Master's Thesis – Master Water Science and Management

Following the water: Investigating de facto wastewater reuse in the Netherlands

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ABSTRACT

De facto wastewater reuse is the practice of extracting from surface water bodies which are impacted by treated wastewater (TWW) for the purposes of agricultural irrigation, managed aquifer recharge or drinking water supply. This process may be responsible for the propagation of TWW related contaminants through the hydrological system but may conversely represent a valuable way of accessing fresh water. The extent to which surface water bodies in the Netherlands are impacted by TWW is poorly understood, and the distribution of de facto reuse even more so. This study aims to address these knowledge gaps, with a focus on de facto reuse through agricultural irrigation. This is achieved via a novel application of the Water Framework Directive (WFD)-Explorer water quality model paired with a spatial analysis step, allowing for the distribution of different flow components – namely TWW and flow from transboundary rivers – to be discerned for the national surface water network. When paired with data on surface water extractions for irrigation, this identifies notable areas of de facto reuse. Results show that during dry conditions, TWW is a significant flow component in many surface water bodies, particularly in smaller water bodies located close to WWTPs. De facto reuse is indicated as widespread, with several areas in which extractions make use of water from impacted surface water bodies. This study represents a first attempt to directly link TWW emissions to agricultural irrigation, highlighting a mechanism by which wastewater associated contaminants can propagate through the hydrological system. It is likely to be useful for water managers in assessing sources of water quality issues and better regulating de facto reuse.

ACKNOWLEDGEMENTS

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CONTENTS

1 INTRODUCTION

1.1 Background

Wastewater, following treatment processes, is generally discharged into surface water bodies, whereupon it is transported and diluted. In areas with high population densities, this leads to a large percentage of water courses being impacted by treated wastewater (TWW) (Coppens et al., 2015; Thebo et al., 2017). Indeed, a recent study by the EU Commission indicates that TWW is of a near ubiquitous presence in the surface water bodies of EU nations (Drewes at al., 2017). In many regions globally TWW discharge helps to maintain environmental flow capacities in surface water bodies (Luthy et al., 2015) and to recharge shallow aquifers (Hubbard et al., 2016). The contribution of contaminants from TWW effluent, however, may also have negative impacts on water quality and ecosystems due to the elevated levels of antecedent contaminants not removed during treatment processes. Wastewater treatment processes generally focus on removing solid waste, BOD and excess nutrients (Nitrates, Phosphates) from water but neglect disinfection or (ultra)filtration steps, leading to persistent levels of pathogens, heavy metals and contaminants of emerging concern (CECs) in effluent (Mateo-Sagasta et al., 2015).

De facto wastewater reuse is the practice of extracting from a surface water body which is heavily influenced by discharge from a wastewater stream (see Figure 1) and repurposing this for anthropogenic use (Drewes et al., 2017). In this form, TWW can constitute a valuable freshwater resource for agriculture (irrigation), industry (cooling purposes) and public services (drinking water), making use of ecosystem services provided by surface water bodies (Rietveld et al., 2011). The extent of its usefulness is, however, limited by its quality and the regulations associated with reuse (see Ricart & Rico, 2019). This practice therefore requires careful management. In the Netherlands, it is likely that wastewater is an important flow component in surface water bodies owing to high population densities and an extensive network of TWW treatment plants (WWTPs). This suggests that de facto reuse is prevalent, but this has largely gone uninvestigated to date.

In recent years, the EU have begun to incorporate water as a resource which should be managed as a key element of the circular economy (EU, 2015). This has led to increased efforts to formulate regulations on the direct reuse of TWW with an aim to reducing associated risks in terms of environmental contamination and health issues (JRC, 2017). The same cannot be said for de facto reuse, whereby the lack of knowledge on the prevalence and impacts of the practice are roadblocks to formulating and enacting appropriate regulations. Indeed, the distribution of TWW in the surface water network of the Netherlands and its subsequent de facto reuse are largely uncharacterised and regulations entirely absent. This research investigates this issue, with an aim to better informing water managers on the potential impacts of TWW on surface water bodies.

1.2 Previous work

A number of useful papers explore TWW reuse both globally and in the Netherlands. At a national scale, a review paper by Rietveld et al. (2011) suggests that although the Netherlands is a low water stressed country, TWW represents a useful resource for industry and agriculture in regions which experience issues of localised water scarcity. In general, literature points toward the direct reuse of TWW becoming a more common trend into the future both internationally (UNESCO, 2017) and in the EU (EU, 2015) if appropriate regulations can be devised (Jeong et al., 2016) and the societal "yuck" factor can be overcome (Ricart & Rico, 2019). These papers, however, generally neglect to include the utility of de facto reuse within water reuse schemes. Indeed, a general shortcoming in discussing de facto reuse at all in relation to this topic is evident within this body of literature. This is an important research gap as it is becoming more evident that this practice is widespread (see Drewes et al., 2017), outlining a need for a more integrated research approach toward the two practices.

Numerous studies have sought to characterise the influence of TWW on surface water bodies, in terms of both quantity and quality. These studies generally make use of river flow and TWW discharge data coming from publicly available sources to construct a general picture of surface water impaction over catchment to global scales. Drewes et al. (2017) use this data to determine the extents of TWW impaction for three

Figure 1 | Discharge of TWW from the Haaksbergen WWTP into the Bolscherbeek (right) with dry stream bed shown upstream (left). Taken in drought conditions experienced in 2018.

catchments within the EU, with an aim of characterizing "unplanned water reuse". Larger scale studies of this issue are also available; Thebo et al. (2017) interrogate population and hydrological data using GIS methods to identify impacted water bodies and subsequently areas in which de facto reuse occurs globally, with a specific focus on agricultural use in developing countries. Similarly, Rice & Westerhoff (2014) assess surface water impaction across the entire USA to determine the extent of TWW entering drinking water systems. All of these studies find surface water bodies to be impacted by TWW, with the degree of impaction particularly dependent on flow conditions. Pertinent to this research is a number of studies within the Netherlands which make use of a water quality model (WFD Explorer) fed by river and WWTP discharge data to determine how WWTP effluent impacts surface water quality in the Netherlands (Coppens et al., 2015; van Wezel et al., 2018). Both studies find that WWTP effluent exerts a significant impact on many surface water bodies in terms of water quality. Although these studies succeed in characterizing TWW impaction of surface water bodies, they rarely draw links between this and societal water uses, hence neglecting the assessment of the occurrence of de facto water reuse. This represents a key research gap.

In terms of methodologies to track the movement of TWW through surface and groundwater systems, an emerging body of literature seeks to identify effective tracer compounds for this purpose. The primary aim of this literature is to detect effluent leakages, but this is also highly pertinent to tracing TWW through surface water bodies and the subsurface. Gasser et al. (2014) suggest that chloride can be used as an effective tracer for large scale TWW emissions and is simple and easy to measure. Pharmaceuticals and artificial sweeteners are also recently identified as effective due to their conservative properties, and the fact that their primary source is domestic wastewater (Lim et al., 2017; Tran et al., 2014). These tracer methodologies are therefore evidently useful but have rarely been combined with the large-scale TWW impaction studies listed above. The application of these two methodologies together would represent a more robust approach to assess the presence of TWW in surface water bodies.

1.3 Problem definition

The Netherlands is likely to be an area in which de facto reuse is prevalent owing to high population densities and an extensive network of TWW treatment plants (WWTPs), but this has largely gone uninvestigated to date. Efforts to assess this issue are confounded by a lack of knowledge on the extent to which surface water flows are influenced by TWW. In this way, it is hard to identify areas in which surface water used for agricultural irrigation is impacted by TWW. Given more knowledge on the topic, the awareness of the de facto reuse of TWW may increase and regulations on the practice could be developed. Indeed, given proper management strategies de facto reuse could represent an important freshwater resource for agriculture into the future (Drewes et al., 2017).

In terms of scientific problems, a key issue is that assessing flow contributions of TWW is difficult in a highly dynamic hydrological system. Several sources contribute to the discharge of surface water bodies, all of which are variable over time and space. Complications in measuring the presence of TWW stem from the wide-ranging contribution of contaminants to surface water bodies from other land use practices. For any given contaminant, multiple emission pathways exist and hence it is difficult to accurately identify the "fingerprint" of TWW within the hydrological system. As an example, pathogens detected in surface water may be derived from TWW, but may also be present in similar concentrations due to the extensive use of manure in agriculture (Pachepsky et al., 2011).

1.4 Aims and research questions

This process identifies research gaps in that the extent of TWW impaction of surface water bodies in the Netherlands is mostly unknown, and the prevalence of de facto reuse totally unknown. Addressing these could aid water managers in better characterizing the source of surface water quality issues and more precisely reusing TWW as a functional source of freshwater. This informs the following aims:

- Determine the impaction of surface water bodies in the Netherlands with TWW
- Link emissions of TWW from WWTPs to agricultural irrigation
- Determine to what extent TWW is currently de facto reused in the Netherlands

The above aims translate to the following research questions:

- Does TWW make up a significant flow component for surface water bodies in the Netherlands?
- To what extent is de facto wastewater reuse for agricultural irrigation currently occurring in the Netherlands?

2 METHODOLOGY

2.1 General approach

This research aims to "follow the water"; tracing TWW through the hydrological system of the Netherlands, with an emphasis on de facto reuse in agricultural systems. This therefore follows its release at WWTPs, transport and distribution among surface water bodies and eventual extraction and application on agricultural fields (Figure 2). In this way, TWW propagates from a point source to a linear network to a planar distribution, becoming increasingly pervasive within the hydrological system. A national scale water quality model is used to simulate the partitioning of flow components within surface water bodies and the transport of TWW through the surface water network. Subsequently, a spatial analysis step is applied to relate the distribution of TWW in the surface water system to agricultural extractions for irrigation, hence identifying areas in which de facto reuse is occurring. A modelling study is selected as appropriate in answering the identified research questions as it is currently impossible to provide a national scale, continuous representation of TWW distribution within surface water bodies from direct measurements. Modelling the problem therefore represents the next best thing in that it allows for the simulation of the emission and transport of TWW.

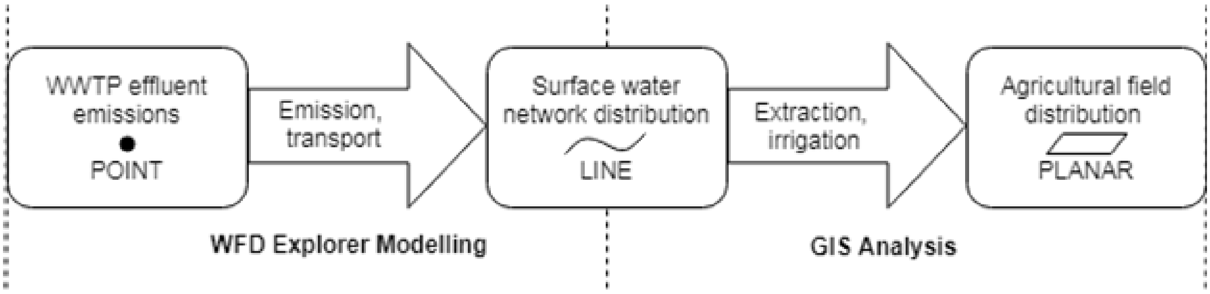


Figure 2 | Methodological setup: following the (waste)water from emission to application

2.2 WFD-Explorer

The Water Framework Directive (WFD)-Explorer is a national scale implementation of the surface water quality model D-WAQ, created to aid decision makers in meeting Water Framework Directive related water quality and ecology targets (Deltares, 2017). The model is based on a simple version of the advection-diffusion equation to simulate reactive transport processes in open water systems. Average fluxes of WFD identified contaminants through surface water bodies under steady state flow conditions are simulated at timesteps of one quarter (3 months) for the entire Netherlands. The model discretises the surface water network as surface water units (SWUs); these are nodes between which the model can simulate the dynamic transfer of water and substances. Surface water hydrodynamics are therefore resolved as transfers through links between SWUs (Figure 3). Contaminant sources are resolved as either point or diffuse inputs to the model domain, with WWTPs and transboundary rivers (TRs) both resolved as point sources of both discharge and contaminant fluxes.

In this implementation of the WFD-Explorer, around 27,000 nodes are incorporated, accounting for all major surface water bodies in the Netherlands (Deltares, 2017). Steady state simulations are conducted for two specific periods: the first quarter of 2007 and the second quarter of 2011. These periods are chosen as they represent relatively wet and dry conditions based on their respective precipitation deficits and surpluses (see KNMI, 2019), thus representing a wide range of discharge conditions in surface water bodies. 363 WWTPs are included as point source inputs to the model domain. The influence of TRs on discharge and water quality are also simulated as 65 point source inputs located on the German and Belgian borders.

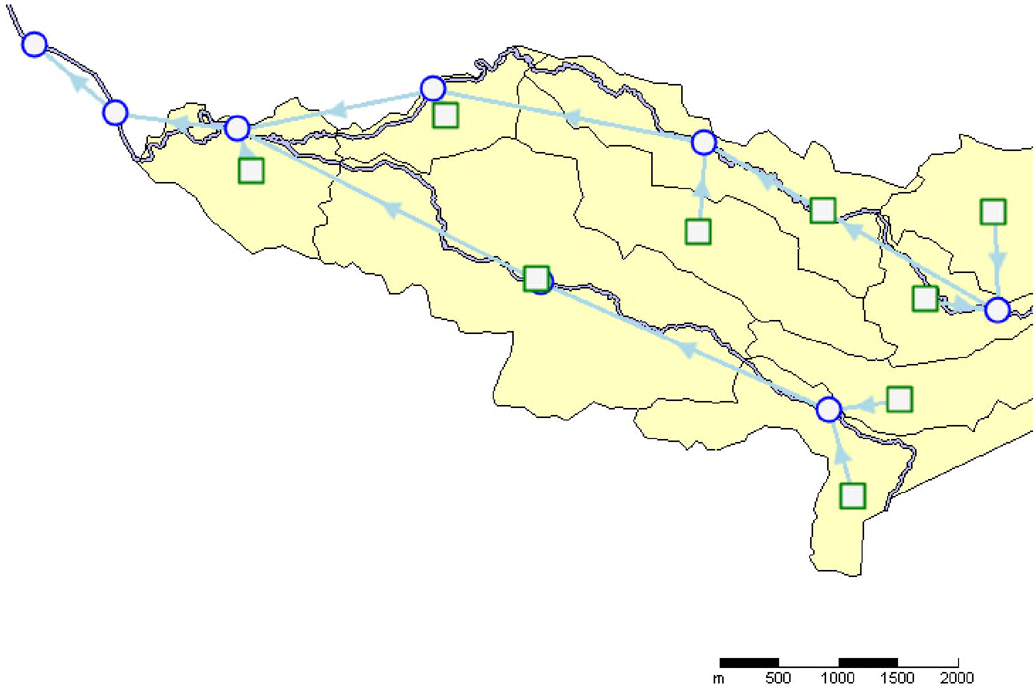


Figure 3 | Schematisation of the WFD explorer model, with SWUs as blue circles, point sources of discharge as green squares and links in blue. Taken from Deltares Wiki (Deltares, 2017).

2.2.1 Metamodel, transfer matrices

The WFD-Explorer is used to iteratively simulate – using the discretisations described above – how a constant flux of 1000 [g/s] from each individual point source (WWTPs, TRs) of a nominal conservative and non-conservative (decay rate k = 0.05 [d^{-1}]) tracer will eventually become distributed as fluxes through SWUs. When this procedure is repeated for each WWTP and TR considered, two transfer matrices (one for WWTPs, one for TRs) can be constructed, detailing how emissions from each individual point source will influence fluxes through every SWU given prescribed hydrodynamic conditions. Table 1 gives a simplified representation of one such transfer matrix, but for the implementation detailed in this study transfer matrices with dimensions [27435 SWUs x 360 WWTPs] and [27435 SWUs x 65 TRs] were considered, representing a full national schematisation. The sum of fluxes through each individual SWU gives an indication of which is most impacted by emissions from a given TWW source. This procedure is applied to transfer matrices for wet (first quarter of 2007) and dry (second quarter of 2011) conditions.

Table 1 | Simplified transfer matrix considering 3 WWTPs and 3 SWUs

SWU	Point sources			Σflux through SWU [g/s]
	Flux from WWTP1 [g/s]	Flux from WWTP2 [g/s]	Flux from WWTP3 [g/s]	
SWU1	100	200	500	800
SWU2	200	0	300	500
SWU3	700	800	200	1700
Σflux	1000	1000	1000	

2.2.2 Modelling WW inputs

Previous implementations of the WFD-Explorer have shown how these transfer matrices constructed using nominal tracer values can be used to model the transport of other contaminants through SWUs. These apply simple scaling calculations to the matrices, utilising the assumption that the same partitioning of fluxes between SWUs will apply for other substances given the same hydrological conditions (see Coppens et al., 2015; van Wezel et al., 2018). This study employs a similar procedure, but takes a novel approach via labelling all discharge from point sources (WWTPs, TRs) as fully conservative tracers. This step therefore assumes that the same hydrodynamic processes will dictate the partitioning of a nominal mass flux of conservative tracer between SWUs for as for a nominal unit of discharge (TWW or TR). This allows TWW and discharge from TRs to be traced through the surface water network, eventually yielding the distribution of TWW through every modelled SWU and identifying SWUs which are highly influenced by TWW.

To accomplish this, the distribution of tracer fluxes through SWUs are first redefined as fractions, then scaled with recorded discharge values for WWTPs and TRs (Table 2 shows this procedure for WWTPs). In the case of emissions from Dutch WWTPs, the transfer matrix is scaled by recorded TWW discharge data from the CBS-Micro database. For TRs, two methodologies are applied: the first scales the transfer matrix with observed discharge values for TRs over the period considered, hence returning proportion impaction for each SWU by TR discharge; the second uses equivalent derived TWW discharge values extrapolated from Carbamazepine (CBZ) concentrations (see Section 2.3.1). Following scaling, the sum of discharges from each SWU originating from each TWW can be calculated. This yields an approximation of the average discharge over the given time period for each SWU originating as either TWW or TR discharge. In this way, the flow components of each SWU in the model domain can be derived.

Table 2 | Simplified scaling procedure applied on Table 1

SWU	**WWTP1** WWQ = 2 [m^3s^{-1}]		**WWTP2** WWQ = 0.5 [m^3s^{-1}]		**ΣWWQs through SWU (Q^{WW})** [m^3s^{-1}]	**Total Q SWU (Q^{TOTAL})** [m^3s^{-1}]	**Percentage TWW impaction (I^{WW})** [%]
	Scaling Fraction [-]	**WWQ equivalent** [m^3s^{-1}]	**Scaling Fraction** [-]	**WWQ equivalent** [m^3s^{-1}]			
SWU1	0.1	0.2	0.1	0.05	0.25	0.5	50
SWU2	0.2	0.4	0	0	0.4	40	1
SWU3	0.7	1.4	0.4	0.2	1.6	1.6	100

From this methodology, two model schematisations can be derived, with the difference between the two lying in the treatment of TR inputs described above. The first, henceforth referred to as the three component model, uses the transfer matrix generated for WWTPs and TRs to work out the proportion impaction (*I*) of SWUs by TWW and discharge from TRs. Using these two calculated values, a third parameter can also be derived which is representative of internal non-anthropogenic flow (runoff) – more specifically the combined contributions of direct precipitation to surface water bodies, surface runoff, shallow subsurface runoff and groundwater seepage. This may be summarised as such, with subscripts *WW,nl*, *TR* and *RU* representing discharges from Dutch WWTPs, transboundary rivers and runoff respectively:

Three component model:

$$I_{WW,nl} = \frac{Q_{WW,nl}}{Q_{TOTAL}} \quad , \quad I_{TR} = \frac{Q_{TR}}{Q_{TOTAL}} \quad , \quad I_{RU} = 1 - I_{WW} - I_{TR}$$

The second model schematization, henceforth referred to as the combined model, treats TRs as point sources of TWW, therefore attempting to represent the inputs of extranational WWTPs on SWUs within the Netherlands. The combined discharges from TWW therefore give a total impaction value for each SWU; this is a useful metric as it is useful in relating total wastewater impaction to de facto reuse via agricultural extractions for irrigation. This scheme is summarised as such, with subscripts *WW,tr* and *WW,TOTAL* representative of extranational and total TWW discharges respectively:

Combined model:

$$I_{WW,TOTAL} = \frac{Q_{WW,nl} + Q_{WW,tr}}{Q_{TOTAL}}$$

2.2.3 Travel time

Using the transfer matrix described in Table 1 for a nominal conservative and non-conservative tracer, it is also possible to calculate travel times between each point source of and respective receiving SWU. The following equation, first implemented and further described by Coppens et al. (2015) may be used to derive travel time *T* between point source *i* and receiving SWU *j*, with *F* values representing fluxes conservative (*c*) and non-conservative (*nc*) tracer and *K* representing decay rate:

$$T_{i,j} = -\frac{\ln\left(\frac{F_{nci,j}}{F_{c\,i,j}}\right)}{K_{nc}}$$

This equation is applied to give a respective travel time between each point source and SWU, hence generating a distribution for each individual SWU. This distribution can subsequently be used to generate an average travel time per SWU; this is a useful metric as it is indicative of the amount of time TWW associated compounds have had to undergo degradation during reactive transport.

A large caveat here is an inability to predict total travel times for TWW which originates outside of the Netherlands due to the spatial constraints of the model; this means that results can only include average travel times between point sources of effluent as treated in the model framework (Dutch WWTPs, TRs at the point at which they enter the Netherlands). Although restrictive, this still allows for some insight into decay processes occurring during transport.

2.3 Deriving wastewater impaction using Carbamazepine concentrations

For the purposes of both informing model inputs and validating results (Sections 2.3.1, 2.3.2), observations of wastewater impaction values for surface water bodies are required. Directly measuring this metric is, however, not possible and therefore it must be extrapolated from measurable concentrations of TWW related contaminants. CBZ is chosen as for this purpose is widely acknowledged as an effective tracer of TWW contamination. This is due to the fact that it is highly conservative, originates primarily from domestic wastewater and has no natural background concentration in surface water systems (Lim et al., 2017). Observed CBZ concentrations and surface water discharges are used to extrapolate TWW discharge equivalent values using the following linear dilution model:

$$Q_{WW} = Q_{TOTAL,obs}\frac{C_{CBZ,obs}}{C_{CBZ,effluent}} \quad or \quad I_{WW} = \frac{C_{CBZ,obs}}{C_{CBZ,effluent}}$$

Whereby $Q_{TOTAL,obs}$, $C_{CBZ,obs}$, and $C_{CBZ,effluent}$ represent observed discharge values, the observed concentration of CBZ and concentration in pure TWW effluent respectively. $C_{CBZ,effluent}$ is set at the constant value of 0.62 [ug l^{-1}] CHECK UNITS in line with average concentrations found in a comprehensive review of effluents from Dutch WWTPs by RIVM (2014). This methodology obviously represents a large assumption but is chosen as CBZ has proven a robust tracer for TWW in the aquatic environment in numerous studies (Dickenson et al., 2011; Gasser et al., 2014). Furthermore, although generally scarce, CBZ measurements are available in a variety of locations for the time periods considered in this study.

2.3.1 Application: transboundary river inputs

To quantify total TWW impaction values ($I_{WW,TOTAL}$) for SWUs and give a more complete picture of impaction at the national scale, it is necessary to quantify the average TWW discharge originating from extranational WWTPs and transported in to the Netherlands by TRs. The linear model described in Section 2.3 therefore gives a methodology to accomplish this. This study uses data on discharges and contaminant concentrations from monitoring programs located close to the modelled node for each major TR considered to construct derived TWW discharge equivalent values for each transboundary input. Measurements utilised come from RIWM and Hydronet databases in the Netherlands; also making use of datasets from German water authorities who keep high resolution water quality records for measurement points located close to the Dutch border (NLWKN, ELWAS databases).

For these purposes, Carbamazepine (CBZ) concentration measurements for all significant TRs are considered. "Significant" rivers are hereby defined as having a discharge greater than 1 [m^3s^{-1}] during the wet period considered (first quarter of 2007). This procedure generated the results for each TR detailed in Table 3, which are subsequently used as model inputs for the combined model described in Section 2.2.2.

Table 3 | Calculated equivalent wastewater discharge values for transboundary inputs at the point at which each river crosses the border.

River	**2007 Q1**			**2011 Q2**		
	Average concentration CBZ ($C_{CBM,obs}$) [ug l^{-1}]	**Average Q ($Q_{TOTAL,obs}$)** [m^3 s^{-1}]	**Equivalent derived WWQ (Q_{WW})** [m^3 s^{-1}]	**Average concentration CBZ ($C_{CBM,obs}$)** [ug l^{-1}]	**Average Q ($Q_{TOTAL,obs}$)** [m^3 s^{-1}]	**Equivalent derived WWQ (Q_{WW})** [m^3 s^{-1}]
Rhine (Lobith)	0.17	3140	125.6	0.02	1180	250.75
Meuse (Eijsden)	0.07	637	22.93	0.018	78.8	6.9
Vecht	0.07	43.8	5.26	0.06	5.85	0.51
Dommel	0.275	3.28	0.45	0.069	1.44	0.49
Niers	0.31	13.7	4.66	0.17	5.9	2.32
Roer	0.22	29.6	5.33	0.09	10.7	2.94
Tongelreep	0.1	1.64	0.07	0.02	0.71	0.09
Dinkel	0.24	2.28	0.36	0.08	1.14	0.34
Berkel	0.32	5.38	0.43	0.04	2.23	0.89
Bocholter	0.23	6.77	1.35	0.1	2.24	0.64
Ijssel	0.39	3.06	0.49	0.08	0.75	0.37

2.3.2 Application: quasi-validation of model outputs

Assessing the relative amount of error generated in this framework is difficult due to the lack of data available to validate results, and the inability to measure the presence of TWW directly. Nonetheless, to partially constrain this uncertainty, a quasi-validation step is introduced in areas where appropriate data is available. To these ends, a dataset of 21 CBZ concentration measurements provided by HydroNET (2019) located in the south of the Netherlands and 10 measurements by RIWA (2019) located along the Rhine and Meuse, which span the "dry" model simulation period (2011 Q2) are utilised. The linear model from Section 2.3 is used to convert these measurements to represent TWW impaction. These values are then directly compared with the modelled value at the node closest to the measurement point, located using a "Nearest Neighbour" function implemented using the Python spatial analysis library Shapely.

2.4 Spatial analysis

To gain an insight into how surface water impaction by TWW is likely to be distributed terrestrially during the process of irrigation, a spatial analysis step is applied. This first aims to identify where surface water extractions are occurring for agriculture and attribute the sources of this water to each individual extraction. Subsequently, this is then related to the agricultural area it services. Thus, the distribution of TWW in the surface water network can be related to a planar agricultural surface, hence showing how TWW can be further propagated through the hydrological system. When combined with soil and geological data, this may also help identify areas where pristine groundwater may be at risk of contamination by TWW associated compounds.

This is achieved by using openly available datasets detailing the locations of irrigated areas using surface water from the Netherlands Hydrological Instrument (Deltares, 2014) and registered agricultural crop plot extents (Ministry for Economic Affairs and Climate, 2018). These are raster and polygon datasets respectively (Figure 4). First, surface water irrigation locations are attached to model outputs of TWW impaction using a nearest neighbour join function implemented from the Python library Shapely; this assigns the impaction value generated for the closest (minimum distance between any point along surface water network and centre of pixel) SWU polygon to each pixel of the irrigation raster. Next, the value for wastewater impaction attached to each pixel of the irrigation raster is assigned to underlying crop field extent polygons using an intersection function implemented from Geopandas. This process therefore attaches a TWW impaction value from irrigation water to the fields it services, allowing for the identification of areas of high de facto reuse.

This procedure relies on several assumptions: that agricultural abstractions from surface water bodies come from the closest modelled surface water body; and that these abstractions are applied to fields in the location in which they take place. It is also clear that the resolution of the raster image for irrigation used in this step is poor (250m) when compared to the accuracy of the field extent polygons. These issues may produce errors at the individual field scale but are considered justified to give a more general picture of de facto reuse trends at the national scale.

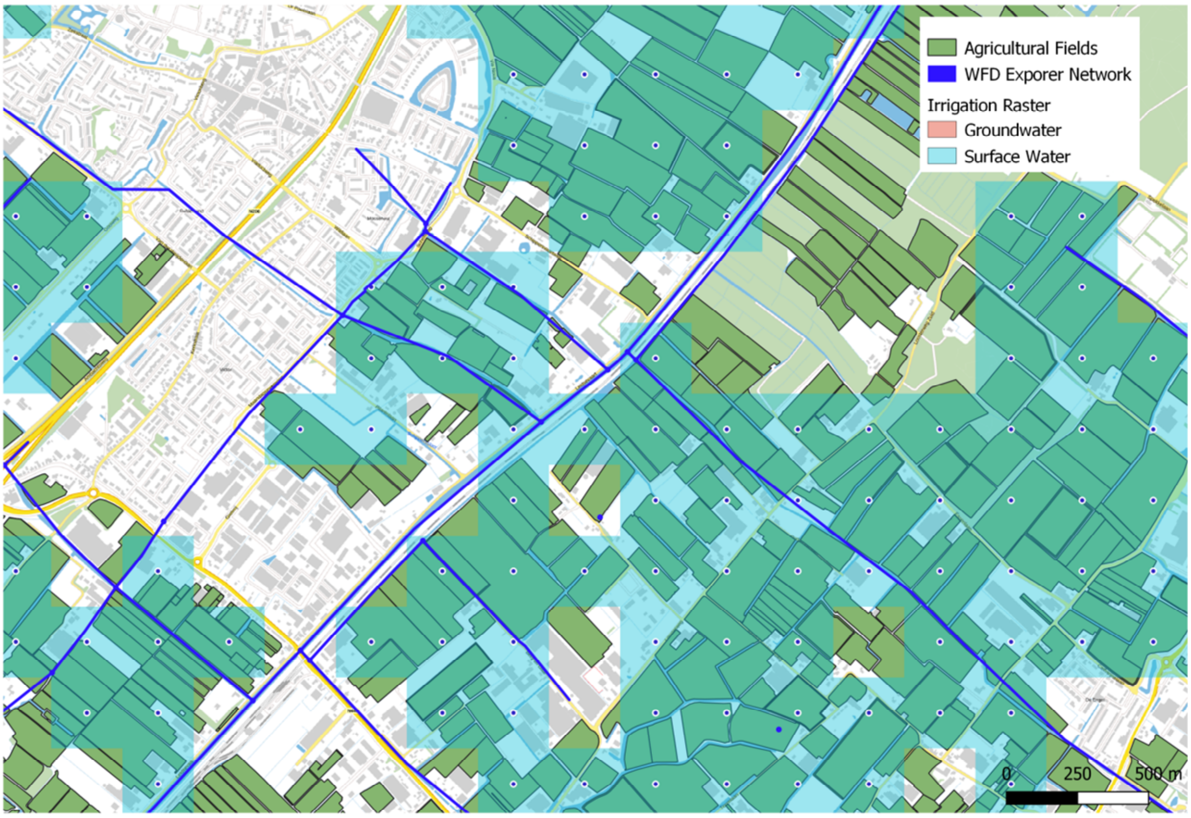


Figure 4 | Visual representation of datasets used in spatial analysis step.

3 RESULTS

The following section presents results obtained through the application of the methodological framework described above. As much as possible, these results are compared with previous studies and appropriate datasets to serve as a reliability check for model outputs.

3.1 National scale analyses

3.1.1 Three Component Model

The three component model shows a number of interesting large scale trends (Figure 5). The schematisation clearly succeeds in representing the transport and dilution of different water sources; the flow of water from international sources through large river systems is clearly visible in both periods and TWW discharge in small streams is diluted downstream of WWTPs by runoff. In general, it appears that smaller streams are impacted to a greater extent by TWW effluents; this is intuitive as discharges from WWTPs are likely to be of similar magnitudes to baseflows in these channels (evidenced further in Section 3.1.3).

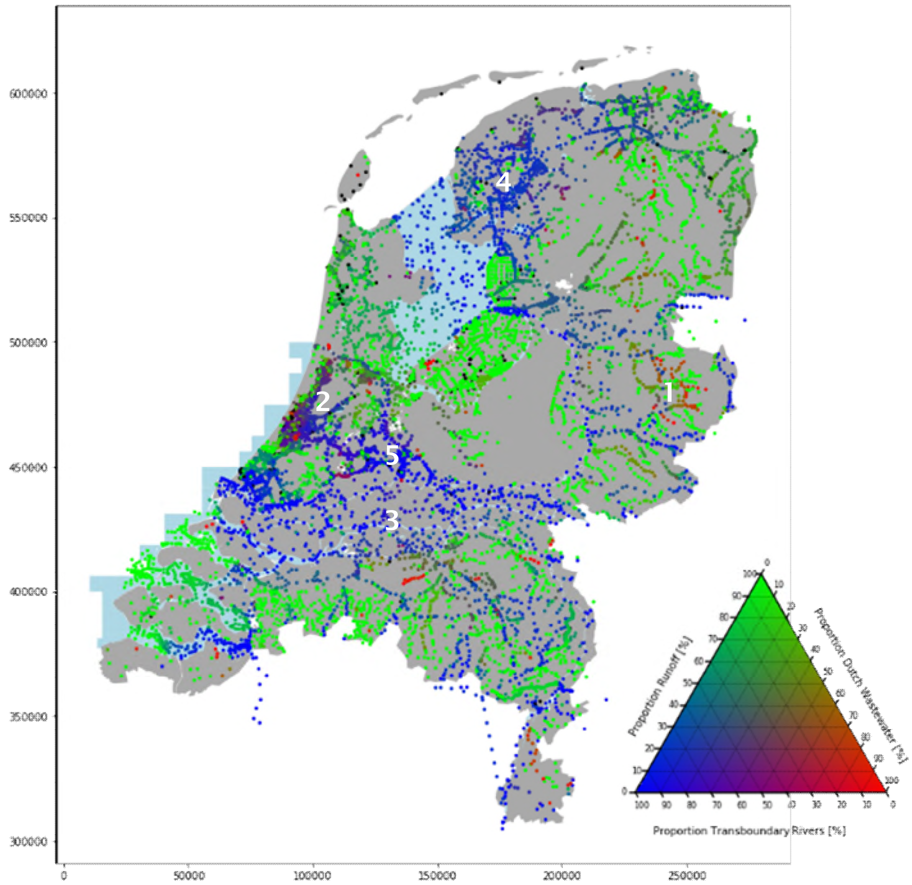
Differentiation between wet and dry conditions is marked. In dry conditions SWUs are impacted to a greater extent by Dutch WWTPs, especially in smaller surface water bodies in central eastern (Overijssel, #1) and western (Zuid Holland, #2) regions where impaction often approaches 100%. This occurs as dilution by runoff to these surface water bodies is low during Q2 2011. International rivers are clearly a vital component of surface water flow in dry periods, around the Meuse-Rhine corridor (#3), where the majority of flow comes from transboundary sources. Perhaps surprisingly, the model also indicates that in dry periods a large proportion of flow in Northern (Friesland, #4) and central (Utrecht, #5) SWUs originates from transboundary sources. Greater spreading of this water likely results from the model simulating the opening of water networks during low flow periods, a decision often enacted to allow access to water during drought conditions (Deltares, 2012). This reflects the high extent of control water management capabilities present in the Netherlands.

In wet conditions, discharge through the majority of SWUs stems from runoff, with discharge from international rivers less pervasive in the surface water network, although still dominant along the central Meuse-Rhine corridor. This reflects the greater influence of precipitation during this period and the reversal of flow networks to encourage draining of surface water bodies to the sea, also resolved in the model framework. The impact of TWW from Dutch WWTPs in wet conditions is visible in some areas, notably for the Zandelij in the Brabant area (#6) but is in general negligible for the majority of SWUs.

3.1.2 Combined Model

When international rivers are included as point sources of TWW effluent, a general representation of TWW impaction is obtained (Figure 6). These results show a compelling difference in terms of TWW impaction in wet and dry periods, with TWW making up a much higher percentage of flow in the majority of SWUs during the dry period. Encouragingly, the model again clearly produces dilution effects along the course of surface water bodies (especially visible in streams in eastern areas.

During dry conditions, results show that TWW is a pervasive component of surface water flow in the hydrological system of the Netherlands. Central eastern (Overijssel) and western (Noord-Holland) areas are again shown to be impacted, with discharge through many SWUs being composed of 80-100% TWW. This implementation shows that TWW inputs from the Rhine and Meuse contribute to the impaction of a large number of surface water bodies in the central Netherlands. Results also show clearly the influence of TWW originating from transboundary sources, with around 20% of the flow component of large surface water bodies in the Rhine-Meuse corridor made up of TWW. This effect is also evident in the central southern area (Noord-Brabant, #1) which is influenced by TWW transported by the Dommel. TWW transported by the Rhine and Ijssel also has a clear influence on impaction in Friesland and Utrecht areas.



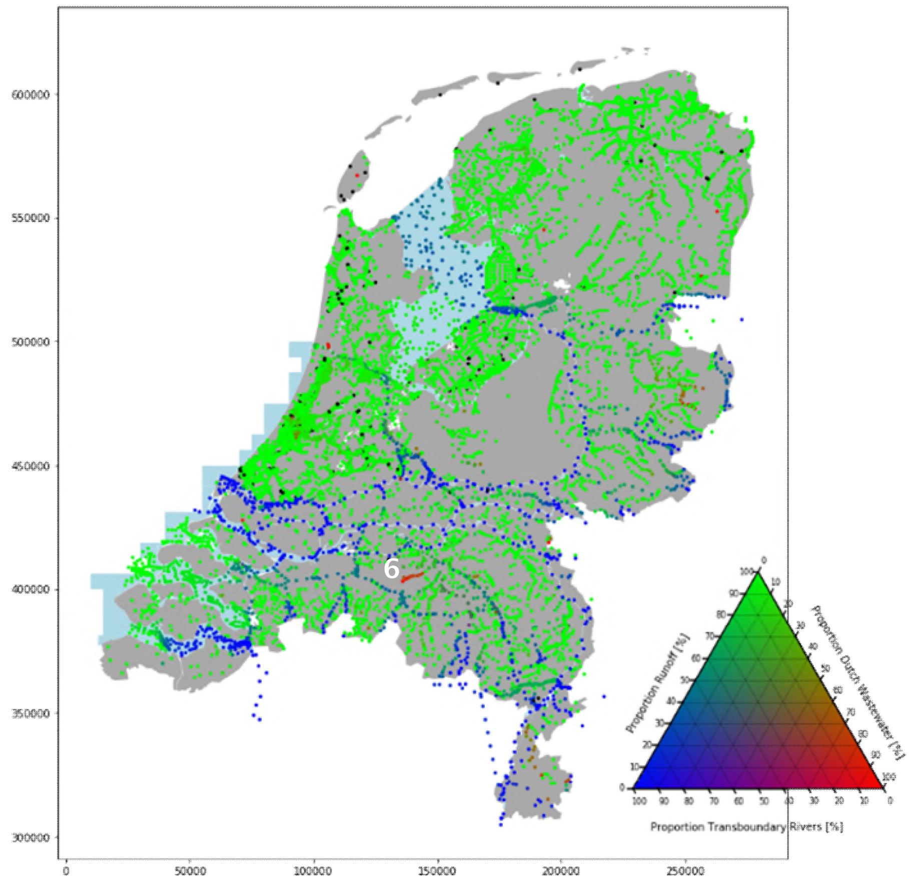


Figure 5 | Proportion discharge through SWUs originating from each component for 2011 Q2 (top) and 2007 Q1 (bottom), representing dry and wet conditions respectively.

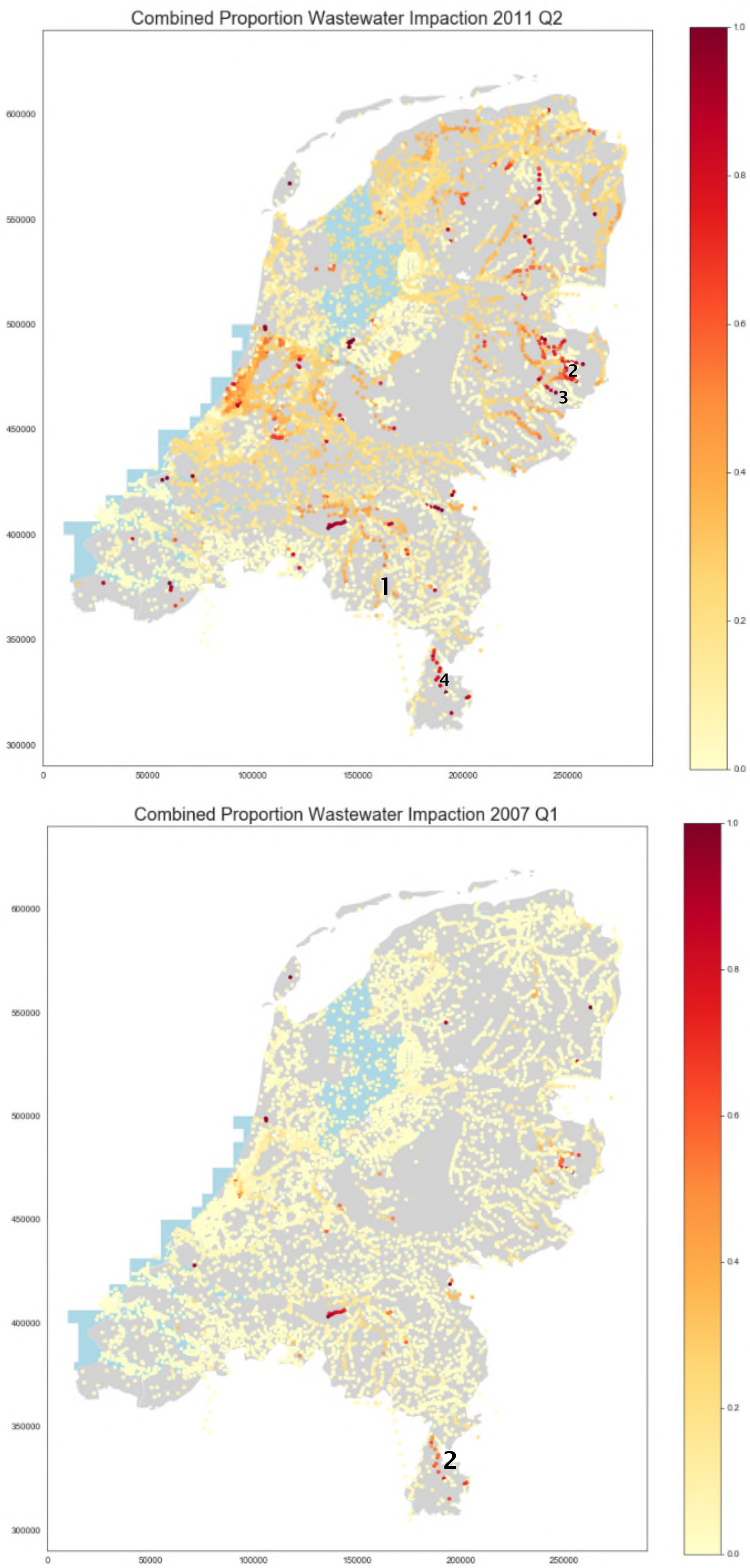


Figure 6 | Proportion discharge through SWUs originating from TWW effluent from both Dutch and international WWTPs

In wet conditions, conversely, results suggest that TWW is largely unimportant in maintaining flow for the majority of SWUs. Indeed, the only areas in which TWW makes up a significant flow component are in a few smaller watercourses in the Noord-Holland and Limburg (#2) areas, and the Zandelij. Some wastewater impaction is also still indicated for the Dommel.

Model validation indicates that predictive accuracy is in some ways limited – especially at higher predicted and observed wastewater impactions, where two data points in particular show large relative under- and over-predictions (Figure 7). A general trend toward overprediction is also shown in relation to the Hydronet data, which consists mostly of measurements taken in smaller streams. Overall, however, a fair level of agreement is shown, indicating that the model is useful in predicting the occurrence of TWW in surface water bodies.

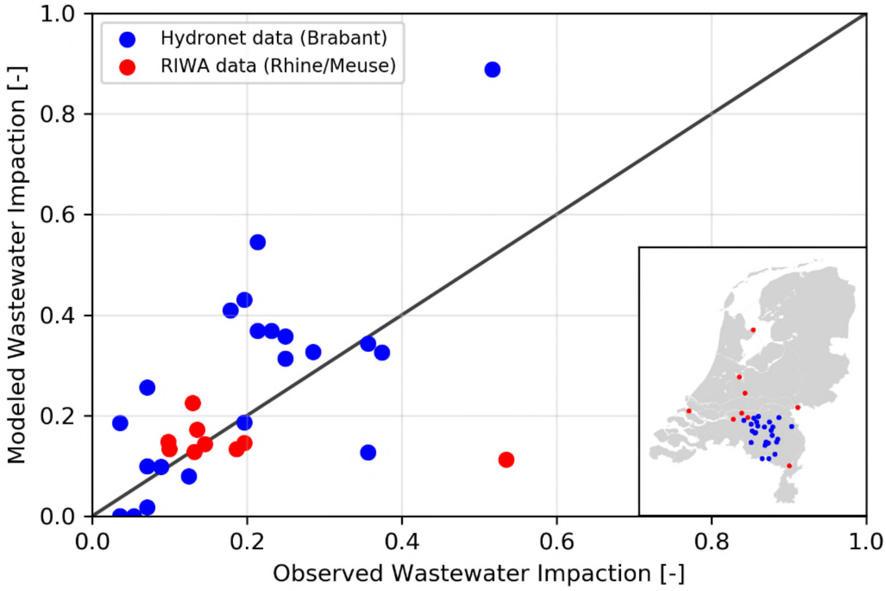


Figure 7 | Relationship between modelled and observed results from Hydronet and RIWA data with the locations of measurement points for CBZ inset.

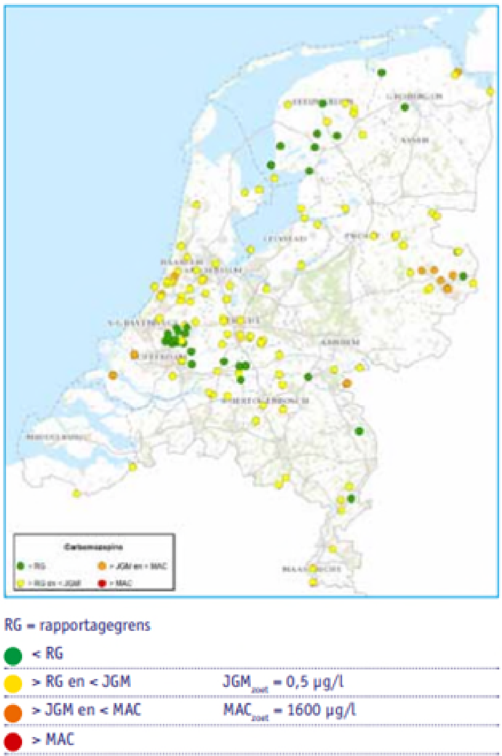


Figure 8 | Results of STOWA (2015) study on CBZ concentrations in surface water bodies.

Furthermore, when compared with the distribution of maximum CBZ concentrations in surface water bodies explored in a report by STOWA (2015), identified areas of TWW impaction show good agreement with model outputs for the dry period considered (Figure 8). This finds CBZ concentrations of above 0.5 [ug l^{-1}] in the same areas this study identifies as areas of high TWW impaction, notably in 5 samples taken from small streams in the east (Overijssel) and 3 samples taken from polder systems in the west (Zuid Holland area).

3.1.3 Travel times

Average travel times for TWW effluent to reach SWUs range from 0-200 days in both wet and dry conditions (Figure 9), with a large amount of variation within this range. In general, travel times are higher at the northern and western boundaries of the country, and lowest along the Rhine-Meuse corridor and central eastern areas. A clear distinction is again evident between wet and dry conditions, with travel times being higher in all areas during dry conditions. This is reflective of the same drainage patterns as earlier described, whereby water is retained terrestrially during dry periods – leading to longer residence times – and flushed to the sea during wet periods to alleviate flood risk. This also reveals an interesting dynamic; in wet periods travel times between sources of effluent and receiving SWUs is smaller, resulting in less time for decay to occur and meaning contaminants will be prevalent in higher amounts and to greater spatial extents. In this way, the spread of contaminants contained in TWW effluent is likely greater during wet periods.

Results in terms of TWW impaction become more informative in terms of predicting water quality when combined with estimates of travel time between sources of effluent and the receiving SWU. This allows for some insight into the decay processes which are likely to occur during transport. Highly conservative contaminants such as chloride, some pharmaceuticals and artificial sweeteners are unlikely to decay on time scales of 0-200 days (Dickenson et al., 2011) but for pathogens, travel times in the range of 20-100 days can lead to >90% reductions in concentrations under open flow systems; (see Brooks et al., 2015). In general, it should be noted that for dry conditions in the areas of significant impaction outlined above (Noord-Holland, Overijssel), travel times are relatively low (<50 days) and thus the model suggests that contaminants related to TWW are unlikely to have decayed by the time they reach these SWUs.

3.2 Discharge – Impaction relationships

A clear relationship between discharge and percentage impaction exists in modeled results for both dry and wet periods (Figure 10). The highest levels of impaction are found in small SWUs (< 1 [m^3 s^{-1}]), with the occurrence of high impaction (80-100%) clearly decreasing with increasing discharge. For large rivers, (> 20 [m^3 s^{-1}]), impaction is on average smaller and does not exceed 40%. Another interesting feature also found in this figure is the clear presence of the large number of SWUs impacted by TWW coming from the Rhine (line of SWUs at around 22% impaction).

This step reflects model outputs which identify several small streams as highly impacted in dry periods. These are in multiple areas, but most commonly in eastern regions (Overijssel). The following small streams are identified as being made up of 90-100% TWW discharge in dry periods:

- Bornsebeek, Overijssel (#2 in Figure 6)
- Bolscherbeek, Overijssel (#3 in Figure 6)
- Geleenbeek, Limburg (#4 in Figure 6)
- Zandleij, Noord Brabant

These results can be qualitatively assessed through several methods. For example, the description of the Zandleij by Waterboard de Dommel (2015) notes that in dry periods it is almost entirely constituted by TWW. Pictures of the Bolscherbeek from the drought period in 2018 also show that flow in this channel is entirely made up of TWW (Figure 1). Moreover, STOWA (2015) single out the Bornsche Beek as having CBZ concentrations in the same range as pure TWW effluent. This indicates a strength of the model in correctly predicting highly impacted small streams.

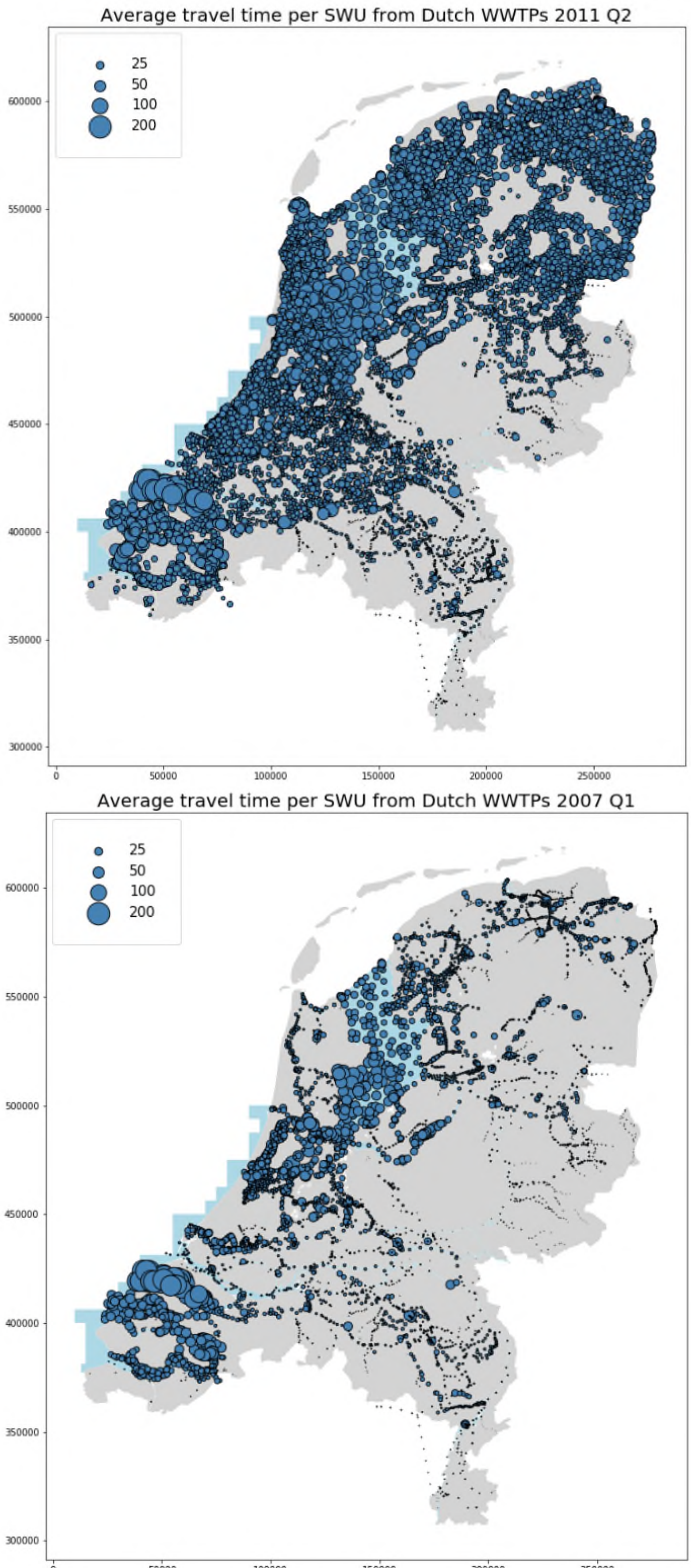


Figure 9 | Average travel times [days] between Dutch wastewater treatment plants and receiving surface water bodies.

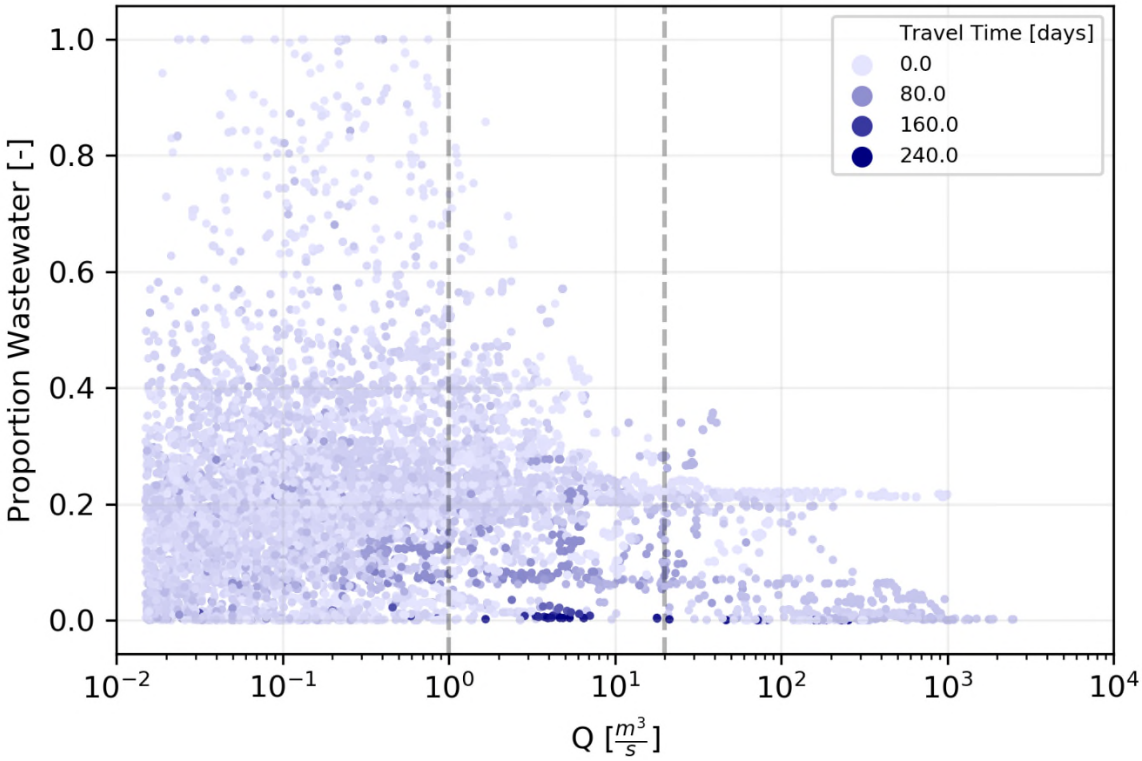


Figure 10 | Relationship between discharge and proportion wastewater impaction for 2011 Q2. Travel time is also included.

3.3 Distribution of de facto reuse

When the spatial analysis step is applied, a clear picture is generated of de facto wastewater reuse. Outputs detail agricultural fields that are irrigated using surface water, and the proportion impaction by TWW for the associated water body from which extractions for irrigation take place (Figure 11). Results are only shown for the dry period considered (2011 Q2) as in wet conditions irrigation does not take place.

These results indicate that de facto reuse is indeed occurring within the Netherlands, but with a large amount of spatial variation in the extent to which this occurs. Along the Rhine and Meuse, especially in central areas, it is clear that de facto reuse is occurring, with irrigation extractions generally containing 0-40% TWW. In central western areas of the Netherlands (Zuid Holland, #1), a large extent of de facto reuse is present, with extractions coming from surface water bodies 40-80% impacted with TWW. Another key area which was not obvious during earlier modelling stages is a patch of high de facto reuse (40-60% TWW) in the agricultural area of Noord Holland (#2) in which one WWTP seems to produce a large effect in terms of impaction. Smaller areas in which water used for irrigation seems is highly impacted (80-100%) by TWW do exist but are scattered and only seem to occur for individual fields, these results should be approached with caution as it is unlikely this framework will produce accurate results to the field scale.

It is difficult to compare these results with others as they represent the first of their kind in terms of linking TWW emissions to agricultural irrigation. It is, however, worth noting that a recent study on contamination of groundwater by pharmaceuticals (KWR, 2017) detected CBZ in groundwater samples in the same area of concern hereby identified in the western Netherlands (Zuid Holland). This study may therefore indicate some link between de facto wastewater reuse for agriculture and contamination of the subsurface by pharmaceuticals.

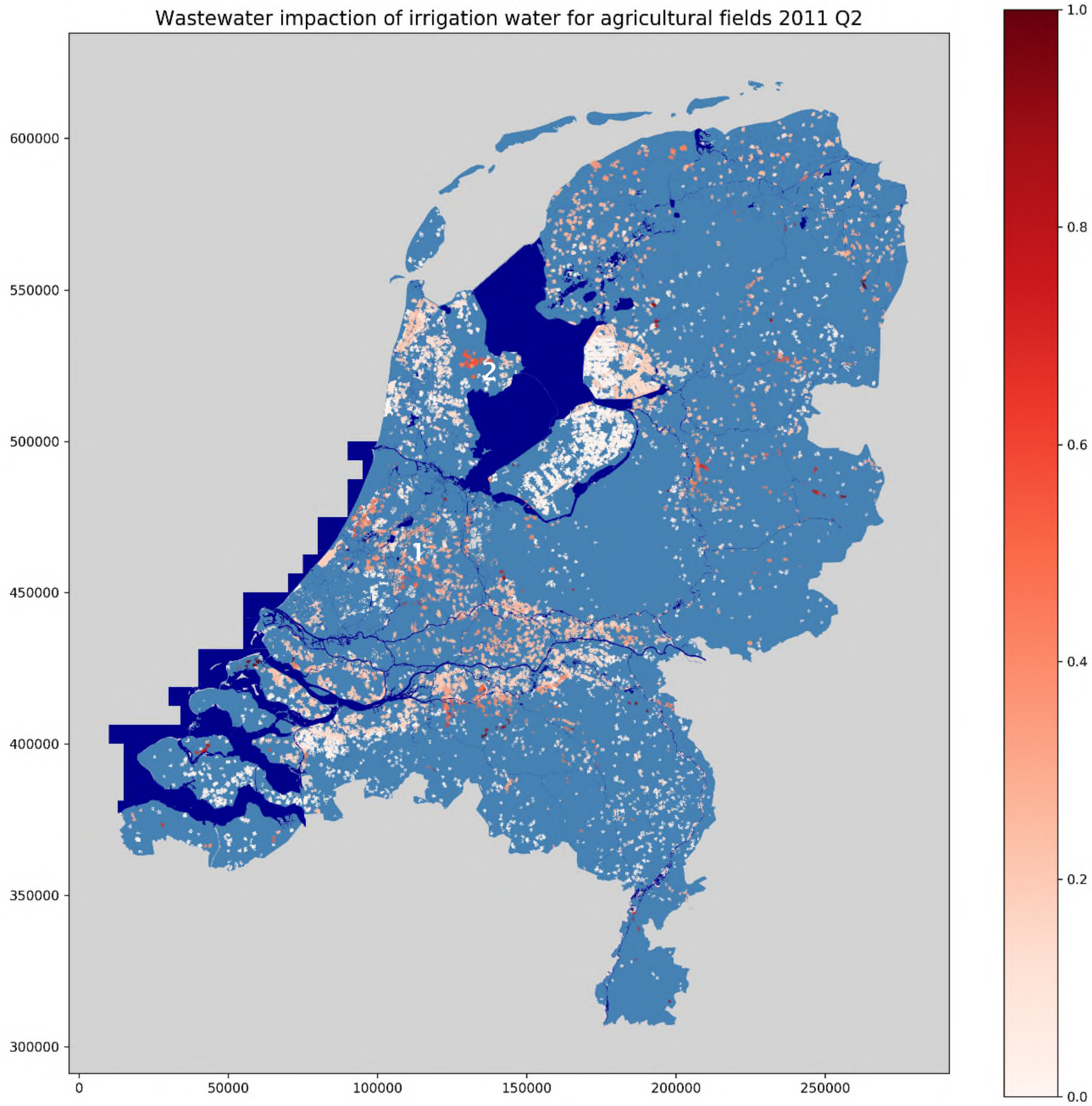


Figure 11 | TWW impaction of surface water extracted and used to irrigate the agricultural fields for the dry period considered (2011 Q2).

4 DISCUSSION

4.1 Model successes, limitations, uncertainties

The framework used in this paper has undoubtedly contributed to filling research gaps identified in Section 1.2. It has helped provide an insight into the distribution of TWW within surface water bodies in the Netherlands, and the flow components (Dutch WWTPs, TRs) which influence this distribution. This is a valuable outcome as it suggests a mechanism by which wastewater propagates through the surface water network. Moreover, the assessment was performed at a national scale, differentiating it from other studies which have been employed at the catchment (Drewes et al., 2009) or reach (STOWA, 2017a) scales. The study also provides a first step towards linking WWTP emissions to de facto reuse within the Netherlands. Indeed, this is one of very few studies internationally to explore this problem (see also Rice & Westerhoff, 2014; Thebo et al., 2017) and the only one – to the authors knowledge – that follows initial TWW emissions to the planar surface of agricultural fields. This is an important step in that it outlines a pathway by which TWW associated contaminants may spread through the surface water network and even to the subsurface.

Although results are encouraging, a number of limitations may be identified. One of these is the reliance on the assumption that CBZ can be related directly to TWW discharge. This introduces a large amount of uncertainty due to conflicting literature on pharmaceutical concentrations in TWW: different average values and ranges of CBZ concentrations in pure TWW effluent are suggested by RIVM (2016) and STOWA (2015) and Brunsch et al. (2018) show that pharmaceutical concentrations in effluent vary in dry and wet conditions. Relating water quality to TWW discharge may have been better achieved by using a multiple tracer approach (Gasser et al., 2014; Guillet et al., 2019b) or a more complex model relating CBZ to TWW discharges which included the likely effects of decay after discharge from WWTPs. These steps would help to better constrain uncertainty and improved the reliability of model validation. Another limitation in the study is the fact that modelling efforts are at a national scale and do not cover larger basins (the Rhine and Meuse particularly). This necessitates uncertain estimations for the impaction of TRs as they enter the country. To overcome this, an international basin scale study would be useful; this would also provide more accurate average travel times as water could be consistently traced from WWTP to SWU. The fact that the model treats TWW as a tracer of itself is limiting in that it continues to label it as such even when in reality some of the associated contaminants may have been removed or degraded during transport (Naidoo & Olaniran, 2014; Pachepsky et al., 2011). This is an issue as at this point there may be fewer associated risks related to irrigation, but this factor is hard to include. Future studies would therefore attempt to better integrate parameters of TWW impaction and travel time, hence more clearly showing areas in which de facto reuse may be paired with high contaminant concentrations originating from TWW. Finally, a limitation is clear in the spatial analysis step in that it does not account for different irrigation practices. This is important as the dominant form in the Netherlands is subirrigation using polder networks to supply water directly to the root zone (see Frank & Belcher, 1994; Hagan et al., 1967) whereas the model implicitly assumes that water is extracted and applied using surface irrigation techniques. A better representation of the reuse of TWW in agriculture would therefore identify inlets to polder networks from surface water bodies and attach this value to the entire area serviced by this network. Data on polder networks, however, is only partially available through regional water boards and therefore at present the steps taken in this study represent a best estimate on de facto reuse using openly available national scale datasets.

Uncertainties and errors in results are undoubtedly present. These are generated from several steps which make large assumptions; namely the use of TWW as a tracer of itself and the linear model relating CBZ to TWW discharge. Further uncertainty is also likely generated in modelling due to the coarse resolution of space and time and necessary parameterisations. Validation of model results is far from ideal in that it is only possible for the combined model and data used is only partial in terms of both space (low coverage) and time (only dry period). It does, however, indicate that the framework is indeed accurate in predicting areas of high TWW impaction. Comparison with a number of existing datasets and studies also indicates a good level of agreement, with maximum impaction values in the same ranges (see Drewes et al., 2017; Rice & Westerhoff, 2014; Thebo et al., 2017). Moreover, all assumptions included in this framework have been carefully considered and grounded as much as possible in best practice or examples given in leading literature on the topic. This lends credibility to model results and suggests they are useful in identifying areas of TWW impaction and reuse which are likely of interest to a variety of stakeholders within the Netherlands. In future, a larger dataset of CBZ measurements would be useful to further assess model performance in other areas of the Netherlands. The expansion of current monitoring programmes and increased integration of water quality databases as achieved with the Informatiehuis Water (2018) will undoubtedly improve future research on this topic.

4.2 Water quality and de facto reuse

It is likely that in areas where SWUs are indicated as highly impacted, water quality concerns are applicable. Contaminants found in TWW include pathogenic microorganisms, nutrients (nitrates and phosphates), heavy metals and endocrine disruptors (Akpor et al., 2014). The presence of these in SWUs may lead to problems meeting WFD chemical water quality targets, negative impacts on ecology (Vajda et al., 2008), and reduced societal functions (Coppens et al., 2015). SWUs shown to be impacted by water from the Rhine in the central Netherlands may face other water quality concerns due to industrial processes in the basin (Sjerps et al., 2017). Furthermore, if the surface water body is infiltrating, transfer of contaminants to groundwater may occur. This is highly dependent on subsurface characteristics and hydrological conditions (Gallegos et al., 1999; Kass et al., 2005) but is indicated by studies which find a range of TWW related pharmaceuticals and personal care products in groundwater samples (Sjerps et al., 2017) Increasing the rigorousness of wastewater treatment to include quaternary treatment or a disinfection step would be useful in alleviating some of these water quality concerns (Coppens et al., 2015) but clearly costly to implement for all WWTPs. A key area which would increase the utility of this research in future would therefore link emissions from each (Dutch) WWTP to their total influence in terms of both TWW impaction and subsequent de facto reuse, allowing for the prioritisation of further treatment. The example of other studies which have successfully implemented this in relation to pharmaceutical emissions (see Coppens et al.; STOWA, 2017) would be useful for this purpose.

Perhaps the key result of the study is evidencing the widespread de facto reuse of TWW in Dutch agriculture. The next logical step in this discussion is whether this is necessarily problematic. It may be that the reuse of water in areas along the Rhine-Meuse corridor and in Noord Holland is in breach of guidance by the JRC (Alcalde-Sanz & Gawlik, 2017) on water quality requirements for irrigation but without closer monitoring this is not possible to discern. This undoubtedly requires future research to assess the quality of surface water used for irrigation. Indeed, a large scale review of irrigation water quality akin to studies by STOWA (2015) and KWR (2017) would be highly valuable to better inform farmers and water managers on de facto reuse. This study could certainly be of use in identifying key areas for monitoring programs to be located; namely areas exhibiting high levels of impaction paired with de facto reuse. Whatever the outcomes of such further investigations, it is imperative that this issue is discussed by Dutch water managers, with guidelines and regulations created regarding de facto reuse. These may include the designation of water bodies as extraction free zones and the establishment of minimum distances from WWTPs for extractions to occur, allowing for the potentially negative consequences of this practice to be minimised and the advantageous use of a valuable freshwater resource to continue. An important dimension of this discussion is the agreement of what represents an acceptable risk – in many areas globally (notably including Spain, Italy and Greece) TWW is widely reused in agriculture (see Drewes et al., 2017) and therefore it may be deemed that current levels of reuse within the Netherlands are acceptable within this context. Moreover, risk is dependent on irrigation practices and crop types, with drip and sub- irrigation systems reducing the chances of the contamination of produce (Navarro et al., 2015).

A further key outcome of this study is that it negates discussions over whether TWW should be reused in the Netherlands (see for example Rietveld et al., 2011) by indicating that this practice is already widespread. This is likely to influence the perceptions of water professionals and societal actors, perhaps even helping to reduce the stigma associated with reuse (Ricard & Rico, 2019). These results call into question the basis for prohibiting direct water reuse and will hopefully help to refocus the discussion on the question of how the country can more accurately reuse water as part of a well-managed and sustainable freshwater management plan. The current freshwater plan of the Netherlands (Deltares, 2012) fails to assess the large scale de facto reuse which this study indicates is already occurring and therefore should incorporate this into future reworkings. Useful future work here may also include the compilation of results from this and future studies into a user-friendly tool for visualising TWW impaction and reuse; this would be an important step in raising awareness to the issue and making results more useful to agricultural stakeholders and water managers.

4.3 Further considerations for water managers

A key finding of this study is the effects of surface water discharge on impaction and reuse dynamics, which has important implications when viewed in the context of climatic change. Predictions suggest that hydrological extremes are likely to lead to increased occurrence of drought conditions within the Netherlands (Mateo-Sagasta et al., 2015), suggesting likely impacts on both water quality and availability. This study is pertinent to this discussion in that it suggests during drought conditions TWW is a pervasive component of surface water flow; this supports a body of work that shows surface water quality issues to be more prevalent in dry periods (see for example Mosley, 2014; van Vliet & Zwolsman, 2008). Results also suggest that de facto reuse is more prevalent during drought owing to increased relative impaction paired with a higher demand for irrigation. In this way, under drought conditions TWW is likely to have increased negative impacts in terms of surface and subsurface water contamination, but conversely will represent an increasingly important freshwater resource with continuous discharge when other sources are restricted (Pedrero et al., 2010). This lends further weight to arguments for increasing treatment at WWTPs into the future to increase access to sustainable water resources in surface water channels.

A further key consideration is the identification in this study of “wastewater dominated” streams (Luthy et al., 2015); characterised as such when 80-100% of flow is constituted by TWW in dry conditions. This study has identified several such streams throughout the Netherlands, especially in Eastern areas (Overijssel). It is intuitive that these channels are highly impacted with very short average travel times, suggesting a range of problems in terms of water quality. This study fortunately indicates that extractions from these streams for agricultural irrigation are limited but other considerations are still applicable; literature on this topic identifies wastewater dominated streams as ecologically unique due to the fact that variation in flow is much lower annually (Brooks et al., 2006). Indeed, in drought periods, the streams may be important in supporting key species in channels which would otherwise be dry (Luthy et al.). These areas therefore should be carefully monitored, with the use of water from these streams restricted to both preserve ecological functioning and prevent the spread of TWW related contaminants. Other impacts from these streams should also be considered, including if they may contribute to the transfer of contaminants to pristine groundwater (Kass et al., 2005).

5 CONCLUSION

This paper has successfully applied a model framework to simulate the spread of wastewater through the surface water network of the Netherlands, generating data on impaction characteristics for large surface water bodies. It has also succeeded in relating – for the first time – the distribution of wastewater and other flow components within the surface water network to agricultural extractions, and subsequently created a picture of de facto wastewater reuse at a national scale. This represents a novel insight for Dutch water managers.

Key results include:

- The presence of wastewater is indicated in many SWUs, with areas of highest wastewater impaction identified in Zuid Holland and Overijssel
- Levels of impaction are higher in dry conditions, suggesting reuse will increase with more common incidences of drought predicted in a changing climate
- High percentages of wastewater 90-100% indicated in small streams close to WWTPs, these may be characterised as “wastewater dominated”
- Indirect wastewater reuse is shown to be widespread, with notable areas of reuse occurring in the central Netherlands (along the Rhine) and in Zuid Holland

Several key conclusions may be drawn from this study, relating directly back to the original research questions:

- Wastewater is a pervasive component of the surface hydrology of the Netherlands during dry/low flow periods, with 100% of flow originating from wastewater discharge in some surface water bodies
- Wastewater is being widely reused through de facto methods; acknowledging this will allow for this resource to be better managed, with extractions and treatment taking place at better informed locations

These findings should be useful for water managers to identify potential water quality concerns in different regions nationally, but also for land managers to make better decisions on where surface water extractions for irrigation should take place. More specifically, this study supports the creation and enforcement of guidelines for indirect wastewater reuse within the Netherlands and the establishment of monitoring programs for irrigation water quality. It should, however, be stressed that indirect reuse is a practice which gives access to freshwater resources under the pressures of a changing climate and should be incorporated – perhaps in conjunction with increased treatment processes at WWTPs – as a component of a sustainable national freshwater management plan.

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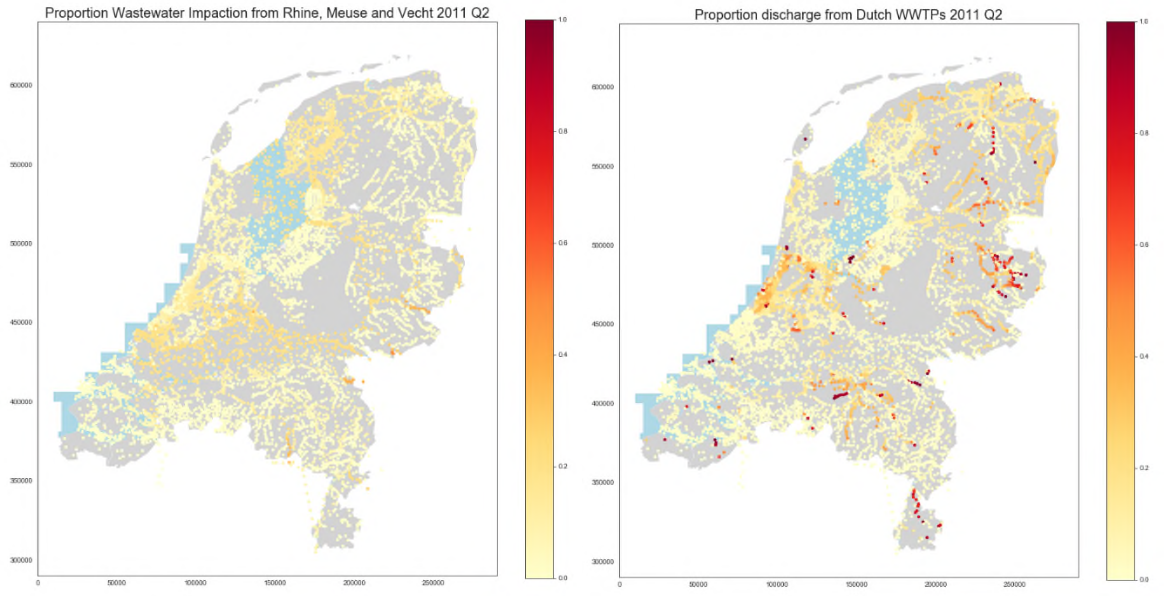
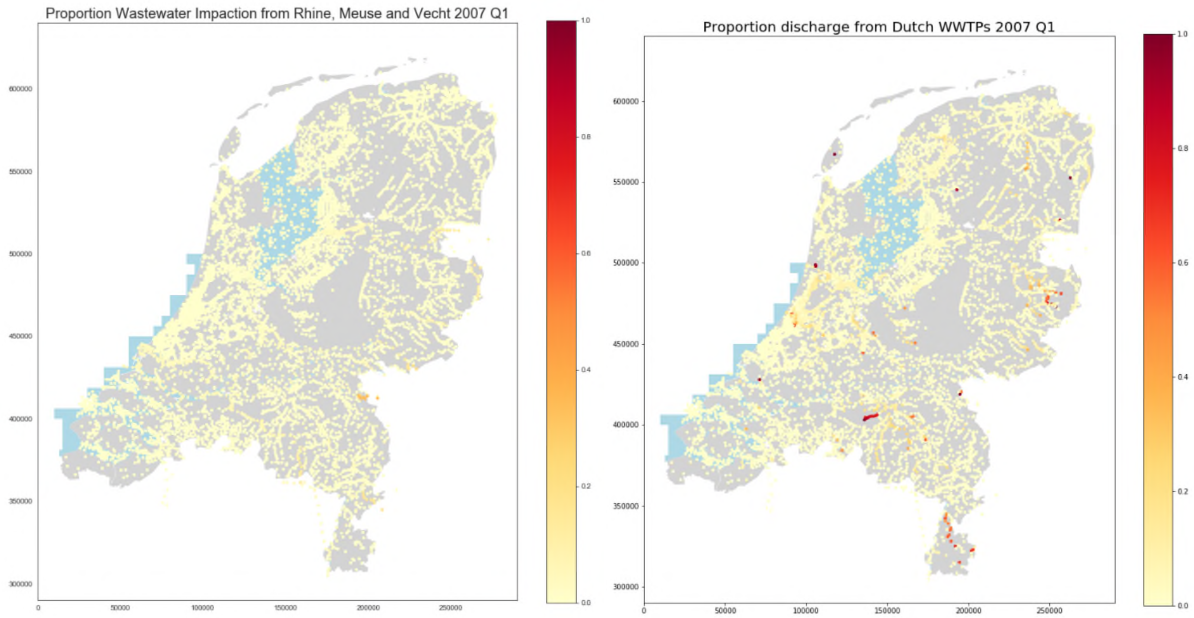
APPENDICES

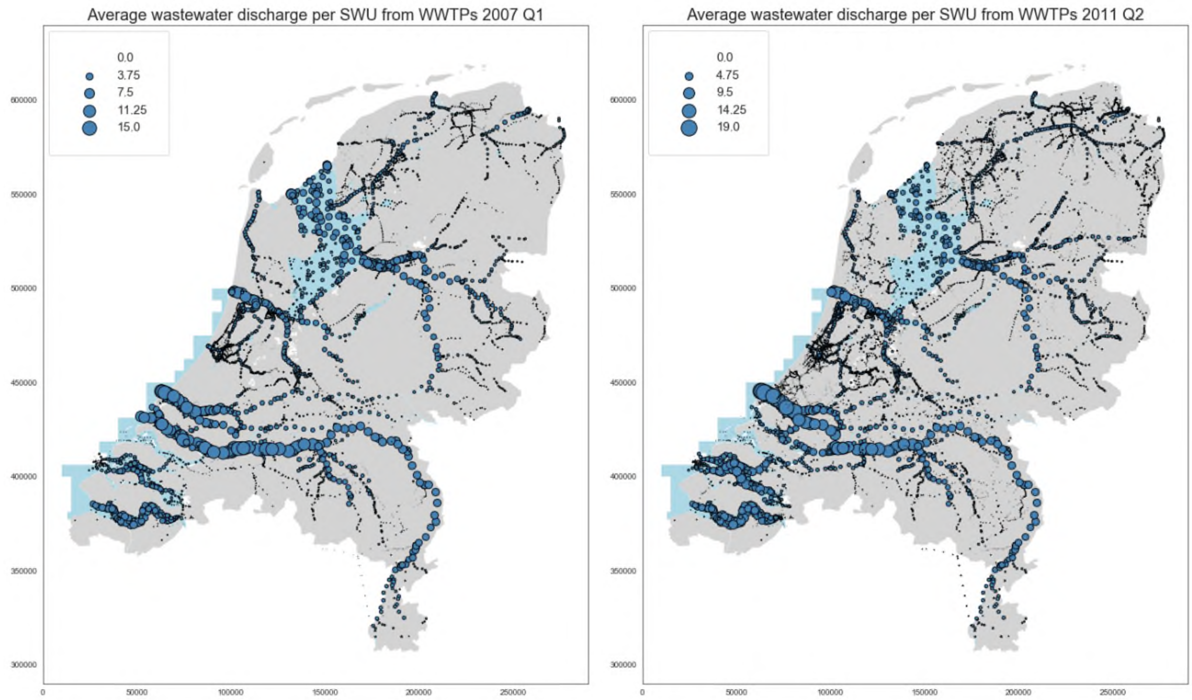
Appendix A: Unused figures



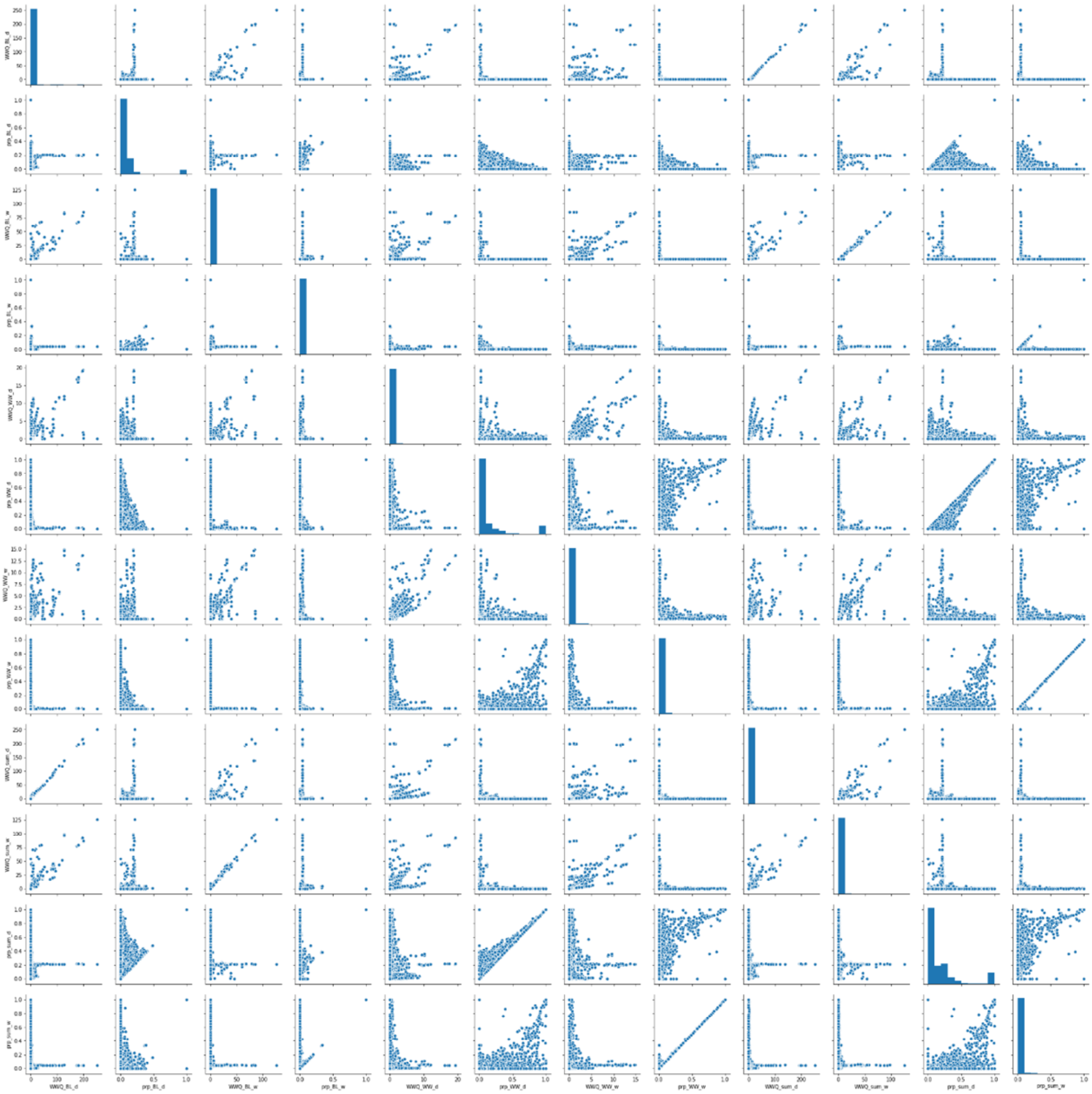
Individual proportion impaction figures for TRs and Dutch WWTPs for the modelled dry period



Individual proportion impaction figures for TRs and Dutch WWTPs for the modelled wet period



Total wastewater discharge per surface water body, indicating the accumulation of wastewater within larger SWUs



Relationships between model outputs (discharge, proportion impaction etc.) and distribution of data for modelled dry period (2011 Q2)

Appendix B: Datasets used in transboundary river scaling

River name	Node_ID	Flow_2007Q1	Flow_2011Q2	CBZ_2007Q1	CBZ_2011Q2	data_source
Rhine	BLLSM1	3,14E+03	1,18E+03	0.17	0.02	RIVM
Meuse	BLLSM2	6,37E+02	7,88E+01	0.07	0.018	RIVM
Overijsselse Vecht	BLLSM3	4,38E+01	5,85E+00	0.07	0.06	NLWKN
Niers	BLLSM4	1,37E+01	5,98E+00	0.31	0.17	ELWAS
Roer	BLLSM6	2,96E+01	1,07E+01	0.22	0.09	ELWAS
Dommel	BLLSM7	3,28E+00	1,44E+00	0.275	0.069	HYDRONET
Tongelreep	BLLSM8	1,64E+00	7,10E-01	0.1	0.02	HYDRONET
Dinkel Gronau	BLTN10	2,28E+00	1,14E+00	0.24	0.08	NLWKN
Dinkel Rammelbeek	BLTN7	1,15E+00	1,15E+00	0.24	0.08	NLWKN
Berkel	BLTN14	5,38E+00	2,23E+00	0.32	0.04	ELWAS
Bocholter	BLTN1	6,77E+00	2,24E+00	0.23	0.1	ELWAS
Issel	BLTN2	3,06E+00	7,50E-01	0.39	0.08	ELWAS