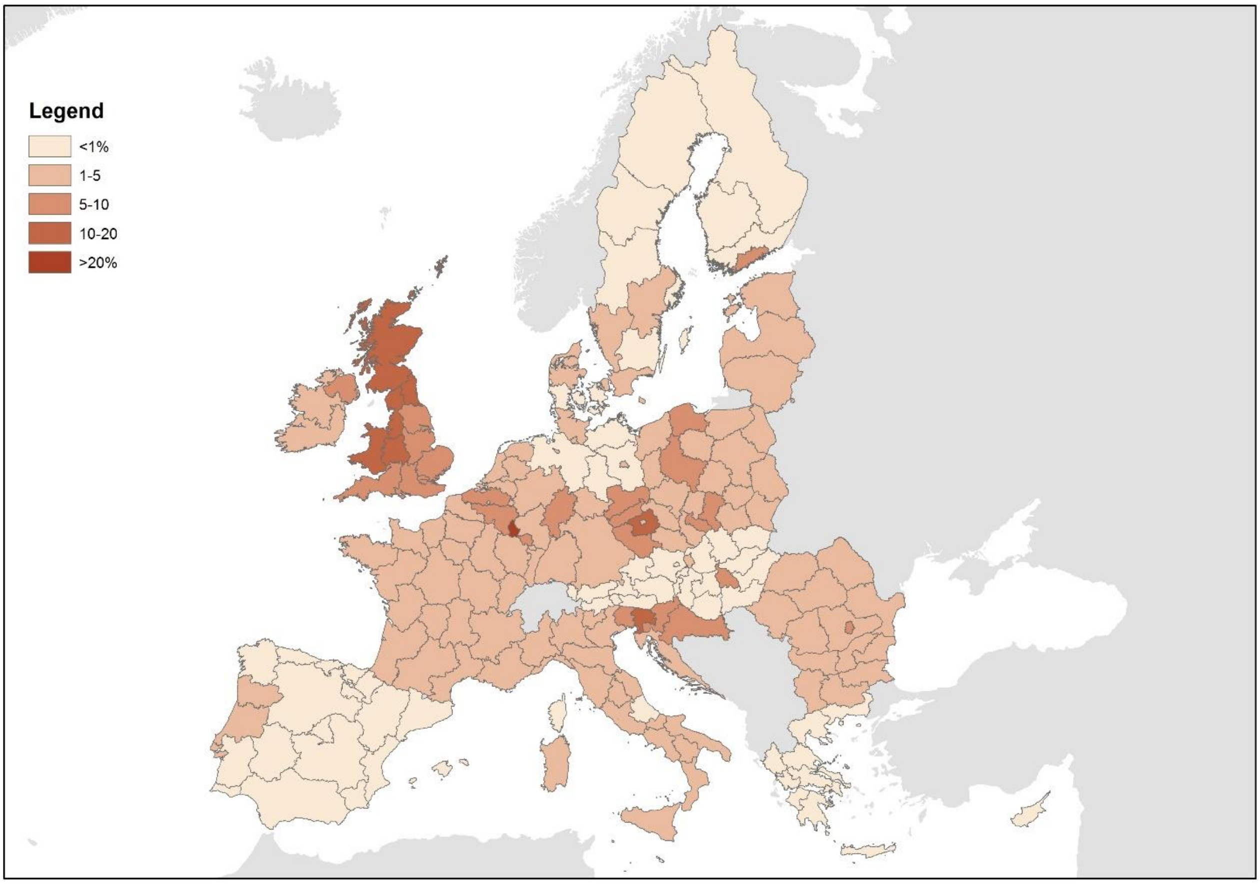



*Figure 64- Potential CSO as a percentage of dry-weather wastewater under full compliance (metach. c#5) assuming d=4 (below) and with correction factors accounting for management (above)*



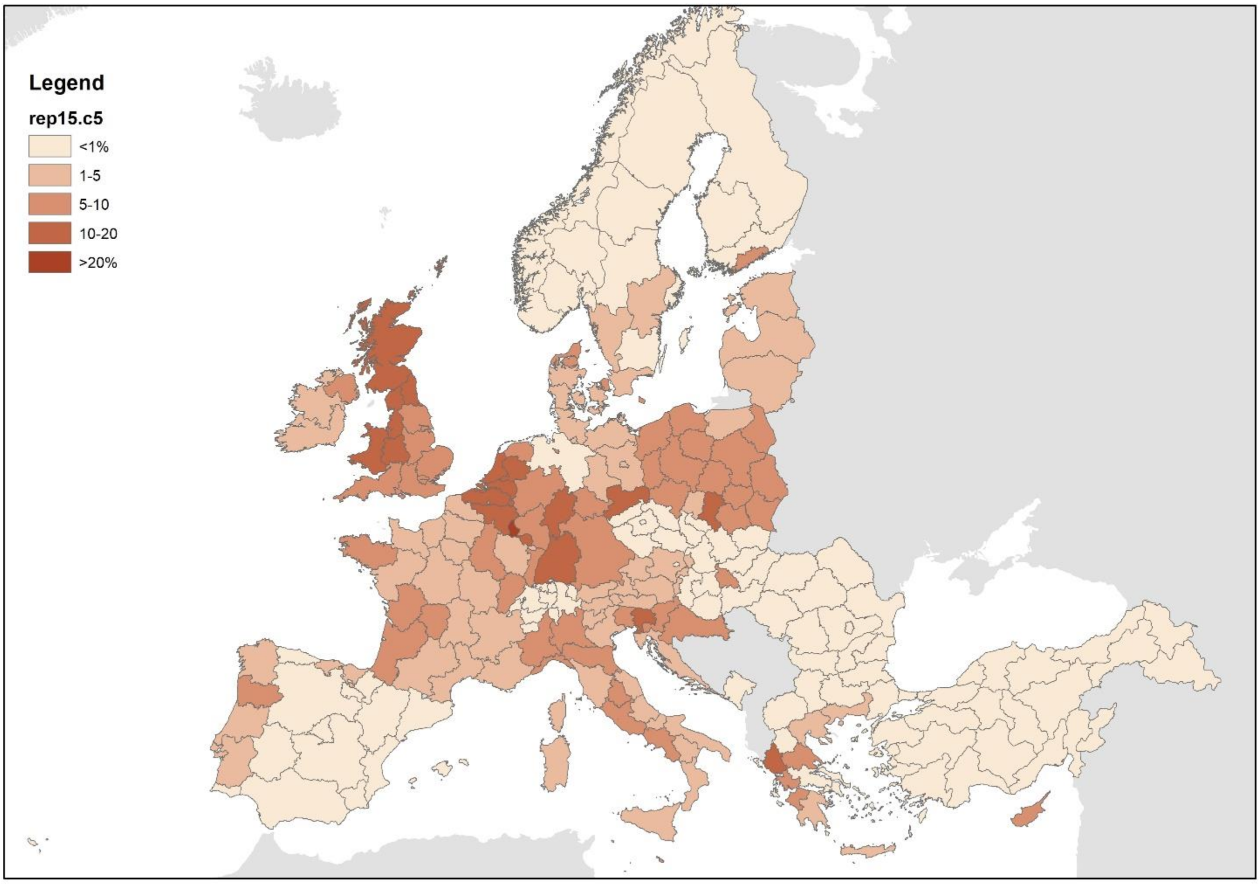
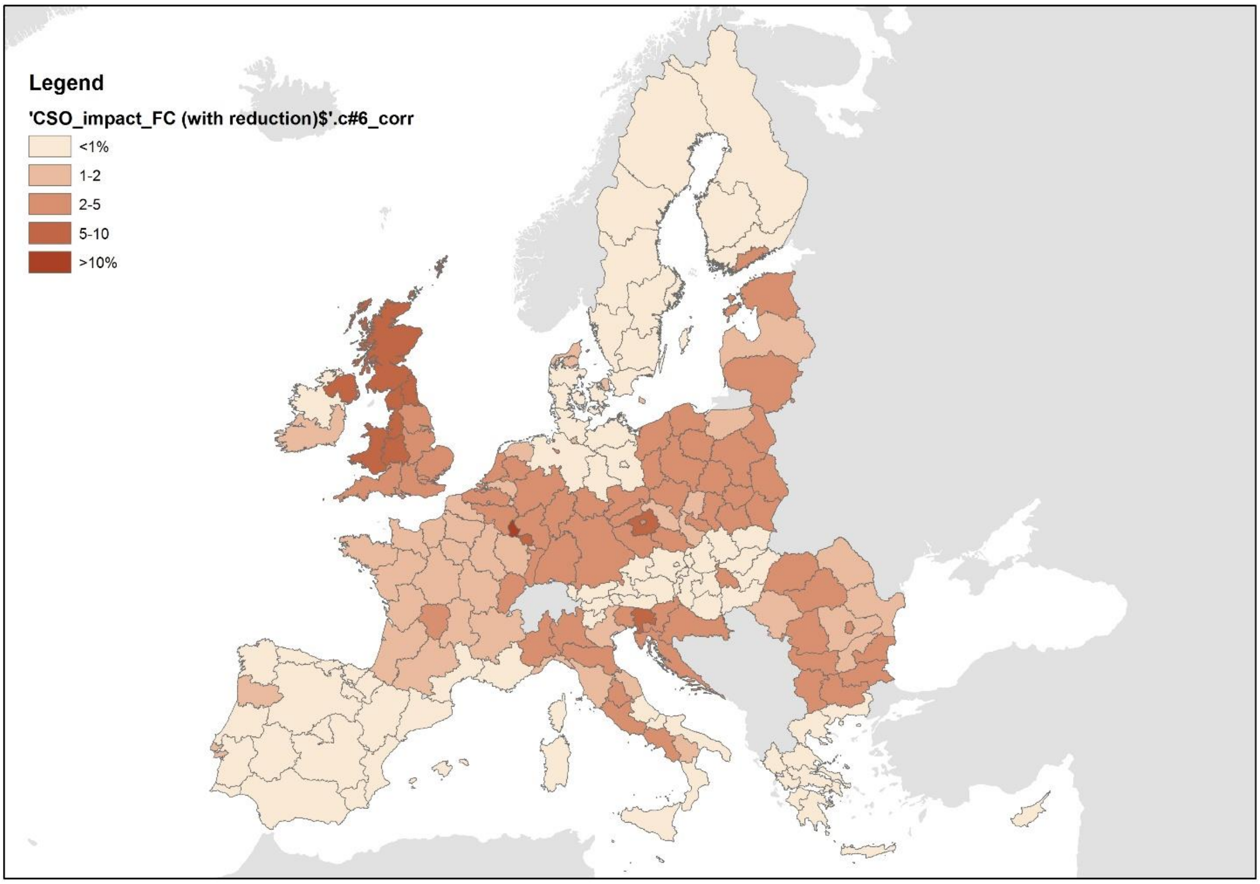



*Figure 65- Potential CSO as a percentage of dry-weather wastewater under full compliance (metach. c#6) assuming d=4 (below) and with correction factors accounting for management (above)*



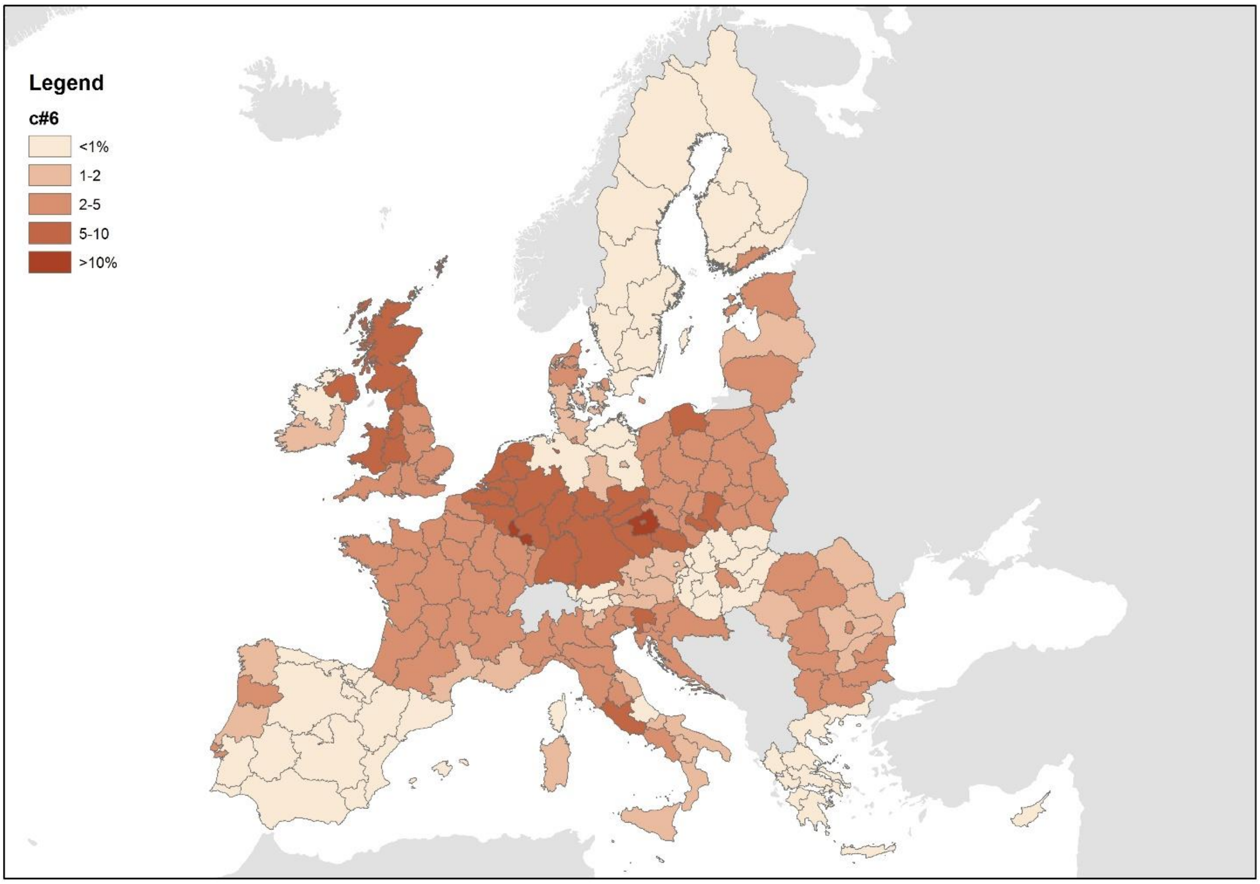
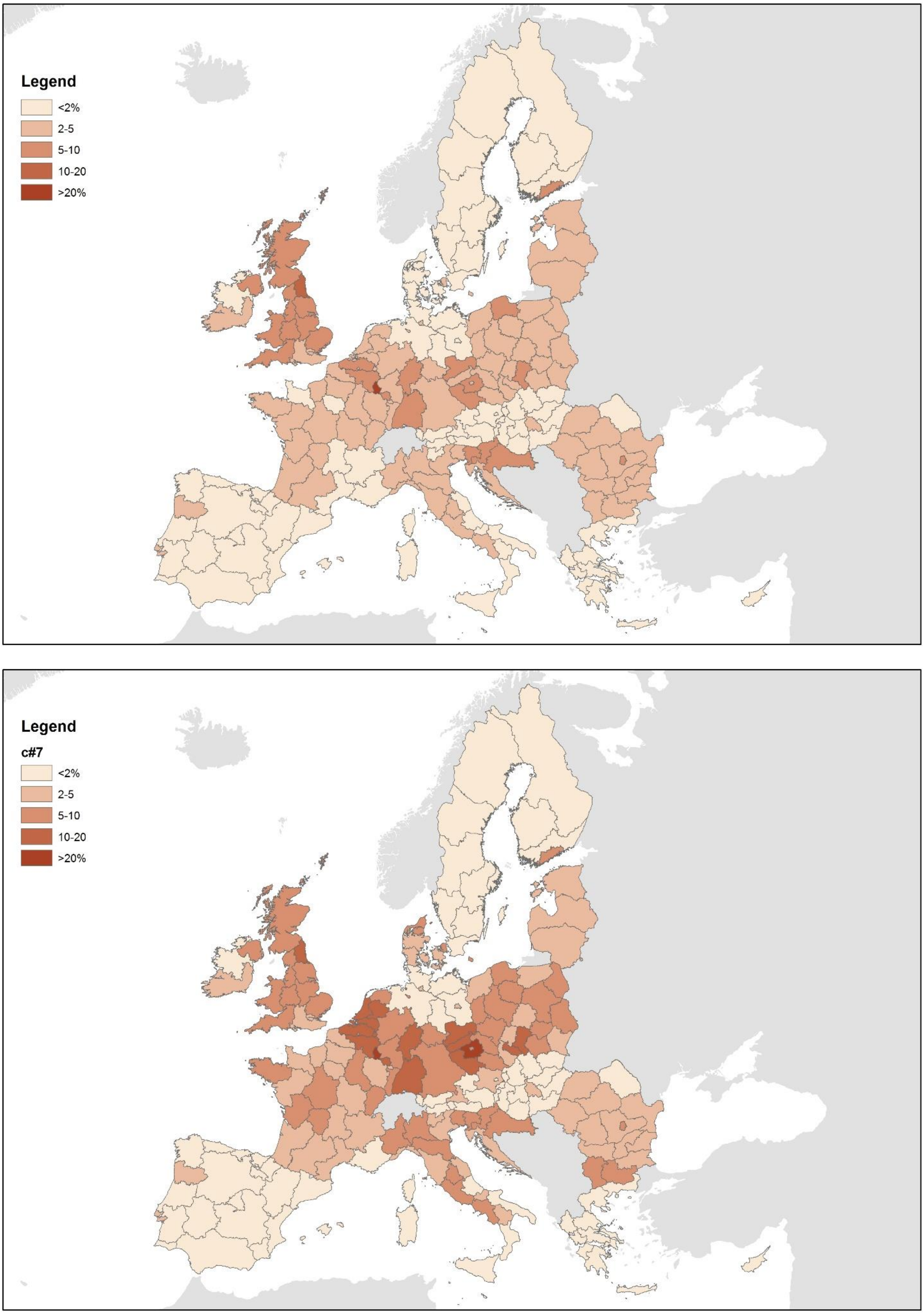



*Figure 66- Potential CSO as a percentage of dry-weather wastewater under full compliance (metach. c#7) assuming d=4 (below) and with correction factors accounting for management (above)*



# 4 Conclusions

This study provides estimates of concentrations and loads of various pollutants in Europe under present conditions, in comparison with pre-UWWTD conditions, in the absence of the directive (What-if-no-Directive or WIND scenario) and under full compliance with the UWWTD, addressing the following, interrelated questions:

- To what extent have the objectives of the UWWTD been achieved?
- To what extent are the main pollutants released by urban areas collected and treated?
- What have been the (quantitative and qualitative) effects of the UWWTD?

Concerning the first question, the implementation assessment reports periodically prepared by the Commission describe the progress towards the specific objectives of the directive. Concerning the overarching objective of "*protect*[ing] *the environment from the adverse effects of [...] waste water discharges*" (Art.1 of the UWWTD), this study has highlighted how the objectives of the UWWTD are mostly achieved when looking at the EU scale, although there are still regions and countries lagging behind and in need of stepping up the implementation. The UWWTD is estimated to have reduced very significantly the loads to the stream network and seas for all substances examined. While for some of them (coliforms, certain chemicals and, to a large extent, BOD), this has been decisive in the improvement of the conditions of water bodies (including bathing waters), for BOD and nutrients the effect on water bodies is less apparent due to the importance of other sources of pollution. Also, some chemicals are insensitive or have limited sensitivity to conventional wastewater treatment, hence the impact of the UWWTD has been less apparent. The impact of individual/appropriate systems (IAS) is generally limited, however the analysis shows that, for certain regions, if IAS are not properly operated and monitored, they might cause a sizable impact on the water bodies. When comparing the loads of pollutants under a full implementation scenario with the potential loads coming from CSO, it appears that the latter may represent sizable, and sometimes very high proportions of the loads, and could consequently become a primary source of pollution gaining all the more importance as the dry weather wastewater loads are treated more and more appropriately.

Finally, it may be of interest to analyze what remains to be done in terms of the management of residual urban wastewater loads[104]. Table 16 summarizes the PE subject to different levels of treatment by country at present and under full compliance with the UWWTD.

The full implementation of the UWWTD can be estimated to imply for certain agglomerations, currently subject to treatment less stringent than biological, to a secondary or more stringent level. This would cause a reduction in the loads of pollution discharged to the receiving water bodies, which can be quantified around 20 to 30 million PE in the whole EU (Figure 67). In comparison, if we assume that IAS usually correspond to a primary level of treatment (which may be excessively conservative in many countries having a clear IAS management system in place), enforcing a secondary or more stringent level of treatment in IAS may correspond to about 5 million PE. Scattered dwellings (SD) account for about 23 million PE (see Vigiak et al., 2018); while the current level of treatment of SD is unknown as it is not regulated by the UWWTD nor reported by the EU member states, it is unlikely to be less than primary. Considering, moreover, that SD should represent agglomerations below 2000 PE without a collection system, the length of pollution pathways from these sources to the receiving water bodies is expected to be an additional factor causing natural attenuation. Based on these considerations, a shift in the level of treatment of SD from primary to secondary or more stringent as for agglomerations above 2000 PE may be estimated to entail a reduction of loads in the order of 10 million PE. Finally, CSO represent at most about 10 million PE, which would reduce to about 6 millions if we assume that the national provisions in place are actually effectively implemented. This figure is likely overestimating the impact of CSO, and is to be regarded as a provisional estimate. On top of these "avoidable loads", we should consider urban runoff, which is a well known source of pollution. Although with a high uncertainty, urban runoff loads are estimated in Pistocchi, 2019, to be in the order of about 30 million PE for BOD, about 18 for N and about 6 for P, and less than 1 million for coliforms. Figure 67 summarizes the relative importance of the avoidable loads, highlighting that full implementation of the UWWTD remains a major target, while among the other sources of pollution scattered dwellings and CSO represent the most important ones, and urban runoff should clearly be factored in at least for what concerns BOD and nutrients.

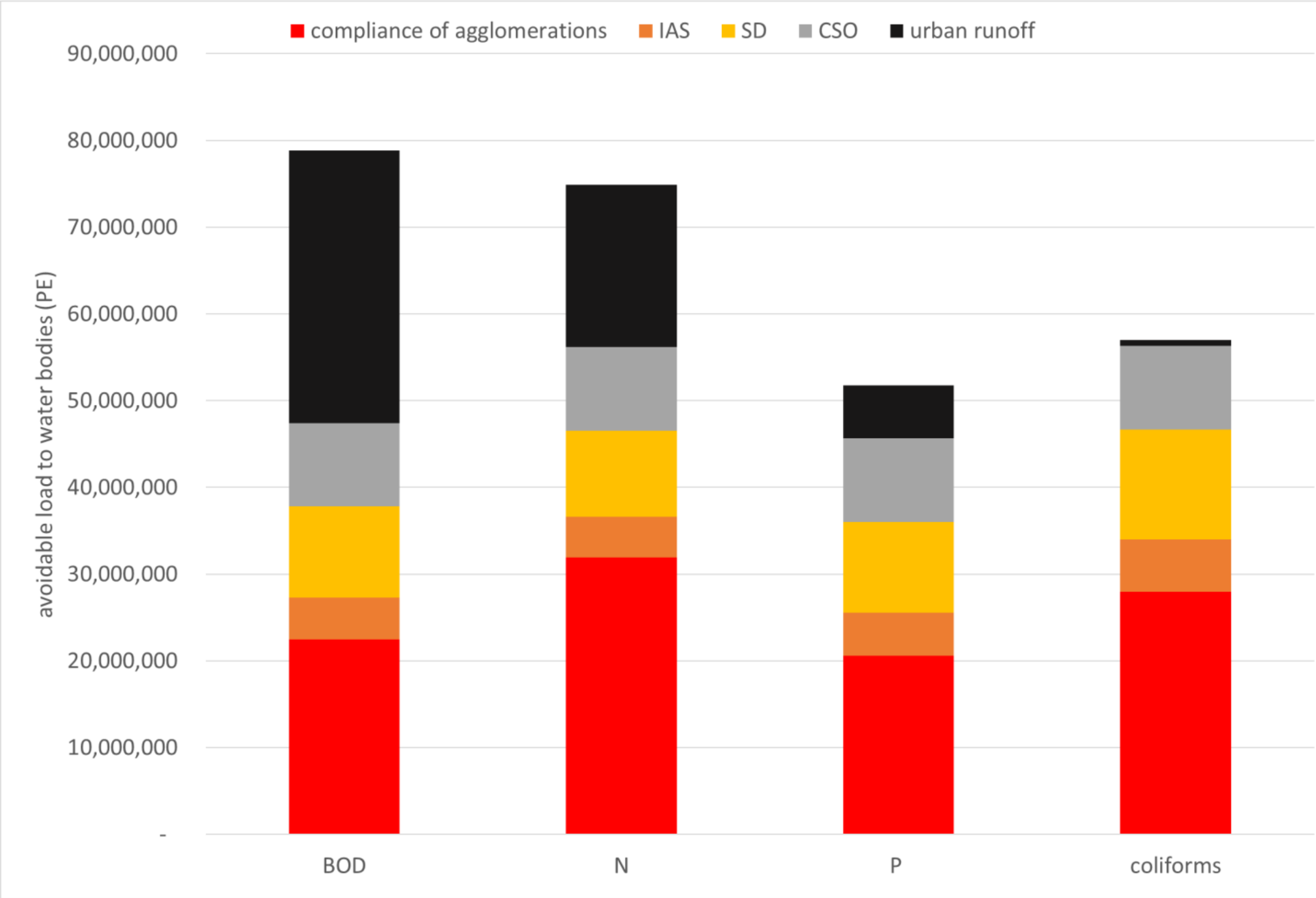
[104] Detailed calculations yielding the figures presented below are described in Annex V.

*Table 16 – PE subject to different levels of treatment, by country, under the current situation and full compliance. SD represents the estimated population in "scattered dwellings", or agglomerations below 2,000 PE without a wastewater collection system in place. For CSO, "bottom line" represents the most conservative assumption made in this work, while "managed" represents a scenario where appropriate measures are implemented in order to enforce national standards of CSO in each of the EU countries (see §2.2.4).*

| | **Current situation** | | | | | | **Full compliance** | | | | **CSO** | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **country** | **SD** | **IAS** | **untreated** | **primary** | **secondary** | **More stringent** | **SD** | **IAS** | **secondary** | **More stringent** | **"bottom line"** | **"managed"** |
| AT | 0 | 75 | 19 | 0 | 34331 | 20310667 | 0 | 75 | 34350 | 20310667 | 92127 | 40570 |
| BE | 983945 | 52 | 1036985 | 129384 | 187645 | 9128969 | 983945 | 52 | 1335730 | 9147254 | 386344 | 192440 |
| BG | 573210 | 526 | 1302561 | 1073202 | 1832867 | 3958096 | 573210 | 526 | 1082310 | 7084417 | 130447 | 101685 |
| CY | 325890 | 14990 | 240545 | 0 | 16830 | 722651 | 325890 | 14990 | 241745 | 738281 | 0 | 0 |
| CZ | 1577864 | 524863 | 550569 | 45174 | 231995 | 6987186 | 1577864 | 524863 | 764683 | 7050241 | 291588 | 128407 |
| DE | 0 | 1267168 | 1309 | 83647 | 1125373 | 109131324 | 0 | 1267168 | 1108147 | 109233506 | 2480770 | 1176765 |
| DK | 0 | 0 | 0 | 2000 | 52210 | 11523643 | 0 | 0 | 52210 | 11525643 | 136784 | 43213 |
| EE | 0 | 516 | 7170 | 0 | 10816 | 1599717 | 0 | 516 | 15493 | 1602210 | 18379 | 18379 |
| EL | 714181 | 1142448 | 765 | 0 | 373724 | 10276349 | 714181 | 1142448 | 14998 | 10635840 | 10438 | 5406 |
| ES | 0 | 937716 | 883809 | 2957866 | 25822430 | 33863804 | 0 | 937716 | 17715589 | 45812321 | 113722 | 50080 |
| FI | 927998 | 0 | 0 | 47300 | 0 | 5430900 | 927998 | 0 | 0 | 5478200 | 33023 | 33023 |
| FR | 5881050 | 270 | 549 | 0 | 11037970 | 59030629 | 5881050 | 270 | 8770918 | 61298230 | 847893 | 440599 |
| HR | 19979 | 259398 | 366502 | 2825382 | 1342191 | 239752 | 19979 | 259398 | 1335514 | 3438314 | 77446 | 77446 |
| HU | 251047 | 1517486 | 29304 | 27243 | 2268139 | 8138690 | 251047 | 1517486 | 896602 | 9566774 | 64666 | 64666 |
| IE | 149175 | 262863 | 0 | 215439 | 3859031 | 1312856 | 149175 | 262863 | 1684558 | 3702768 | 52110 | 40621 |
| IT | 0 | 3453791 | 579646 | 1980998 | 22177995 | 49576119 | 0 | 3453791 | 5088002 | 69226756 | 1093418 | 682976 |

| | Current situation | | | | | | Full compliance | | | | CSO | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| country | SD | IAS | untreated | primary | secondary | More stringent | SD | IAS | secondary | More stringent | “bottom line” | “managed” |
| LT | 692886 | 128663 | 0 | 0 | 30050 | 2489560 | 692886 | 128663 | 30050 | 2489560 | 37261 | 37261 |
| LU | 10267 | 3841 | 411 | 0 | 41597 | 578500 | 10267 | 3841 | 39807 | 580700 | 47265 | 47265 |
| LV | 559156 | 85728 | 8312 | 182148 | 21123 | 1315628 | 559156 | 85728 | 188840 | 1338371 | 14879 | 14879 |
| MT | 0 | 0 | 0 | 0 | 24667 | 488334 | 0 | 0 | 0 | 513001 | 0 | 0 |
| NL | 95803 | 950 | 2693 | 0 | 128533 | 17536724 | 95803 | 950 | 12005 | 17655945 | 580132 | 167475 |
| PL | 7264904 | 200144 | 231026 | 749402 | 6510506 | 30903109 | 7264904 | 200144 | 4084114 | 34309928 | 775095 | 520813 |
| PT | 242816 | 0 | 6090 | 1234100 | 9001967 | 1752817 | 242816 | 0 | 4771937 | 7223037 | 129432 | 80846 |
| RO | 608032 | 71490 | 10478834 | 1616823 | 6227996 | 4752304 | 608032 | 71490 | 8104410 | 14971548 | 266005 | 266005 |
| SE | 0 | 0 | 0 | 0 | 0 | 12517115 | 0 | 0 | 0 | 12517115 | 35209 | 35209 |
| SI | 775267 | 92063 | 243104 | 156932 | 613674 | 493124 | 775267 | 92063 | 431704 | 1075130 | 44613 | 44613 |
| SK | 1680047 | 759860 | 14199 | 44551 | 1365202 | 2525093 | 1680047 | 759860 | 270007 | 3679038 | 7345 | 7345 |
| UK | 869 | 292387 | 432 | 16268 | 44591087 | 26004641 | 869 | 292387 | 33782636 | 36829792 | 1868659 | 1868659 |

*Figure 67 –loads that can be avoided by enforcing full compliance with the UWWTD (for agglomerations); an equivalent treatment level (for scattered dwellings, SD); full control of CSO (neglecting management measures currently in place); and effective enforcement of IAS treatment equivalent to the WWTP of the corresponding agglomeration.*




Finally, it should be stressed that urban wastewater treatment is important but cannot address, per se, other sources of pollution, and particularly agricultural diffuse sources (for nutrients) and livestock (for BOD). As shown in Figure 68, livestock accounts for about one third of the emissions of BOD to European water bodies, but sizable contributions come from industrial sources and forestry as well; similarly, Figure 69 presents the results of the apportionment of nutrient loads to the European seas calculated by the GREEN model, highlighting that fertilizer use in agriculture typically contributes in the range of one third to half of the load of N. For P, the contribution tends to be smaller although still very significant. In any case, a successful management of pollution cannot leave aside diffuse agricultural sources.

*Figure 68 – loads of BOD to EU water bodies by source (tonnes per year): above, by country; below, EU28 totals (source: Vigiak et al., 2019)*

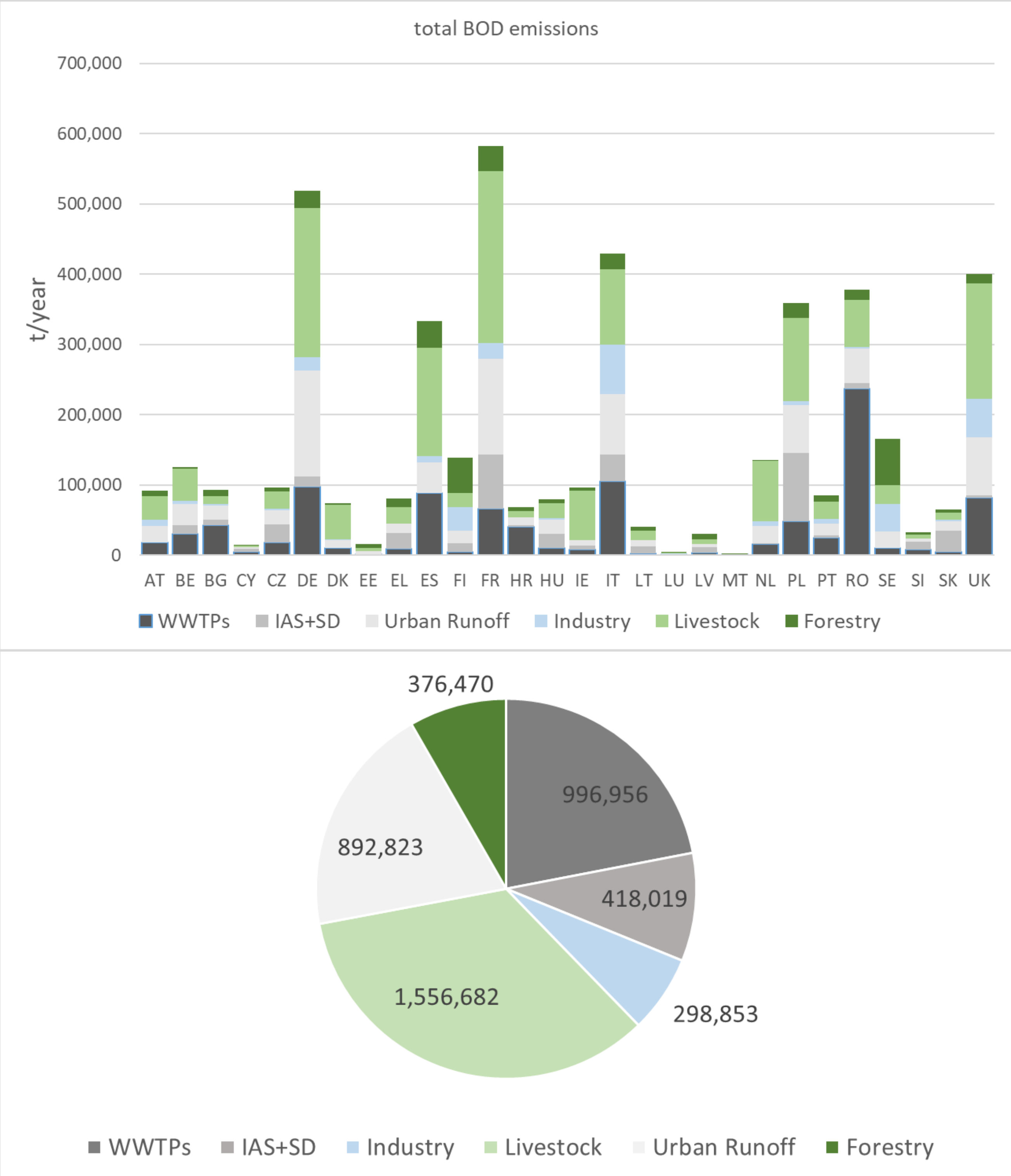
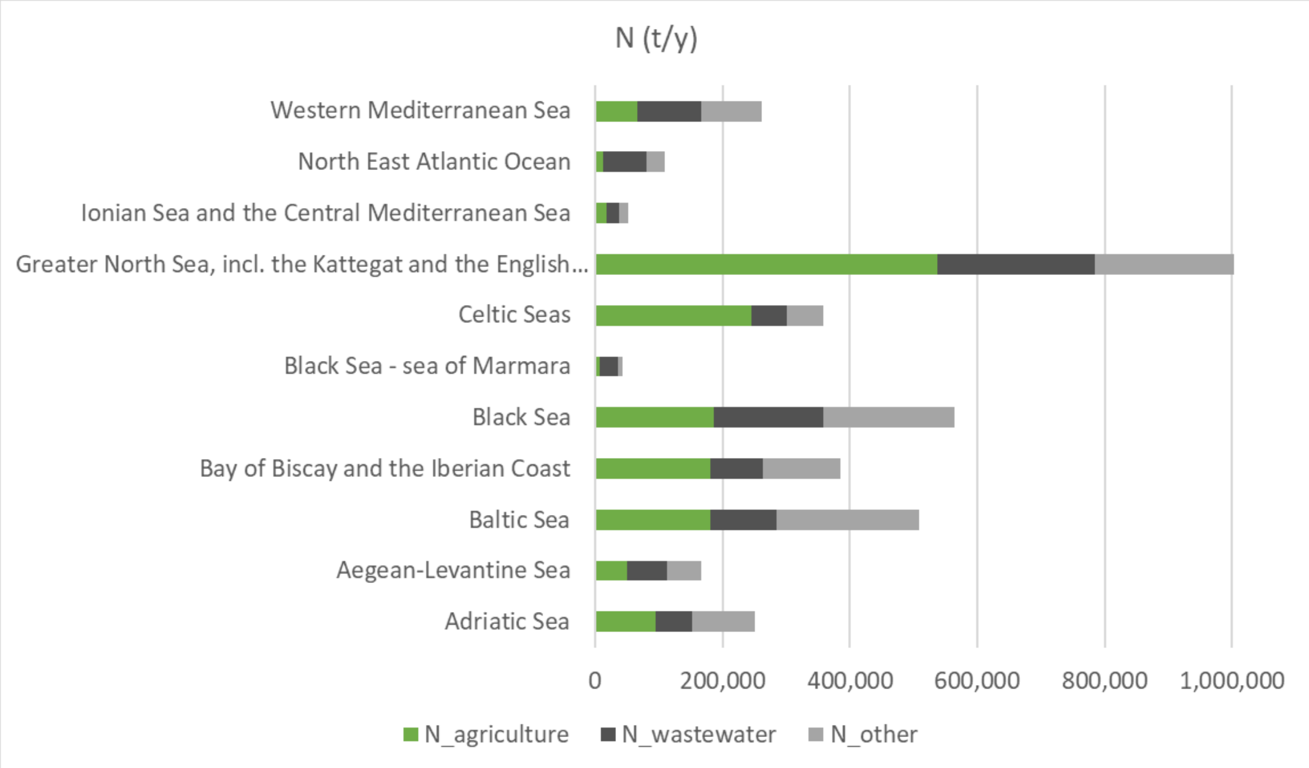
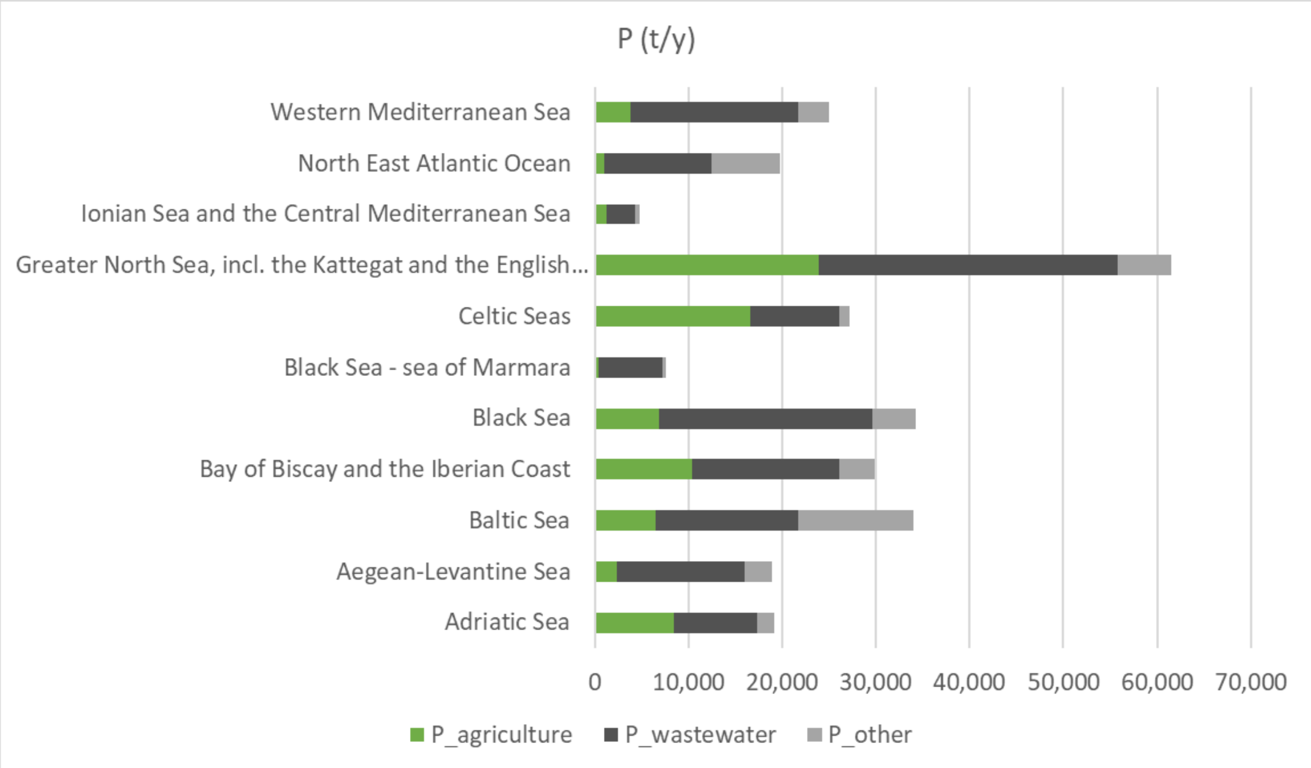


*Figure 69 – loads of N and P to EU regional seas by source (tonnes per year): above, N; below, P*



## References


[1] Acuña V, Schiller D, García-Galán MJ, Rodríguez-Mozaz S, Corominas L, Petrovic M, Poch M, Barceló D, Sabater S (2015) Occurrence and in-stream attenuation of wastewater-derived pharmaceuticals in Iberian rivers. Sci Total Environ 503-504:133–141

[2] Adeel et al. (2016) Environmental impact of estrogens on human, animal and plant life: A critical review; Environment International, https://doi.org/10.1016/j.envint.2016.12.010 .

[3] Araujo et al. (2014) "Persistence of Ibuprofen, Ketoprofen, Diclofenac and Clofibric Acid in Natural Waters" "JOURNAL OF ENVIRONMENT AND HUMAN ISSN(Print): 2373-8324 ISSN(Online): 2373-8332 Volume 1, Number 2, July 2014"

[4] Baena Nogueras et al., 2017, Degradation kinetics of pharmaceuticals and personal care products in surface waters: photolysis vs biodegradation. Science of the Total Environment, https://dx.doi.org/10.1016/j.stotenv.2017.03.015

[5] Barbosa et al., 2016, Occurrence and removal of organic micropollutants: an overview of the watch list of EU Decision 2015/495. Water Research, https://dx.doi.org/10.1016/j.watres.2016.02.047

[6] Beck, H.E., A.I.J.M. van Dijk, V. Levizzani, J. Schellekens, D.G. Miralles, B. Martens, A. de Roo: MSWEP: 3-hourly 0.25° global gridded precipitation (1979–2015) by merging gauge, satellite, and reanalysis data, Hydrology and Earth System Sciences, 21(1), 589–615, 2017a. Information on the dataset can be found at http://gloh2o.org/

[7] Bouraoui, F., Grizzetti, B. and Aloe, A., 2011. Long term nutrient loads entering European seas. JRC scientific and technical reports, EUR 24726 EN, Publications Office of the European Union, Luxembourg.

[8] Braga, O. (2005) Steroid estrogens in primary and tertiary wastewater treatment plants. Water Sci Technol. 2005;52(8):273-8.

[9] Brunel et al 1997, Collecte des donnees relatives a l'utilisation des resources en eau et leur gestion en France. Vol I: Donnees quantitative recueillie. Centre Commun di Recherche et la Commission Europeenne, contract n 12519-96-12 FIPC SEV F

[10] Cantinho et al. (2016) Behaviour and fate of metals in urban wastewater treatment plants: a review, Int. J. Environ. Sci. Technol. 13:359–386

[11] Capela et al., 2017, volatile methylsiloxanes through wastewater treatment plants – a review of levels and implications. Environment International, https://doi.org/10.1016/j.envint.2017.03.005 .

[12] Christian et al., 2017, Use of COD, TOC, and fluorescence spectroscopy to estimate BOD in wastewater. Water Environment Research, https://doi.org/10.2175/106143016X14504669768976

[13] Csanady, G.T., Turbulent Diffusion in the Environment, 1970

[14] Dai, G., HuangJun Huang, J., Chen, W., DengShubo, S. (2014) Major Pharmaceuticals and Personal Care Products (PPCPs) in Wastewater Treatment Plant and Receiving Water in Beijing, China, and Associated Ecological Risks. Bulletin of Environmental Contamination and Toxicology 92(6) 655-661

[15] Dubber and Gray, 2010 Replacement of chemical oxygen demand (COD) with total organic carbon (TOC) for monitoring wastewater treatment performance to minimize disposal of toxic analytical waste. Journal of Environmental Sciences and Health Part A, https://doi.org/10.1080/10934529.2010.506116

[16] European Commission (Eurostat, Joint Research Centre and DG Regional Policy - REGIO-GIS), GEOSTAT population Grid 2011. Information on the dataset can be found at http://ec.europa.eu/eurostat/statistics-explained/index.php/Population_grids

[17] European Commission, Joint Research Centre (JRC); Columbia University, Center for International Earth Science Information Network - CIESIN (2015). GHS population grid, derived from GPW4, multitemporal (1975, 1990, 2000, 2015). European Commission, Joint Research Centre (JRC) [Dataset] PID: http://data.europa.eu/89h/jrc-ghsl-ghs_pop_gpw4_globe_r2015a [accessed 4/4/2018]

[18] European Environment Agency (EEA 2012) Copernicus Urban Atlas: https://land.copernicus.eu/local/urban-atlas/urban-atlas-2012?tab=metadata

[19] European Environment Agency (EEA). 2017. Waterbase - UWWTD: Urban Waste Water Treatment Directive – reported data. v6. https://www.eea.europa.eu/data-and-maps/data/waterbase-uwwtd-urban-waste-water-treatment-directive-5 [accessed 10th May 2018]

[20] European Environment Agency (EEA). 2018. The European Pollutant Release and Transfer Register (E-PRTR) is a web-based register established by Regulation (EC) No 166/2006 which implements the UNECE PRTR Protocol, signed in May 2003 in Kiev. https://www.eea.europa.eu/data-and-maps/data/member-states-reporting-art-7-under-the-european-pollutant-release-and-transfer-register-e-prtr-regulation-19 [accessed 1/8/2018]

[21] Francy, D.S., Myers, D.N. , Metzker, K.D. Escherichia coli and fecal-coliform bacteria as indicators of recreational water quality. Water-Resources Investigations Report 93-4083, 1993. https://doi.org/10.3133/wri934083

[22] Gackowska et al. (2018) "Estimation of physicochemical properties of 2-ethylhexyl-4-methoxycinnamate (EHMC) degradation products and their toxicological evaluation" Environmental Science and Pollution Research (2018) 25:16037–16049; https://link.springer.com/content/pdf/10.1007%2Fs11356-018-1796-6.pdf

[23] Gao et al. (2016) Fate and removal of typical pharmaceutical and personal care products in a wastewater treatment plant from Beijing: a mass balance study Front. Environ. Sci. Eng. 2016, 10(3): 491–501

[24] Gericke J.G., Smithers J.C. 2014. Review of methods used to estimate catchment response time for the purpose of peak discharge estimation. Hydrological Sciences Journal 59, 1935-1971, doi: 10.1080/02626667.2013.866712

[25] GLOBAQUA. 2018. Policy recommendations, Globaqua Deliverable D012. Globaqua project report D11.12. http://www.globaqua-project.eu/files/repository/20180720131147_20180205110856_GLOBAQUADeliverable012(D11.12).pdf

[26] Grizzetti, B., Bouraoui, F., Aloe, A., 2012. Changes of nitrogen and phosphorus loads to European seas. Global Change Biology 18, 769-782.

[27] Grizzetti, B., A. Pistocchi, C. Liquete, A. Udias, F. Bouraoui & W. van de Bund. Human pressures and ecological status of European rivers. Scientific Reports 7, Article number: 205 (2017) https://doi.org/10.1038/s41598-017-00324-3

[28] Gros et al. (2010) "Removal of pharmaceuticals during wastewater treatment and environmental risk assessment using hazard indexes". Environment International 36 (2010) 15–26

[29] Hanamoto et al., 2018 The different fate of antibiotics in the Thames River, UK, and the Katsura River, Japan. https://link.springer.com/article/10.1007%2Fs11356-017-0523-z

[30] Harder-Lauridsen NM, Kuhn KG, Erichsen AC, Mølbak K, Ethelberg S (2013) Gastrointestinal Illness among Triathletes Swimming in Non-Polluted versus Polluted Seawater Affected by Heavy Rainfall, Denmark, 2010-2011. PLoS ONE 8(11): e78371. doi:10.1371/journal.pone.0078371

[31] HELCOM. 2004. The fourth Baltic sea Pollution Load compilation (PLC-4). Baltic Sea Environ. Proc. 93. Baltic Marine Environment Protection Commission, Helsinki. ISSN 0357-2994.

[32] Hernando, M.D. et al (2006) LC-MS analysis and environmental risk of lipid regulators Anal Bioanal Chem, DOI 10.1007/s00216-006-0781-y

[33] ICPDR 2004, Danube basin analysis, https://www.icpdr.org/flowpaper/viewer/default/files/nodes/documents/danube_basin_analysis_2004.pdf

[34] Jessick (2010) "Detection, fate, and bioavailability of erythromycin in environmental matrices" https://pdfs.semanticscholar.org/fbf1/5019cf309909ace82a62cc4c3d38436217e4.pdf

[35] Kristensen and Hansen, 1994 European rivers and lakes. assessment of their environmental state. EEA monographs 1. European Environmental Agency, Copenhagen, Denmark, pp 124. ISBN: 87-90198-01-08. https://www.eea.europa.eu/publications/87-90198-01-8

[36] Lajeunesse et al. (2013) Ozone oxidation of antidepressants in wastewater –Treatment evaluation and characterization of new by-products by LC-QToFMS. Chem Cent J. 2013; 7: 15.

[37] Lautz et al., 2017, Evaluation of SimpleTreat 4.0 : Simulations of Pharmaceutical removal in wastewater treatment plant facilities. Chemosphere, https://doi.org/10.1016/j.chemosphere.2016.10.123

[38] Lavalle, Carlo; Kompil, Mert; Barbosa, Ana (2015): UI - Artificial Areas per Inhabitant (LUISA Platform REF2014). European Commission, Joint Research Centre (JRC) [Dataset] PID: http://data.europa.eu/89h/jrc-luisa-ui-art-areas-per-inh-ref-2014

[39] Liu, Y., Sun, W., Ni, J. Biodegradation of bisphenol A, 17β-estradiol, and 17α- ethynylestradiol in river water (2011) International Journal of Environment and Pollution, 45 (1-3), pp. 225-236.

[40] Magnér et al. (2016) Fate of pharmaceutical residues - in sewage treatment and on farmland fertilized with sludge. IVL Swedish Environmental Research Institute Ltd.

[41] Malve, O., Tattari, S., Riihimäki, J., Jaakkola, E., Voß, A., Williams, R., Bärlund, I. 2012. Estimation of diffuse pollution loads in Europe for Continental scale modelling of loads and in-stream river water quality. Hydrological Processes 26, 2385-2394. Doi: 10.1002/hyp.9344

[42] Marttinen, Sanna K, Riitta H Kettunen, Kai M Sormunen, Jukka A Rintala, Removal of bis(2-ethylhexyl) phthalate at a sewage treatment plant, Water Research, Volume 37, Issue 6, 2003, Pages 1385-1393, https://doi.org/10.1016/S0043-1354(02)00486-4.

[43] Metcalf and Eddy 1991, Wastewater Engineering, 1991.

[44] Milieu (2016) Assessment of impact of storm water overflows from combined waste water collecting systems on water bodies (including the marine environment) in the 28 EU Member States. European Commission Specific Contract No. 070201/2014/SFRA/693725/ENV/C.2. URL https://circabc.europa.eu/sd/a/c57243c9-adeb-40ce-b9db-a2066b9692a4/Final%20Report

[45] Nakagawa (1992) in EFPIA Comments to the Draft WFD Dossier for Clarithromycin https://circabc.europa.eu/webdav/CircaBC/env/wfd/Library/working_groups/priority_substances/priority_substances_1/substance_dossiers/final_draft_dossiers/comments_background

[46] Naughton, C. and Rousselot, O. 2017. Activated Sludge. In: J.B. Rose and B. Jiménez-Cisneros, (eds) Global Water Pathogen Project. http://www.waterpathogens.org (C. Haas, J.R. Mihelcic and M.E. Verbyla (eds) Part 4 Management Of Risk from Excreta and Wastewater) http://www.waterpathogens.org/book/activated-sludge Michigan State University, E. Lansing, MI, UNESCO.https://doi.org/10.14321/waterpathogens.62

[47] Oakley, S. 2018. Preliminary Treatment and Primary Settling. In: J.B. Rose and B. Jiménez-Cisneros, (eds) Global Water Pathogen Project. http://www.waterpathogens.org (C. Haas, J.R. Mihelcic and M.E. Verbyla) (eds) Part 4 Management Of Risk from Excreta and Wastewater) http://www.waterpathogens.org/book/preliminary-treatment Michigan State University, E. Lansing, MI, UNESCO.https://doi.org/10.14321/waterpathogens.60

[48] Ottmar et al., 2012, Fate and transport of atorvastatin and simvastatin drugs during conventional wastewater treatment. Chemosphere, https://doi.org/10.1016/j.chemosphere.2012.03.066

[49] Pan, Y. et al. (2011) Evaluation of perfluorinated compounds in seven wastewater treatment plants in Beijing urban areas Volume 54, Issue 3, pp 552–558

[50] Parkerton, T. (2001) A Preliminary Analysis of Benzene Fate in Industrial Wastewater Treatment Plants: Implications for the EU Existing Substances Risk Assessment On behalf of the CEFIC Aromatics Producers Association and CONCAWE

[51] Pistocchi, A., Loos, R., A Map of European Emissions and Concentrations of PFOS and PFOA, Environmental Science & Technology 2009 43 (24), 9237-9244

[52] Pistocchi, A., Chiara Dorati, Alberto Aloe, Antoni Ginebreda, Rafael Marcé, River pollution by priority chemical substances under the Water Framework Directive: A provisional pan-European assessment, Science of The Total Environment, Volume 662, 2019, Pages 434-445, https://doi.org/10.1016/j.scitotenv.2018.12.354.

[53] Pistocchi, A., Pennington, D., European hydraulic geometries for continental scale environmental modeling Journal of Hydrology (2006) 329, 553– 567

[54] Pistocchi, A., GIS Based Chemical Fate Modeling: Principles and Applications. Wiley, Hoboken, April 2014, ISBN: 978-1-1180-5997-5, 520 pp

[55] Pistocchi A., Dorati C. (2019) Combined Sewer Overflow Management: Proof-of-Concept of a Screening Level Model for Regional Scale Appraisal of Measures. In: Mannina G. (eds) New Trends in Urban Drainage Modelling. UDM 2018. Green Energy and Technology. Springer, Cham. First Online 01 September 2018. https://doi.org/10.1007/978-3-319-99867-1_161

[56] Pistocchi, A., A preliminary pan-European assessment of pollution loads from urban runoff. Submitted, 2019

[57] Pistocchi, A., Aloe, A.; Bouraoui, F.; Grizzetti, B.; Pastori, M.; Udias Moinelo, A.; Van De Bund, W.; Vigiak, O. Assessment of the effectiveness of reported Water Framework Directive Programmes of Measures - Part II – development of a system of Europe-wide Pressure Indicators, 2017. http://publications.jrc.ec.europa.eu/repository/handle/JRC105299

[58] Reder, K., Flörke, M., Alcamo, J., Modeling historical fecal coliform loadings to large European rivers and resulting in-stream concentrations. Environmental Modelling & Software, (2015) 63, 251-263, https://doi.org/10.1016/j.envsoft.2014.10.001.

[59] Riddle, A.M., Lewis, R.E., Dispersion Experiments in U.K. Coastal Waters. Estuarine, Coastal and Shelf Science (2000) 51, 243–254. https://doi.org/10.1006/ecss.2000.0661

[60] RIVM (2012)and references within"Environmental risk limits for octamethylcyclotetrasiloxane in water A proposal for water quality standards in accordance with the Water Framework Directive" http://rivm.openrepository.com/rivm/bitstream/10029/259145/3/601714020.pdf

[61] Robinson et al., 2014, Mapping the global distribution of livestock. PLoS One, 9 (5) (2014), Article e96084, https://doi.org/10.1371/journal.pone.0096084

[62] Rodriguez-Mozaz et al. (2015) Occurrence of antibiotics and antibiotic resistance genes in hospital and urban wastewaters and their impact on the receiving river. Water Research 69 ( 2015) 234-242

[63] Sim et al. (2010)"Occurrence and fate of pharmaceuticals in wastewater treatment plants and rivers in Korea". Environmental Pollution 158 (2010) 1938–1947

[64] Staub , 2002, Concentrations of the UV filter ethylhexyl methoxycinnamate in the aquatic compartment : a comparison of modelled concentrations for Swiss surface waters with empirical monitoring data. Toxicology Letters, https://doi.org/10.1016/S0378-4274(02)00042-5

[65] Styrishave et al., 2011, Environmental risk assessment of three selective serotonin reuptake inhibitors in the aquatic environment : a case study including a cocktail scenario. Env. Tox. Chem., https://doi.org/10.1002/etc/372

[66] Sulaiman, S. et al., (2015): Stability and Removal of Atorvastatin, Rosuvastatin and Simvastatin from Wastewater, Environmental Technology, DOI: 10.1080/09593330.2015.1058422

[67] Tong et al. (2011) Photodegradation of azithromycin in various aqueous systems under simulated and natural solar radiation: kinetics and identification of photoproducts. Chemosphere. 2011 Apr;83(3):340-8. doi: 10.1016/j.chemosphere.2010.12.025

[68] USEPA(2007) Final Report: The Environmental Occurrence, Fate, and Ecotoxicity of Selective Serotonin Reuptake Inhibitors (SSRIs) in Aquatic Environments https://cfpub.epa.gov/ncer_abstracts/index.cfm/fuseaction/display.highlight/abstract/1755/report/F

[69] Verlicchi et al., 2012, Occurrence of pharmaceutical compounds in urban wastewater : removal, mass load and environmental risk after a secondary treatment. Sci. Total Environment, https://doi.org/10.1016/j.scitotenv.2012.04.028

[70] Vigiak O., Grizzetti B., Udias-Monielo A., Zanni M., Dorati C., Bouraoui, F., Aloe A., Pistocchi A. Predicting Biochemical Oxygen Demand in European freshwater bodies. Accepted for publication, Science of the Total Environment, in press, 2019.

[71] Vigiak, O., Grizzetti, B., Zanni, M., Dorati, C., Bouraoui, F., Aloe, A., Pistocchi A., Estimation of domestic and industrial waste emissions to European waters in the 2010s, EUR 29451 EN, Publications Office of the European Union, Luxembourg, 2018, ISBN 978-92-79-97297-3 , doi:10.2760/08152, JRC113729.

[72] Vigiak, O., Pistocchi, A., Aloe, A., Ludwig, R. 2017. Empirical modelling of river water temperature in water scarce European basins. In Syme, G., Hatton MacDonald, D., Fulton, B. and Piantadosi, J. (eds) MODSIM2017, 22nd International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2017, pp. 880-886. ISBN: 978-0-9872143-7-9. https://www.mssanz.org.au/modsim2017/G4/vigiak.pdf

[73] Vogt J, Soille P, De Jager A, Rimaviciute E, Mehl W, Foisneau S, Bodis K, Dusart J, Paracchini M, Haastrup P, Bamps C. 2007. A pan-European River and Catchment Database. EUR 22920 EN. Luxembourg (Luxembourg).

[74] Voulvoulis et al., 2004: Removal of Organotins during Sewage Treatment: A Case Study. Env. Tech., http://dx.doi.org/10.1080/09593330.2004.9619363

[75] Wang and Kannan, 2018, Inventory, loading and discharge of synthetic phenolic antioxidants and their metabolites in wastewater treatment plants. Water Research, https://doi.org/10.1016/j.watres.2017.11.028

[76] Wen, Y., Schoups G., van de Giesen, N. 2017. Organic pollution of rivers: combined threats of urbanization, livestock farming and global climate change. Scientific Reports 7: 43289. Doi: 10.1038/srep43289

[77] Zhang et al. (2016) Removal of Cyclic and Linear Siloxanes in Effluents from a Cosmetic Wastewater Treatment Plant by Electrochemical Oxidation.Int. J. Electrochem. Sci., 11 (2016) 6914 – 6921

[78] Moelants, N., Smets, I., Dekort, B., Van den Broeck, R., Van Impe, J.F., 2006a. Towards an optimal and sustainable performance of small scale wastewater treatment systems. In: Proceedings of the conference on Biofilm Systems VI, Amsterdam RAI, The Netherlands, 24–27 September 2006, 77–86 (10p).

[79] Moelants, N., Smets, I.Y., Van Impe, J.F., 2006b. Onderhoudsvereisten en duurzame performantie van individuele waterzuiveringssystemen: enqueteresultaten. Afvalwaterwetenschap, 4, 212–221.

[80] Moelants, N., Janssen, G., Smets, I., and Van Impe, J., 2008. Field performance assessment of onsite individual wastewater treatment systems. Water Science & Technology, 58(1), 1-6. https://doi.org/10.2166/wst.2008.325

[81] Van Tomme, I., Pieters, J., Verstraete, W., 2001. Individual wastewater treatment in Flanders (Belgium). Meded Rijksuniv Gent Fak Landbouwkd Toegep Biol Wet, 66(3a), 205-9.

[82] Venier S. (2018) Waste Water Management in Seaside Tourism Areas: The Rimini Seawater Protection Plan. In: Gilardoni A. (eds) The Italian Water Industry. Springer, Cham

## List of abbreviations and definitions

REFIT Regulatory Fitness and Performance

UWWTD urban wastewater treatment Directive

WWT(P) wastewater treatment (plant)

BOD ($BOD_5$) biochemical oxygen demand (5-days biochemical oxygen demand)

N nitrogen

P phosphorus

TSS Total Suspended Sediments (solids)

CSO Combined Sewer Overflow

DWD Drinking Water Directive

PAH Polycyclic Aromatic Hydrocarbons

PFA Per-Fluorinated Acids

WFD Water Framework Directive

EQS(D) Environmental Quality Standards (Directive)

BWD Bathing Waters Directive

BHT Butylhydroxytoluene

EHMC 2-ethylhexyl-4-methoxycinnamate

GREEN Geospatial Regression Equation for European Nutrient Losses

TOC Total Organic Carbon

EEA European Environment Agency

D4 Octamethylcyclotetrasiloxane

E1 Estrone

E2 17b-estradiol

EE2 Ethinylestradiol

LSU Livestock Standard Unit

CFU Colony Forming Units

PE Population Equivalent

HELCOM Helsinki Commission

NUTS2 nomenclature unités territoriales statistiques / level 2 aggregation (regions within European countries)

NUTS1 nomenclature unités territoriales statistiques / level 1 aggregation (regions within European countries)

NP Nonylphenol

## List of figures

## List of tables

# Annexes

## Annex I. Estimation of the impact of inland sources of faecal coliforms on marine coastal waters.

The transport of coliforms in marine coastal waters can be described by numerically solving the advection-dispersion-reaction equation (ADRE; see e.g. Csanady, 1970):

$$\frac{\partial C}{\partial t} = \bar{\bar{D}}\nabla^2 C - \bar{U}\nabla C - k_c C$$

Where C is coliform concentration, $k_c$ is the coliform decay constant in coastal waters, $\bar{U}$ the velocity field and $\bar{\bar{D}}$ is the diffusion/dispersion tensor. Solving the ADRE entails detailed description of the bathimetry, velocity field and calibration of the dispersion tensor in coastal waters, which is apparently beyond the scope of this assessment. However, the analytical solutions available for this equation under idealized conditions allow a first screening-level modelling of coastal concentrations, usually sufficient to define the orders of magnitude of the problem and to appreciate the variation of conditions across Europe.

If we assume full vertical mixing of coastal waters, when a strong current dominates, we may assume $\bar{\bar{D}}$=0. If the velocity field is one-dimensional, uniform and stationary, a steady state solution of the ADRE is:

$$C = C_0 \exp\left(-\frac{k_c x}{u}\right)$$

which gives the concentration of coliforms as a function of the abscissa along the line of a current having velocity u. This "advection only" solution provides the upper limit of concentrations. At the opposite extreme, when $\bar{U}$=0, the transport of coliforms is only due to dispersion. A steady state solution of the ADRE is, under full vertical mixing is

$$C = C_0 \mathrm{f}\left(\sqrt{\frac{k_c}{D}}x\right)$$

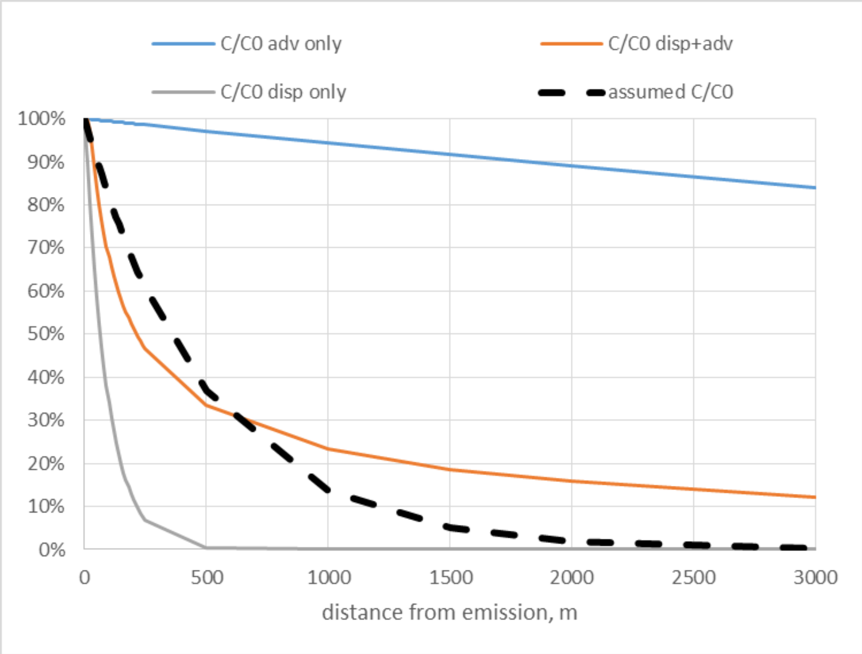
where f is a function depending on the geometry (a modified Bessel function of the second kind for radial propagation, and a simple negative exponential for the case of parallel (one-dimensional) propagation - see e.g. Thibodeaux, 1996, pp 486-487; Pistocchi, 2005).

When both advection and dispersion play a role, an analytical steady state solution under full vertical mixing, assuming an initial width b of the emission source, is:

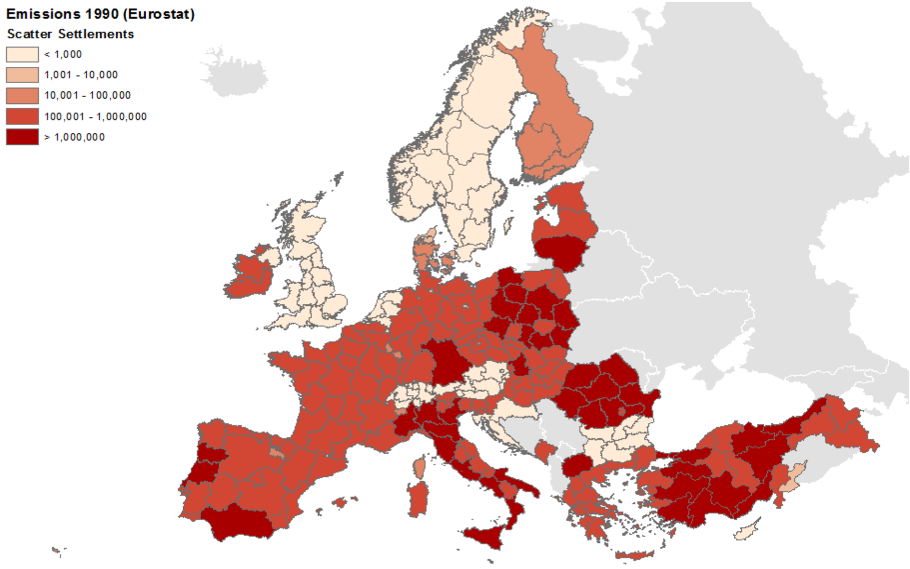
$$C = C_0 \exp\left(-\frac{k_c x}{u}\right)\left(erf\left(\frac{\frac{b}{2}}{\sqrt{\frac{4Dx}{u}}}\right)\right)$$

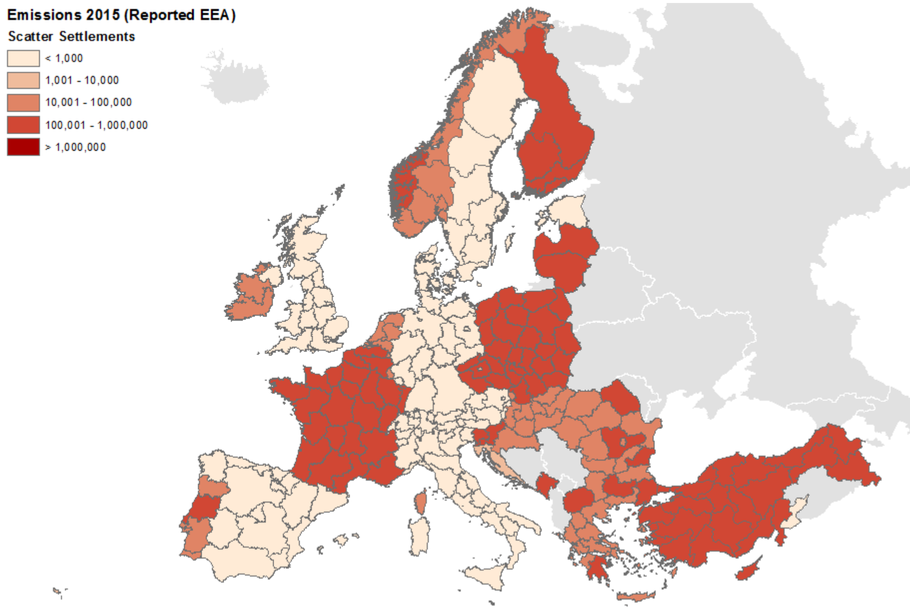
In marine coastal waters, we expect coliforms to die-off following a $k_c$ in or above the upper range of the values predicted by the Mancini equation (i.e. $k_c$ =0.5 $d^{-1}$), while the dispersion coefficients are in the order of D =0.05 $m^2$ $s^{-1}$, representative of the range typically observed (e.g. Riddle and Lewis, 2000; Csanady, 1970). A conservative value of coastal current velocity may be u=0.1 m/s. With these parameters and b=20 m, the three analytical solutions are plotted for comparison in Figure 70. The graph shows a plausible distribution of coliform concentrations downstream of the emission point, along the trajectory of the current passing through the centre of the emission zone. The dispersion-only solution is not sufficiently conservative, while the advection-only one seems excessively conservative. The advection-dispersion solution is fairly intermediate, and an exponential decay with a constant $\alpha$=0.002 $m^{-1}$ (also shown in the graph) provides a more conservative estimate at shorter distances, while showing a faster decrease at further distance (where the assumptions of constant dispersion and velocity appear less and less realistic, hence excessively conservative). Based on these considerations, this exponential decay is assumed as a pragmatic representation of coliform transport in coastal waters.

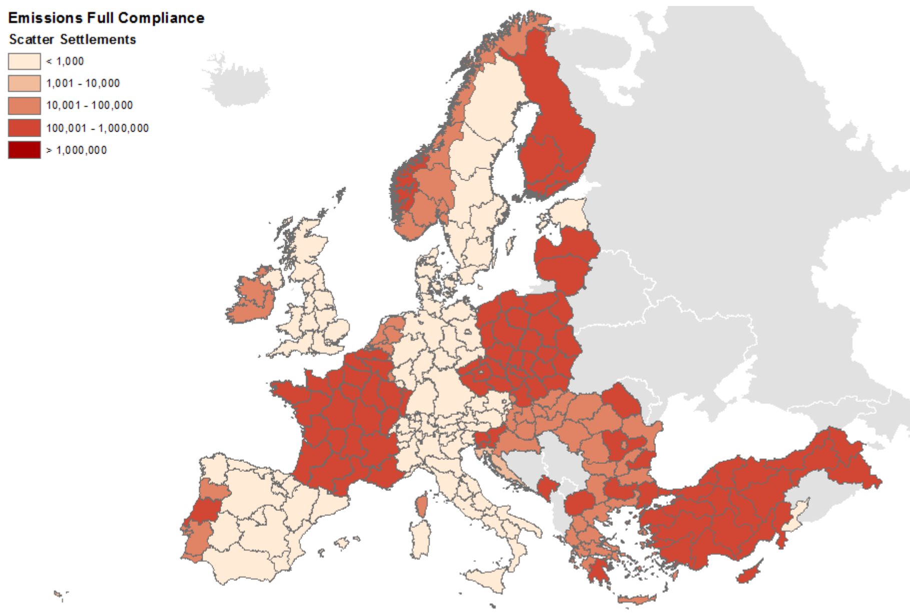
*Figure 70 - decrease of concentration from the emission point, following different simplified dispersion schemes.*

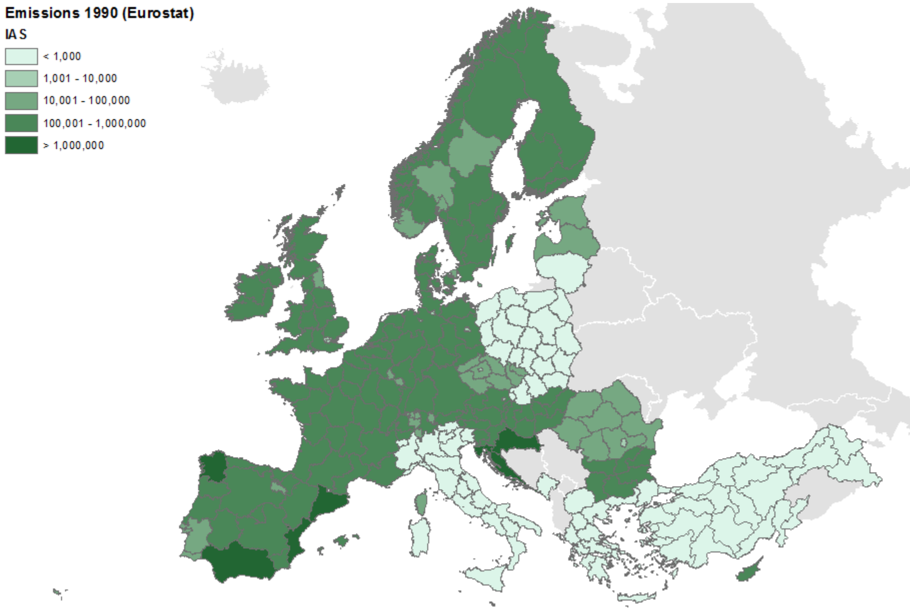


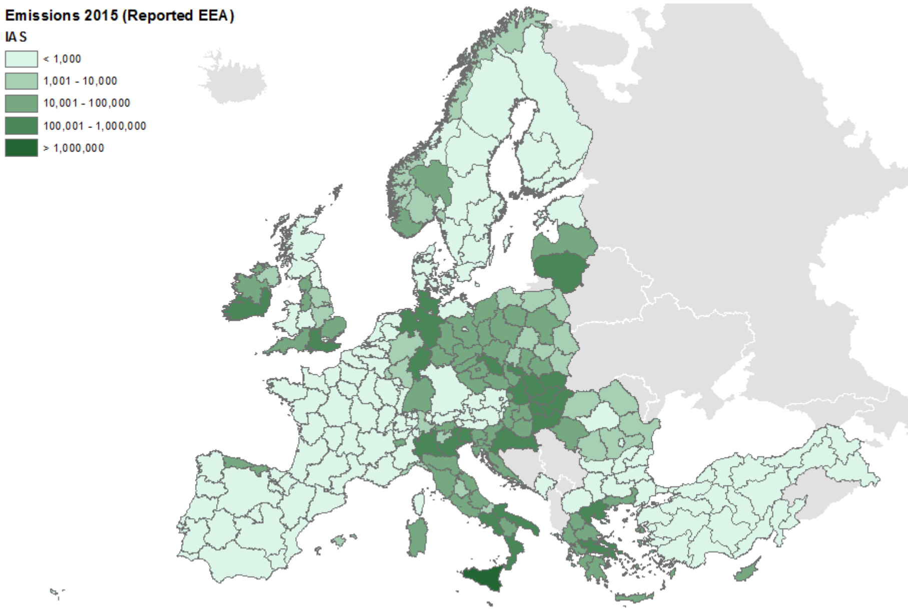
**Annex II. Spatial distribution of emissions (in P.E.) by source type (point/diffuse), treatment level and scenario, and summary of loads by country**

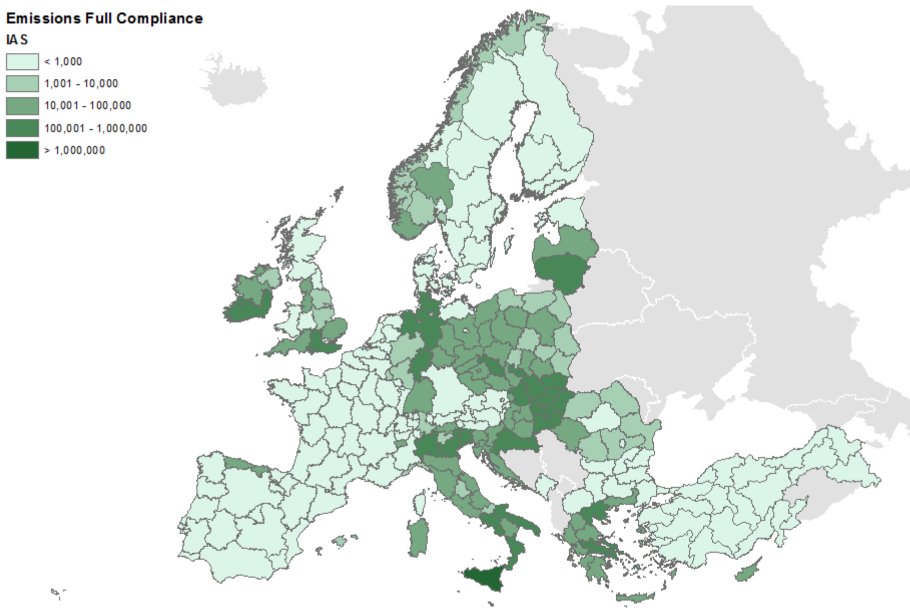


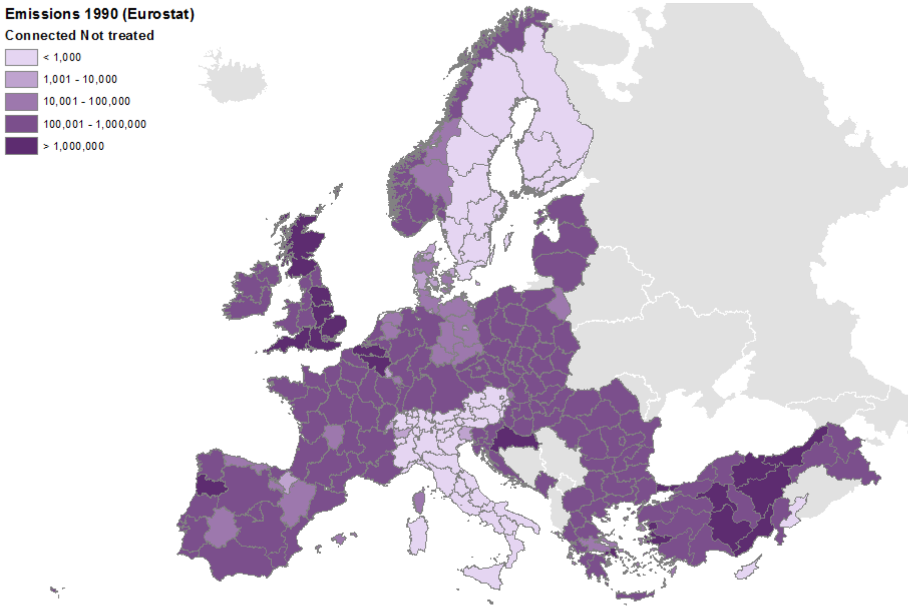



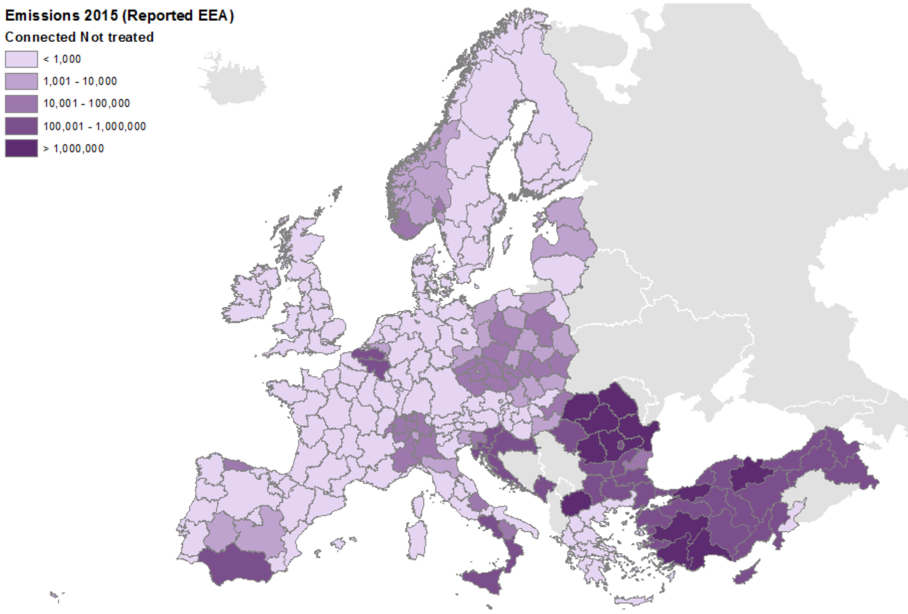



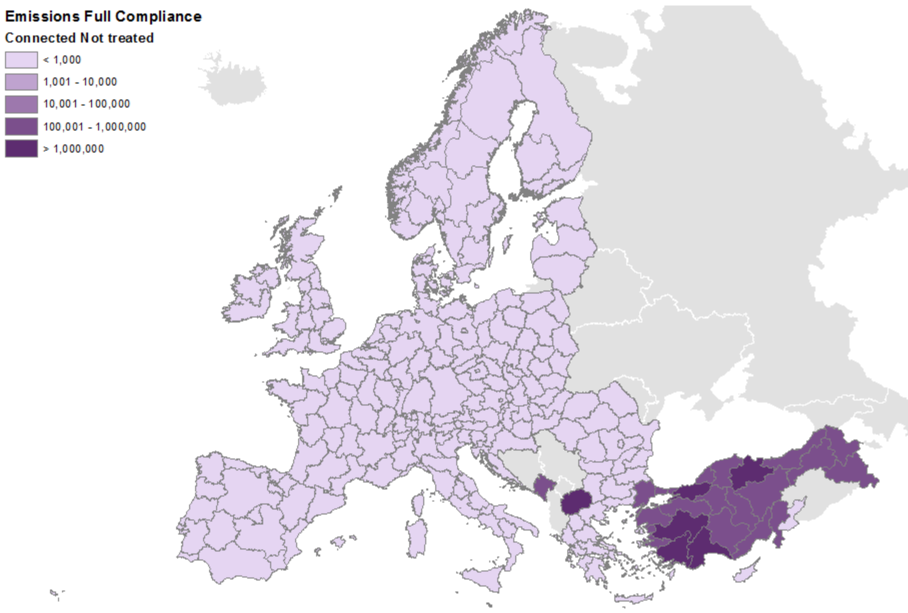

Emissions 1990 (Eurostat)
IAS
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000

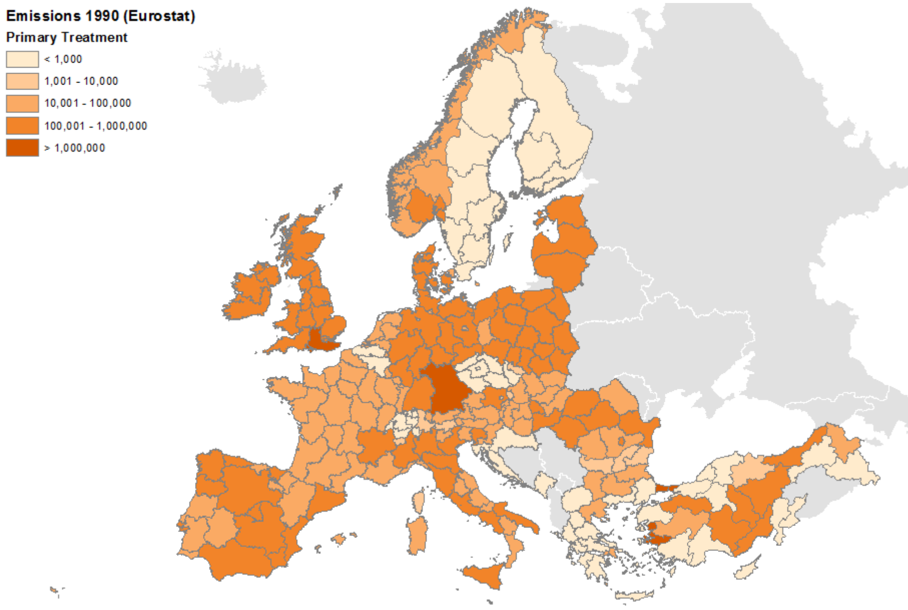

Emissions 2015 (Reported EEA)
IAS
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000

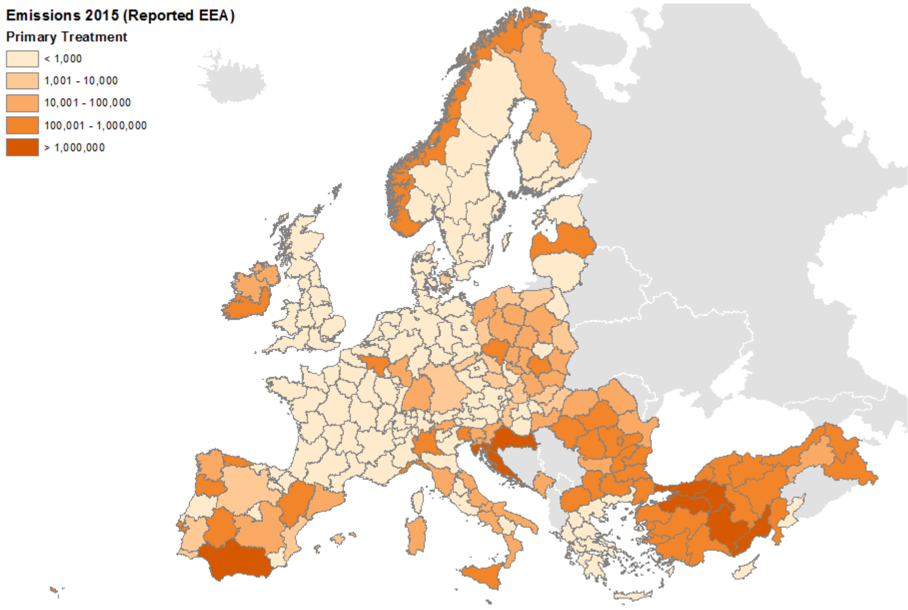

Emissions Full Compliance
IAS
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000

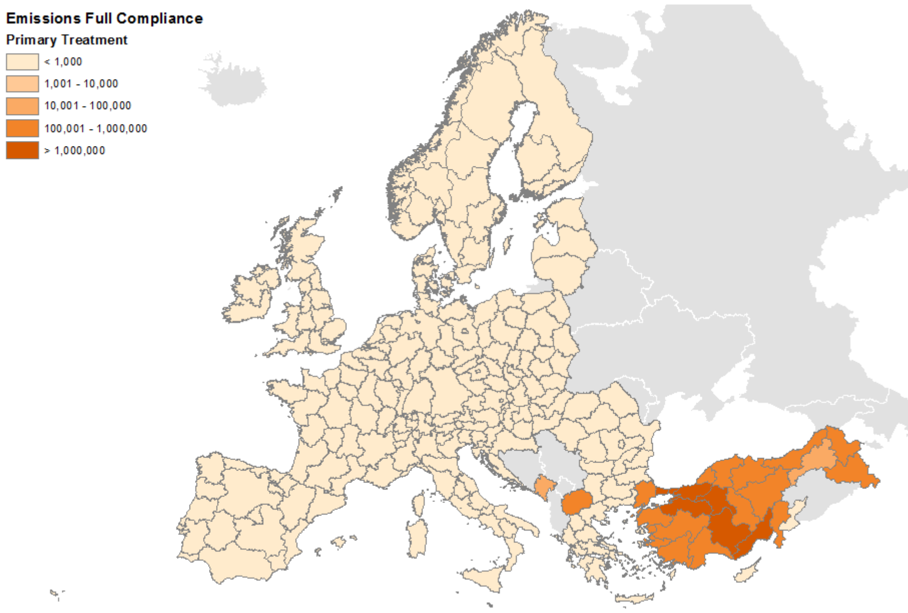

Emissions 1990 (Eurostat)
Connected Not treated
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000

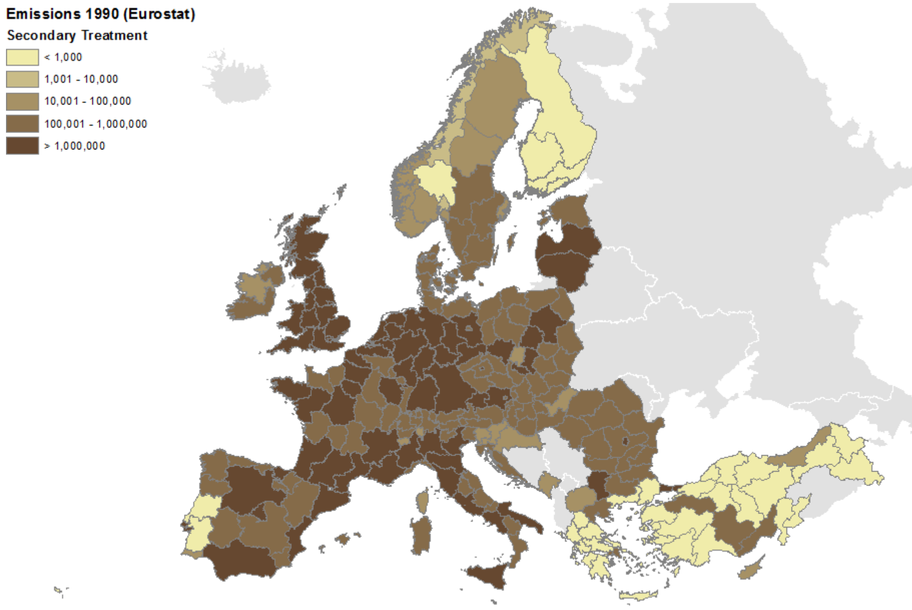

Emissions 2015 (Reported EEA)
Connected Not treated
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000

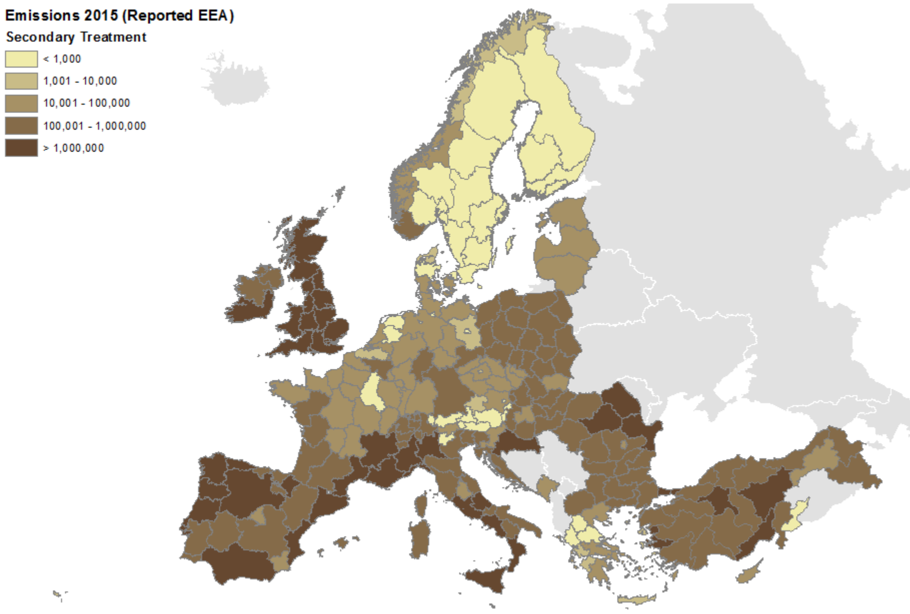

Emissions Full Compliance
Connected Not treated
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000

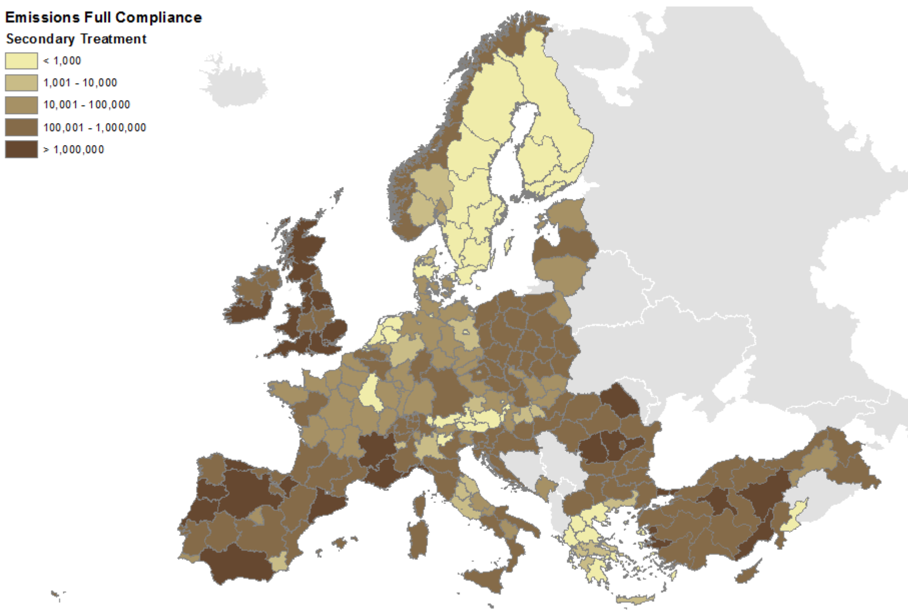

Emissions 1990 (Eurostat)
Primary Treatment
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000

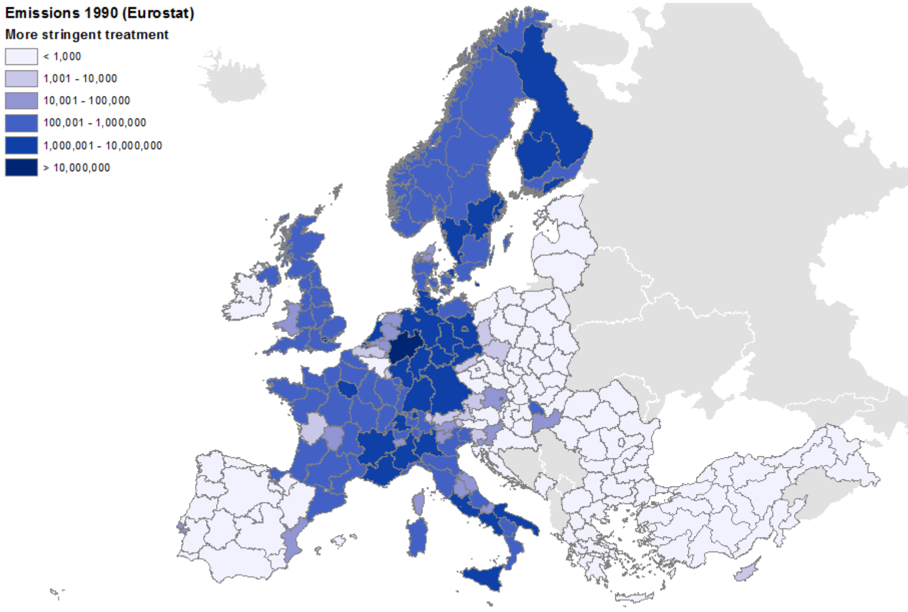

Emissions 2015 (Reported EEA)
Primary Treatment
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000

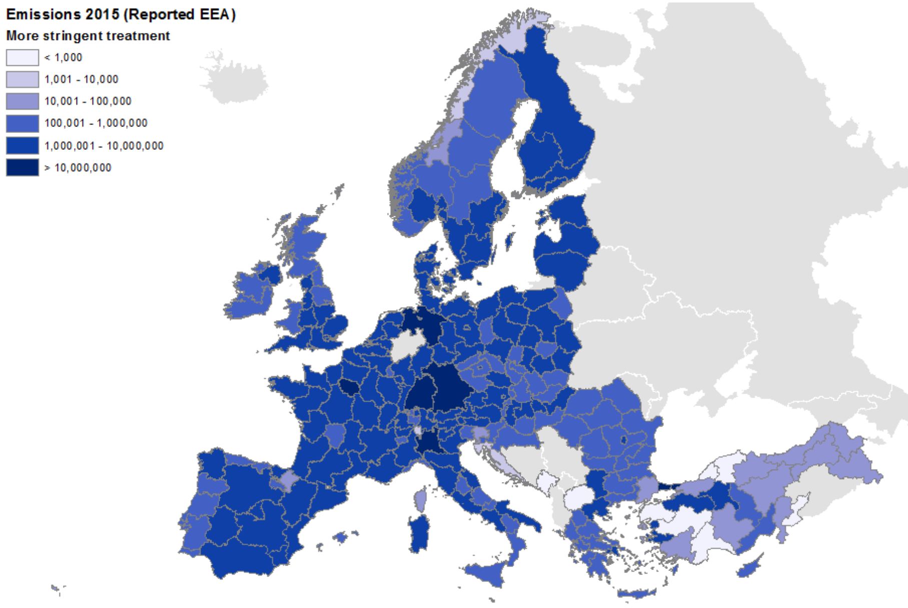

Emissions Full Compliance
Primary Treatment
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000


Emissions 1990 (Eurostat)
Secondary Treatment
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000


Emissions 2015 (Reported EEA)
Secondary Treatment
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000


Emissions Full Compliance
Secondary Treatment
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
> 1,000,000


Emissions 1990 (Eurostat)
More stringent treatment
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
1,000,001 - 10,000,000
> 10,000,000


Emissions 2015 (Reported EEA)
More stringent treatment
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
1,000,001 - 10,000,000
> 10,000,000

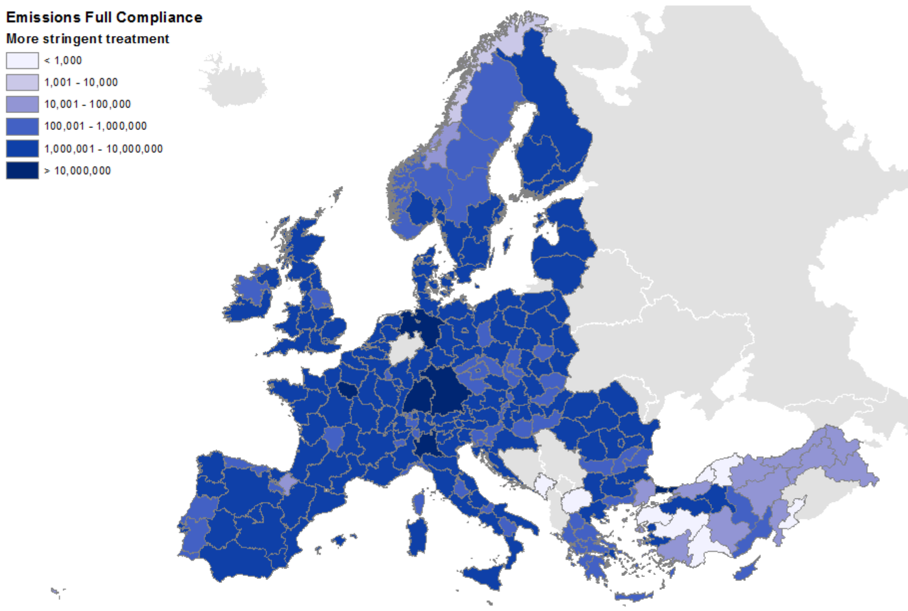

Emissions Full Compliance
More stringent treatment
< 1,000
1,001 - 10,000
10,001 - 100,000
100,001 - 1,000,000
1,000,001 - 10,000,000
> 10,000,000

*Table 17 – summary by country of loads of N, P and BOD under present and full compliance scenarios*

| country | BOD t/y | | | N t/y | | | P t/y | | |
|---|---|---|---|---|---|---|---|---|---|
| | baseline | pre-directive | full compliance | baseline | pre-directive | full compliance | baseline | pre-directive | full compliance |
| AT | 17,851 | 41,148 | 17,850 | 14,391 | 16,710 | 14,391 | 1,184 | 2,574 | 1,184 |
| BE | 43,058 | 162,462 | 22,713 | 12,472 | 30,235 | 10,743 | 1,505 | 4,806 | 1,194 |
| BG | 50,873 | 112,600 | 15,171 | 10,838 | 19,890 | 6,287 | 1,802 | 3,652 | 809 |
| CY | 9,849 | 9,442 | 5,270 | 1,643 | 1,753 | 1,336 | 505 | 566 | 391 |
| CZ | 44,283 | 97,616 | 28,510 | 11,033 | 22,036 | 9,507 | 2,035 | 4,796 | 1,652 |
| DE | 111,972 | 299,057 | 98,470 | 80,051 | 118,704 | 77,775 | 7,455 | 23,579 | 7,042 |
| DK | 10,192 | 23,080 | 10,172 | 8,319 | 9,250 | 8,316 | 767 | 1,768 | 767 |
| EE | 1,564 | 15,602 | 1,426 | 1,053 | 3,656 | 1,040 | 107 | 671 | 104 |
| ES | 123,757 | 336,843 | 64,678 | 77,499 | 93,772 | 61,522 | 16,621 | 21,364 | 14,100 |
| FI | 17,482 | 19,748 | 17,005 | 6,787 | 7,037 | 6,690 | 785 | 1,683 | 776 |
| FR | 143,608 | 343,838 | 142,602 | 79,986 | 120,817 | 77,845 | 14,835 | 25,713 | 14,441 |
| GB | 84,859 | 388,090 | 77,082 | 86,312 | 119,220 | 76,749 | 17,086 | 27,531 | 16,594 |
| GR | 31,433 | 156,859 | 20,228 | 13,430 | 30,915 | 11,784 | 3,910 | 6,641 | 3,625 |
| HR | 43,268 | 75,228 | 5,277 | 9,101 | 12,314 | 3,752 | 2,811 | 3,921 | 1,794 |
| HU | 30,932 | 145,760 | 14,851 | 10,919 | 23,510 | 8,626 | 1,917 | 5,071 | 1,623 |

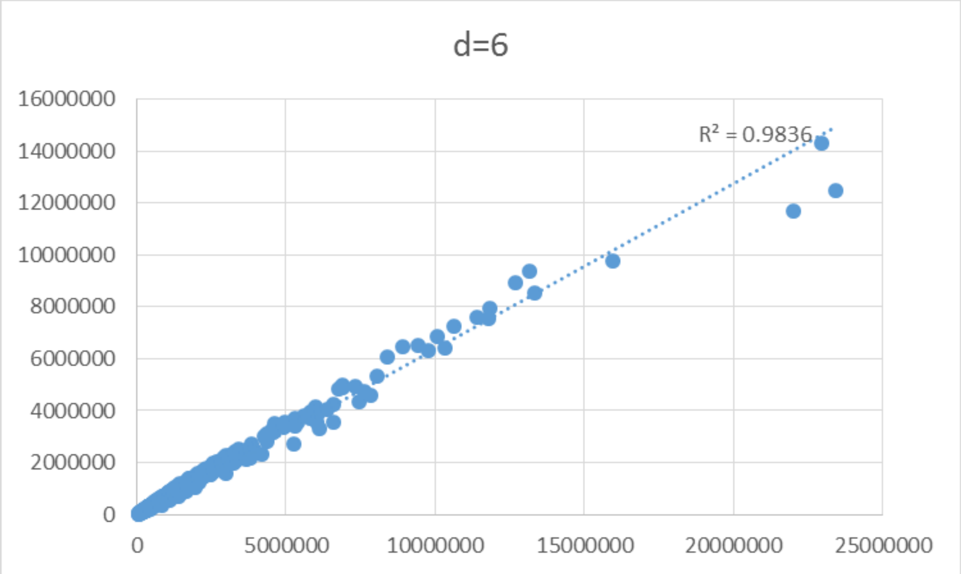
| country | BOD t/y | | | N t/y | | | P t/y | | |
|---|---|---|---|---|---|---|---|---|---|
| | baseline | pre-directive | full compliance | baseline | pre-directive | full compliance | baseline | pre-directive | full compliance |
| IE | 13,428 | 44,325 | 7,689 | 8,853 | 10,269 | 6,036 | 1,189 | 1,515 | 1,105 |
| IT | 143,605 | 346,133 | 71,589 | 89,919 | 110,099 | 64,255 | 13,323 | 18,030 | 10,761 |
| LT | 12,757 | 36,564 | 11,475 | 4,659 | 10,823 | 4,402 | 632 | 1,849 | 581 |
| LU | 747 | 1,849 | 700 | 543 | 810 | 533 | 51 | 120 | 50 |
| LV | 11,633 | 26,315 | 8,867 | 2,906 | 6,136 | 2,581 | 465 | 1,133 | 398 |
| MT | 461 | 6,899 | 450 | 392 | 1,208 | 370 | 184 | 298 | 184 |
| NL | 16,865 | 36,538 | 16,754 | 13,189 | 26,712 | 13,076 | 1,111 | 3,668 | 1,028 |
| PL | 146,139 | 421,434 | 131,179 | 51,566 | 91,949 | 47,323 | 11,806 | 26,063 | 10,296 |
| PT | 30,209 | 150,068 | 15,799 | 20,954 | 31,625 | 14,341 | 4,554 | 7,362 | 4,176 |
| RO | 245,615 | 305,193 | 31,881 | 51,943 | 63,008 | 24,835 | 8,577 | 10,392 | 5,066 |
| SE | 10,973 | 22,996 | 10,973 | 9,053 | 10,357 | 9,053 | 795 | 2,645 | 795 |
| SI | 18,956 | 30,151 | 11,806 | 4,523 | 5,995 | 3,392 | 702 | 986 | 582 |
| SK | 35,194 | 48,036 | 26,557 | 7,289 | 8,699 | 5,828 | 1,355 | 1,724 | 1,005 |
| **total** | **1,451,561** | **3,703,875** | **887,022** | **689,626** | **1,007,511** | **572,386** | **118,069** | **210,118** | **102,123** |

## Annex III. Additional information on the CSO model

*Dependence of CSO loads on dilution rates and allowed number of spill events.*

We used the model of Equation 8 and Equation 9 with values of d=6 and d=10 and we compared the results to those in the case d=4 considered as the bottom line of this study (§2.2.4). The results indicated a very good correlation (Figure 71) and a consistent decrease of the CSO load (slope of the zero-intercept best fit line of the scatter plots) as shown in Table 18.

*Figure 71 – comparison of loads (Equation 9) with d=4 (x-axis) and d=6, d=10 (y-axis). Loads are in yearly m3 of wastewater equivalent (each dot in the scatter plot represents a FUA).*



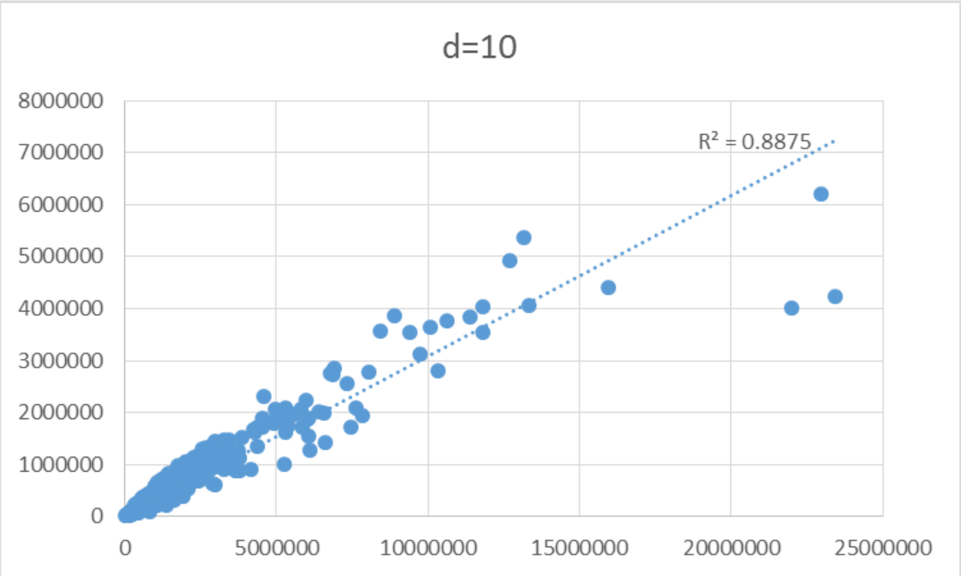




*Table 18 – reduction of CSO loads from the case of d=4, as a function of d.*

| d | % of load (Equation 9) |
|---|---|
| 4 | 100 |
| 6 | 65 |
| 10 | 33 |

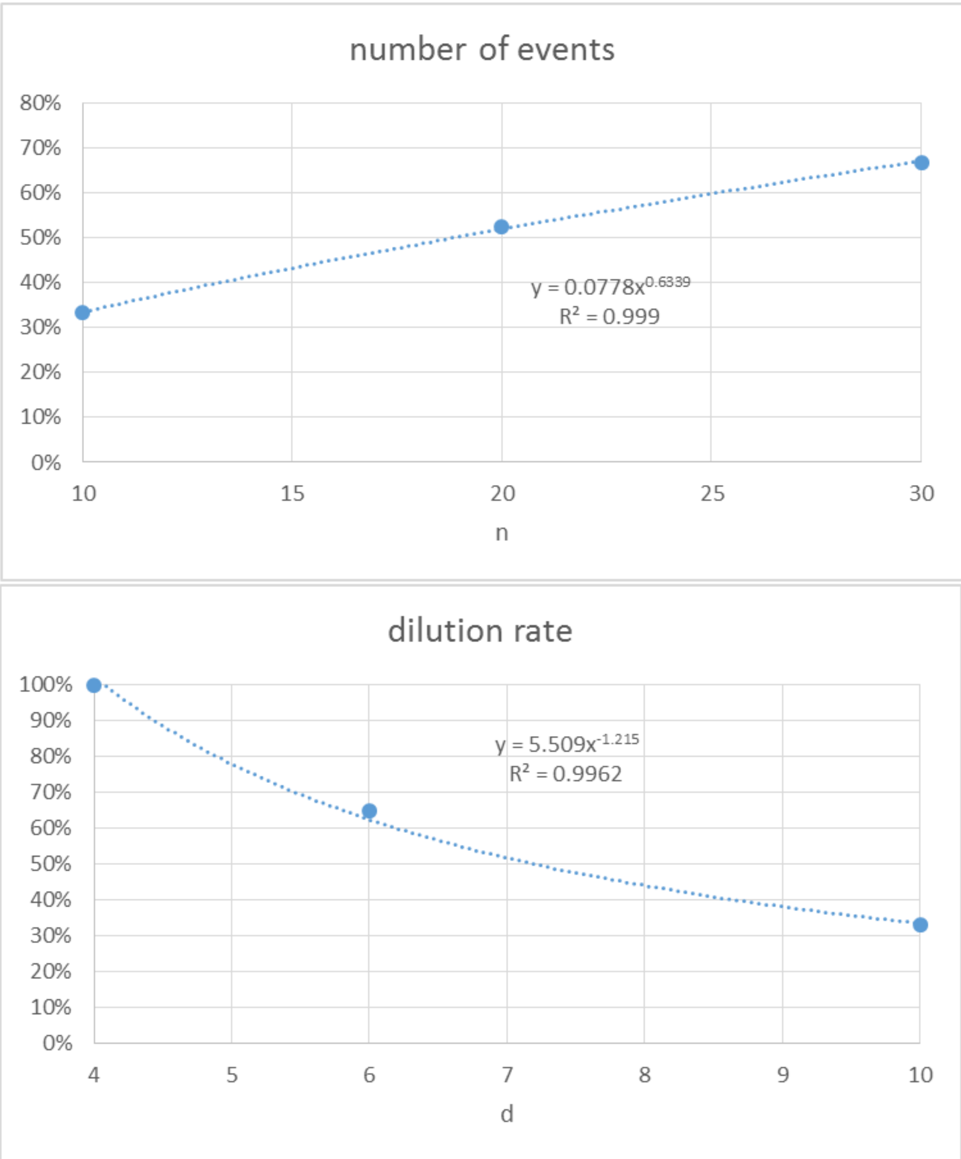
Similarly, we compared the total annual CSO load with the load corresponding to the top *n* most severe events (an event break being defined by at least 3 hours with null precipitation). We considered n=160, n=320 and n=480, corresponding to 10, 20 and 30 events per year on average over a series of 16 years. The comparison highlighted a consistent pattern (Table 19).

*Table 19 – percent of annual CSO load due to the average yearly top n events (n=10, 20, 30)*

| Number of top events | % of annual CSO load |
|---|---|
| 10 | 33 |
| 20 | 53 |
| 30 | 67 |

The values derived from the comparisons were used to build the reduction coefficient functions shown in Figure 72.

*Figure 72 – change in the wastewater equivalent of combined sewer overflows as a function of the number of events sorted by decreasing volume (n) and of the dilution rate (d), with 100% corresponding to d=4 and n= total number of events.*




*Share of combined sewer networks in European Union Member States.*

Information on the share of sewer networks which is combined was compiled for the purposes of this assessment from the available sources, and particularly from Milieu, 2016 (Table 20).

*Table 20 – percentage of separate sewers by country or region in the EU, considered in the calculations.*

| code | name | % separate network | Source |
|---|---|---|---|
| AT | Austria | 72% | Milieu, 2016 |
| BE1 | Bruxelles /Brussels | 0% | Milieu, 2016 |
| BE2 | Flanders | 13% | Milieu, 2016 |
| BE3 | Wallonia | 10% | Milieu, 2016 |
| BG | Bulgaria | 100% | assumed from Milieu, 2016 |
| CY | Cyprus | 0% | Milieu, 2016 |
| CZ | Czech Republic | 100% | Milieu, 2016 |
| DEA | North Rhein Westphalen | 38% | DESTATIS |
| DE2 | Bavaria | 34% | DESTATIS |
| DE1 | Baden-Wurttemberg | 19% | DESTATIS |
| DEB | Rheinland Pfalz | 21% | DESTATIS |
| DEC | Saarland | 8% | DESTATIS |
| DE7 | Hessen | 13% | DESTATIS |
| DEG | Thuringen | 25% | DESTATIS |
| DED | Sachsen | 53% | DESTATIS |
| DE3 | Berlin | 74% | DESTATIS |
| DE4 | Brandenburg | 96% | DESTATIS |
| DE8 | Mecklenburg-Vorpommern | 94% | DESTATIS |
| DE9 | Niedersachsen | 93% | DESTATIS |
| DE6 | Hamburg | 69% | DESTATIS |
| DE5 | Bremen | 59% | DESTATIS |
| DEE | Sachsen-Anhalt | 78% | DESTATIS |
| DEF | Schlesweg-Holstein | 89% | DESTATIS |
| DK | Denmark | 50% | Milieu, 2016 |
| EE | Estonia | 50% | assumed: new structures are separate (Milieu, 2016) |
| EL30 | Attica | 97% | Milieu, 2016 (data for Athens) |
| EL | Greece outside Attica and Central Macedonia | 21% | Milieu, 2016 |
| EL52 | Central Macedonia | 65% | Milieu, 2016 (data for Thessaloniki) |
| ES | Spain | 87% | national average (Milieu, 2016) |
| FI | Finland outside Helsinki | 95% | national average (Milieu, 2016) |
| FI1B | Helsinki-Uusimaa | 70% | Milieu, 2016 |
| FR | France | 68% | Milieu, 2016 |
| HR | Croatia | 41% | Milieu, 2016 |

| code | name | % separate network | Source |
|---|---|---|---|
| HU10 | Budapest | 38% | Milieu, 2016 |
| HU | Hungary outside Budapest | 97% | Milieu, 2016 |
| IE | Ireland | 76% | Milieu, 2016 |
| IT | Italy | 30% | assumed |
| LT00 | Lithuania | 50% | Milieu, 2016 |
| LU00 | Luxembourg | 10% | Milieu, 2016 |
| LV00 | Latvia | 50% | assumed |
| MT00 | Malta | 0% | Milieu, 2016 |
| NL | Netherlands | 27% | Milieu, 2016 |
| PL | Poland | 8% | Milieu, 2016 |
| PT | Portugal | 66% | Milieu, 2016 |
| RO | Romania | 100% | according to legislation (Milieu, 2016) |
| SE | Sweden | 88% | Milieu, 2016 |
| SI | Slovenia | 41% | Milieu, 2016 |
| SK | Slovakia | 92.50% | Milieu, 2016 |
| UK | United Kingdom | 30% | Milieu, 2016 |

## Annex IV. Assumptions on scenarios of IAS treatment and "What-if-no-directive" (WIND)[105]

*Assessment of the effectiveness of IAS in different European Union Member States*

Moelants et el. (2006a, 2006b, 2008) and Van Tomme et al. (2001) highlight that the theoretical installed treatment performance of any small scale WWT system may gradually deteriorate if not properly managed and maintained. Furthermore, Moelants et al. (2008) report the results from a field study conducted in Belgium between October 2006 and March 2007, where the field performance of 23 randomly selected small scale wastewater treatment systems was assessed. The installations were located in Flanders (17) and the Walloon region (6). The majority of the system owners (74%) were satisfied with their system operation, but the performance results demonstrated that their satisfaction did not correspond to the actual treatment conditions achieved. The results of this study also indicated that a certificate alone was not sufficient to guarantee a fair performance of the system. The performance of certified compact systems without a maintenance contract was not better than other systems.

In order to evaluate the situation in the different EU Member States, evidence has been collected on the following aspects:

— What is the typical treatment level of actually installed IAS?

— Is there a specific regulatory framework for the use and design of IAS?

— Does this specific regulatory framework address the management of IAS (e.g. registering, permitting, monitoring, maintaining, inspecting)?

— Is there a clear enforcement mechanism to ensure proper use of IAS provisions?

— Are there clear incentives for using IAS only where it is appropriate and/or promote proper operation of installed IAS?

An equivalent treatment level of IAS has been attributed to each Member State as follows. A positive answer to all questions from — to — above qualifies the IAS to match the treatment level at installation (secondary, primary or more stringent depending on the country). One or more negative answers to the questions triggers an assumption of reduced effectiveness, on the basis of expert judgment. Absence of enforcement mechanisms and incentives weighs more when deciding how much to reduce the effectiveness, while the existence of specific legal and regulatory provisions for IAS is judged less important in driving their performance, also considering that the majority of the Member States have some type of legal framework.

*Definition of the "What-if-no-Directive" (WIND) scenario*

For the WIND modelling, the Member States were classified in 3 groups based on the level of treatment before entry into force of the Directive or accession to the EU; it is assumed that there would be more or less impacts from the Directive depending on the antecedent level of treatment. The following table summarizes the weights adopted in the two scenarios "WIND1" and "WIND2".

[105] This Annex was edited on the basis of material prepared by external consultants in the context of support received from NTUA and CENIA part of the project team from DG Environment led by Wood E&I UK Ltd contract reference 070201/2018/775193/ETU/ENV.C.2 on 'Service request supporting the evaluation of Directive 91/271/EEC concerning urban waste water treatment'. The weights in the "What-if-no-directive" (WIND) scenarios were defined by the consultants and taken as input for the modelling presented in this report.

*Table 21 – level of wastewater treatment in the EU Member States before entry into force of the Directive (or accession to the EU).*

| Reference date | Country code | Treated population (%) | Treatment level [106] | Comments and Sources | weights (WIND1) | weights (WIND2) |
|---|---|---|---|---|---|---|
| before 1995 | AT | 72 | HIGH | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 1 | 1 |
| before 1995 | BE | 28.9 | LOW | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0 | 0 |
| before 1995 | DE | 86.1 | HIGH | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 1 | 1 |
| before 1995 | DK | 85.4 | HIGH | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 1 | 1 |
| before 1995 | ES | 48.1 | MEDIUM | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0 | 0 |
| before 1995 | FI | 76 | HIGH | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 1 | 1 |
| before 1995 | FR | 69 | MEDIUM | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0.5 | 0.5 |
| before 1995 | EL | 11.4 | LOW | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0 | 0 |
| before 1995 | IE | 44 | MEDIUM | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0 | 0 |
| before 1995 | IT | 63 | MEDIUM | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0 | 0 |
| before 1995 | LU | 90.4 | HIGH | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0.5 | 0.5 |
| before 1995 | NL | 94 | HIGH | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 1 | 1 |
| before 1995 | PT | 20.9 | LOW | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0 | 0 |
| before 1995 | SE | 86 | HIGH | earliest possible value for the period 1990-1995 / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 1 | 1 |

[106] Low: <35% / Medium: 35-70% / High: >70%

| Reference date | Country code | Treated population (%) | Treatment level [106] | Comments and Sources | weights (WIND1) | weights (WIND2) |
|---|---|---|---|---|---|---|
| before 1995 | UK | 79 | HIGH | earliest possible value for the period 1990-1995 / Source: Eurostat [env_ww_con: Population connected to wastewater treatment plants]: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_ww_con&lang=en | 0.5 | 0.5 |
| 2004 | CY | 28.4 | LOW | value for 2004 or first earliest year / Source: Eurostat [env_ww_con: Population connected to wastewater treatment plants]: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_ww_con&lang=en | 0 | 0 |
| 2004 | CZ | 71.1 | HIGH | value for 2004 or first earliest year / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0.5 | 0.25 |
| 2004 | EE | 72 | HIGH | value for 2004 or first earliest year / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0.5 | 0.25 |
| 2004 | HU | 58.1 | MEDIUM | value for 2004 or first earliest year / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0.5 | 0.25 |
| 2004 | LT | 59.1 | MEDIUM | value for 2004 or first earliest year / Source: Eurostat [env_ww_con: Population connected to wastewater treatment plants]: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_ww_con&lang=en | 0.5 | 0.25 |
| 2004 | LV | 66.1 | MEDIUM | value for 2004 or first earliest year / Source: Eurostat [env_ww_con: Population connected to wastewater treatment plants]: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_ww_con&lang=en | 0.5 | 0.25 |
| 2004 | MT | 13.3 | LOW | value for 2004 or first earliest year / Source: Eurostat [env_ww_con: Population connected to wastewater treatment plants]: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_ww_con&lang=en | 0 | 0 |
| 2004 | PL | 59 | MEDIUM | value for 2004 or first earliest year / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0.5 | 0.25 |
| 2004 | SI | 48.4 | MEDIUM | value for 2004 or first earliest year / Source: Eurostat [env_ww_con: Population connected to wastewater treatment plants]: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_ww_con&lang=en | 0.5 | 0.25 |
| 2004 | SK | 54.1 | MEDIUM | value for 2004 or first earliest year / Source: OECD Wastewater treatment indicator: https://data.oecd.org/water/waste-water-treatment.htm | 0.5 | 0.25 |
| 2007 | BG | 42.3 | MEDIUM | value for 2007 or first earliest year / Source: Eurostat [env_ww_con: Population connected to wastewater treatment plants]: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_ww_con&lang=en | 0 | 0 |

| Reference date | Country code | Treated population (%) | Treatment level [106] | Comments and Sources | weights (WIND1) | weights (WIND2) |
|---|---|---|---|---|---|---|
| 2007 | RO | 28.3 | LOW | value for 2007 or first earliest year / Source: Eurostat [env_ww_con: Population connected to wastewater treatment plants]: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_ww_con&lang=en | 0 | 0 |
| 2013 | HR | 52.9 | MEDIUM | value for 2013 or first earliest year / Source: Eurostat [env_ww_con: Population connected to wastewater treatment plants]: http://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=env_ww_con&lang=en | 0.5 | 0.25 |

KJ-NA-30003-EN-N

## The European Commission's science and knowledge service

Joint Research Centre

### JRC Mission

As the science and knowledge service of the European Commission, the Joint Research Centre's mission is to support EU policies with independent evidence throughout the whole policy cycle.

**EU Science Hub**
ec.europa.eu/jrc

@EU_ScienceHub

EU Science Hub - Joint Research Centre

Joint Research Centre

EU Science Hub

Publications Office of the European Union

doi:10.2760/303163

ISBN 978-92-76-11263-1