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Research Article

A Comparative Evaluation of Automated Recession Extraction Procedures for Karst Spring Hydrographs

Otomatik Çekilme Eğrisi Seçim Prosedürlerinin Karstik Kaynak Hidrograflarında Karşılaştırılmalı Değerlendirilmesi

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Abstract

Recession Curve Analysis is a common method to characterize karstic aquifers and their discharge dynamics. Although this technique provides crucial information on quantifying system hydrodynamic properties, the manually selected recession curves analysis is neither a practical technique to cover all candidate recession curves, nor it allows extracting the entire hydrological diversity of the recession behavior. This study aimed to comparatively evaluate the applicability of automated recession selection procedures to the late-time recession analysis of karst spring hydrograph. For the comparative evaluation of the three automated recession extraction methods (Vogel Method, Brutsaert Method, and Aksoy and Wittenberg Method), we quantified the late-time recession parameters of spring hydrographs by combining three extraction methods with four recession analysis methods (Maillet, 1905; Boussinesq, 1904; Coutagne, 1948; and Wittenberg, 1999). By applying our experimental design into the five karst springs located in Austria, we identified the possible weaknesses of the automated recession extraction procedures for the late-time recession analysis for spring hydrographs. To explore the value of the karst spring's physicochemical data (electrical conductivity and water temperature) as a completion data for the recession curve analysis, we carried out the hydro-chemograph analysis to examine the recession time and its duration. The research provides a research direction as to how the automated recession extraction procedures for the karst spring hydrographs could be improved by the physicochemical signatures of karst springs.
Keywords: *automated recession extraction methods (REMs), karst spring, hydrograph analysis, hydro-chemograph analysis, recession curve analysis*

Öz

Kaynak hidrograflarında Çekilme Eğrisi Analizi, karstik akifer sistemlerinin akım ve boşalım dinamiklerini karakterize etmek için kullanılan yaygın bir yöntemdir. Bu yöntem, akifer sisteminin hidrodinamik özelliklerinin tanımlanmasında önemli bilgiler sağlamasına karşın, aday bir çekilme eğrisi(leri)nin elle seçimi ne tüm çekilme eğrilerinin analizini kapsayacak şekilde pratik bir tekniktir, ne de hidrolojik bir değişiminin karstik kaynak çekilme davranışı üzerindeki etkisinin tanımlanmasına

* *Corresponding author*

izin vermektedir. Bu çalışmada, otomatik çekilme eğrisi seçim prosedürlerinin kaynak hidrografı geç-dönem çekilme analizlerinde uygulanabilirliği araştırılmıştır. Bu kapsamda, Vogel Metodu, Brutsaert Metodu ile Aksoy ve Wittenberg Metodu olmak üzere üç otomatik çekilme eğrisi seçim prosedürü dört adet çekilme eğrisi analizi metodu (Maillet, 1905, Boussinesq, 1904, Coutagne, 1948 ve Wittenberg, 1999) ile birleştirilerek karstik kaynaklarda geç-dönem kaynak çekilme (boşalım) katsayıları karşılaştırmalı olarak hesaplanmıştır. Çalışmada, Avusturya'da bulunan beş karstik kaynağın çekilme katsayıları belirlenmiş olup karstik kaynak fizikokimyasal verileri (elektriksel iletkenlik ve su sıcaklığı) hidro-kemograflar yardımıyla değerlendirilerek otomatik çekilme eğrisi seçim prosedürlerinin olası zayıf yönleri ortaya konmuştur. Bu araştırma, karstik kaynaklarda geç-dönem çekilme eğrisi analizlerinde çekilme başlangıcını ve süresini tamamlamak için otomatik çekilme eğrisi seçim prosedürlerinin uygulanabilirliği için bir araştırma yönü sağlamaktadır.

Anahtar sözcükler: *otomatik çekilme eğrisi belirleme metotları, karstik kaynak, hidrograf analizi, hidro-kemograf analizi, çekilme eğrisi analizi*

Introduction

Recession curve analysis is a hydrogeological tool to characterize the karst aquifer internal hydraulic properties, catchment characteristics, and climate characteristics (Atkinson, 1977; Amit et al., 2002; Dewandel et al., 2003; Yüce, 2007; Fiorillo, 2014; Ford & Williams, 2013; Kovács & Perrochet, 2008; Padilla & Pulido-Bosch, 1995). For this reason, the structure of mathematical models for the recession curve analysis of karst spring hydrograph is studied in-depth with a particular focus on the quantification of the storage-discharge relationships in karstic aquifers.

Due to the dual-flow characteristics of the karstic aquifers, each segment on the recession curve is characterized by – at least – two flow components, each of which informs about the different sub-regimes in the hydrological system (Bonacci, 1993; Estrela & Sahuquillo, 1997; Fiorillo, 2014; Kovacs et al., 2005; Stevanović, 2015), thereby leading to several recession coefficients that characterize the distinct flow components on a single recession curve (Xu et al., 2018). To a certain extent, while the late-time recession segment on the recession curve represents the more stable part of the spring hydrograph, while indicating the maturity of the system's hydrological response, the early-time recession segment represents rather flashier characteristics of the system of interest (Birk & Hergarten, 2010; Tallaksen, 1995). Therefore, on a single recession curve, while the baseflow characteristics of the karstic system are mainly linked to the matrix-dominated flow component defined as the behavior of the late-time recession curve, the fractured and/or conduit-dominated flow is tracked by the early-time recession behavior. Furthermore, the transition flow between these distinct flow regions could develop, thereby resulting in more than two inflection points on the single recession curve.

The extraction of the candidate recession segment(s) from the recession curve is the first step to analyze the recession characteristics based on the recession parameters. Thus, the recession curve analysis inherently covers the identification of the recession time (or initial discharge value of the recession curve/segment) and recession length (or recession duration) of a candidate recession curve(s). For this reason, these two variables of the recession curve are of great importance to obtain the recession coefficients while covering the recession variability over the spring hydrograph analysis. Therefore, for the identification of the recession time and its duration in the recession curve, different methodological approaches have been proposed with a primary aim to eliminate the potential biases and uncertainties – mainly sourcing from the implementation procedure such as the single-event analysis and master recession curve analysis (Gregor & Malik, 2002; Nathan & McMahon, 1990).

As a traditional karst spring hydrograph analysis approach, the manual selection of a candidate recession curve/segment is the main step to define the recession parameters, followed by the implementation of an appropriate conceptual model for the candidate recession curve –under either the linear reservoir or non-linear reservoir model assumptions. Of all, the linear reservoir model – known as Millet exponential formula – is commonly used for the karst spring hydrograph analysis for the delineation of recession characteristics. This analysis simultaneously covers the matrix-dominated and conduit-dominated flow segments (Çelik & Çallı, 2021; Forkasiewicz & Paloc, 1967; Fu et al., 2016).

Despite the merit of the traditional recession curve analysis (Biswal & Marani, 2010; Shaw & Riha, 2012), the flow characteristics of any hydrological system cannot be only derived by an individual recession curve/segment (Fiorotto & Caroni, 2013). This is mainly because the structure of the recession curve (shape and degree of steepness) significantly varies from one hydrological event to another. Along with this, the manually selected recession curve procedure is neither a practical technique to cover all candidate recession curves, nor it allows extracting the entire hydrological diversity of the karst spring recession behavior (Calli & Hartmann, 2019). For that reason, the recession analysis should be collectively performed to capture the hydrological variability of discharge dynamics and its hydraulic properties considering a long data record (Chen & Krajewski, 2015; Jachens et al., 2019; Sánchez-Murillo et al., 2015; Stewart, 2015; Stoelzle et al., 2013). In this context, the framework of the automated recession curve extraction procedures for the streamflow hydrograph analysis has been gained attention as an alternative approach to objectively extract candidate recession curves(s) while delineating the catchment baseflow conditions (– or late-time recession characteristics) of the streamflow. Therefore, this framework provides an opportunity to eliminate the potential biases and uncertainties caused by the subjectivity of the manually selected curve procedure, thereby allowing to capture the hydrological diversity of the system of interest based on the recession behaviour.

Since the applicability of the recession curve extraction procedure is still not being evaluated for the recession curve analysis in karst spring hydrographs for the delineation of the late-time recession characteristics, the overall goal of this study is to investigate the applicability of the automated recession curve extraction procedures to the late-time recession parameters of the karst spring hydrograph. To achieve so, we applied three automated recession extraction methods (REMs), which are specifically developed to characterize the baseflow characteristics of the streamflow hydrographs. By coupling these three REMs – Vogel; Brutsaert; and Aksoy and Wittenberg Methods – with four recession analysis methods (RAMs) – Maillet (1905); Boussinesq (1904); Coutagne (1948); and Wittenberg (1999) Methods –we comparatively evaluated the applicability of each procedures for the estimation of the late-time recession parameters of each karst springs. Doing that, we simultaneously examined the variations in the range of recession parameters in response to the applied recession extraction procedure. Therefore, to reveal the possible weakness of REMs procedures, we carried out spring hydro-chemograph analysis for the identification of the late-time recession time and its duration considering the independent physicochemical data of spring discharge.

Method

Data Sets

To reveal to what extent the REMs is applicable for the characterization of spring hydrograph's late-time characteristics we selected five karst springs in Austria. Each spring reflects the different hydrological flow regimes (Figure 1). The main properties of the springs are provided in Table 1. The recession curve analysis was performed daily over the 10 years (01/01/2002–31/12/2012). Daily precipitation, spring discharge, and physicochemical dataset including electrical conductivity and water temperature were obtained from https://ehyd.gv.at/#.

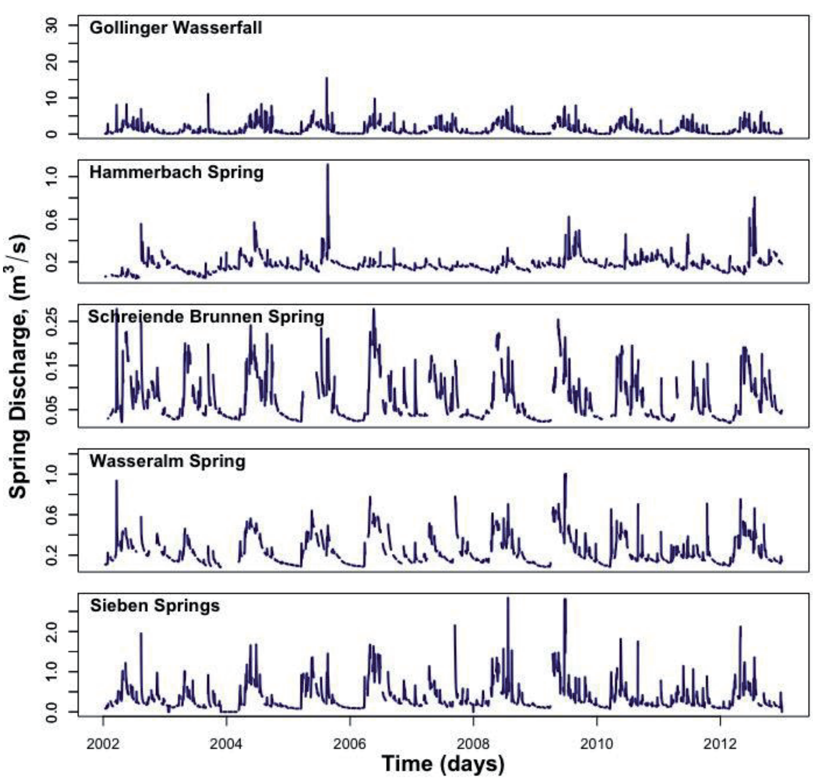
Table 1

Metadata of the Selected Karst Springs in Austria

Spring Name	Elevation (m.asl)	Geology	Mean Annual Discharge (m^3/s)
Gollinger Wasserfall	555	Cretaceous Limestone	1.25
Hammerbach Spring	410	Paleozoic carbonate rocks	0.19
Schreiende Brunnen	980	Limestone	0.08
Wasseralm Spring	802	Triassic Limestone and dolomites	0.24
Sieben Springs	797	Triassic Limestone and dolomites	0.37

Figure 1

The Karst Spring Hydrographs in Austrian Site Over the Period of 01/01/2002 – 31/12/2012



Applied Procedure to Define Late-time Recession Characteristics of Hydrographs

To define the late-time recession characteristics of the karst springs we used the HYDRORECESSION toolbox developed by Arciniega-Esparza et al. (2017) in MATLAB environment while R-Studio was used for the post-processing analysis of the obtained recession parameter sets.

For the estimation of hydrograph recession parameters, we first applied three automated REMs to extract the candidate recession segments from each recession curve from spring hydrographs during the period of 01/01/2002 - 31/12/2012. The REMs procedures are the Vogel Method (Vogel & Kroll, 1992); Brutsaert Method (Brutsaert & Nieber, 1977; Brutsaert, 2008), and Aksoy and Wittenberg Method (Aksoy & Wittenberg, 2011). After the extraction of each recession segment from the 10-year data record, four RAMs including Maillet (1905); Boussinesq (1904); Coutagne (1948); and Wittenberg (1999) were applied to the hydrographs for the estimation of the late-time recession parameters. Then, we compared each parameter estimation procedure by referring to the different combinations of REMs and RAMs. Furthermore, to reveal the variations in the value of estimated parameters due to the applied parameter-fitting techniques (PFTs), the linear regression, lower envelope, and data binning methods were used for the estimation of the recession parameters.

Recession Curve Analysis and Recession Plots

To comparatively evaluate the automated recession curve methods, we performed the automated recession extraction procedures to the karst spring hydrographs. All methods are already successfully applied to the streamflow hydrographs to analyze the baseflow characteristics of the catchment hydrology while discarding the influence of the storm events on the early-time response of the recession curve to capture the late-time hydrological response (baseflow characteristics) of the catchment.

As an analytical model parameterization method, the late-time recession analysis is performed based upon the recession slope curve (hereinafter referred to as hydrograph recession plot) analysis. This analysis involves the selection/extraction of a candidate recession segment(s) and plotting the log – log graph of the recession rate (dQ/dt) as a function of the discharge (Q). This linking approach based on the Boussinesq equation is proposed by Brutsaert and Nieber (1977) to eliminate time dependencies on recession curve analysis (Rupp & Selker, 2005).

The hydrograph recession plot defines the storage-discharge relationship by the slope of the hydrograph (- dQ/dt, (LT^{-2})) and discharge (Q, (LT^{-1})) using a power law of form of storage-discharge relationship expressed by;

$$\frac{-dQ}{dt} = aQ^b \tag{1}$$

where b and a are the recession parameters, referring to the power coefficient (–) and recession coefficient (T^{-1} or $(L^3/T)^{1-b}$), respectively. These parameters vary from catchment to catchment (Brutsaert & Nieber, 1977), thus indicating the instinct properties of the hydrological system (Rimmer & Hartmann, 2012). In Eq. 1, the discharge rate (- dQ/dt) is computed by the differences of two consecutive points on the extracted recession segment ($dQ/dt = (Q_{i+1} - Q_i)/\Delta t$) while Q is calculated as a mean value of these discharge values ($Q = (Q_{i+1} + Q_i)/2$).

In general, the exponent b (–) ranges from less than 1 to larger than 3 (Chapman, 1999; Harman et al., 2009; Kirchner, 2009; Wittenberg, 1999), which is attributed to the catchment heterogeneities (Clark et al. 2009; Tague & Grant, 2004). As a special case, when the exponent b (–) is equal to 1, the hydrological system acts as a linear reservoir, referring to the main concept formula for the recession curve analysis in the karst spring hydrograph analysis – known as the Maillet's exponential formula by Maillet (1905):

$$Q_t = Q_0 e^{-\alpha t} \tag{2}$$

where Q_0 and Q_t are the initial discharge and the discharge at the time, t, respectively. α is the recession coefficient (T^{-1}) indicating the intrinsic hydraulic properties of aquifer system.

To obtain late-time recession parameters of the karst spring hydrograph we used the recession plot approach (- dQ/dt vs. Q) which overlaps the multiple individual recession segments extracted by four REMs. Each procedure for the parameter estimation is detailed in Table 2 and Table 3. After applying the recession plot to estimate the recession parameters of b *(–)* and a *(T^{-1})*, to assess model performance on late-time flow analysis and the quality fit of the recession plots the performance metrics of Nash Suffice Efficiency, NS and Coefficient of Determination, R^2 were performed.

$$NS = 1 - \frac{\sum_{i=1}^{N} |Q_{sim,i} - Q_{obs,i}|}{\sum_{i=1}^{N} |Q_{obs,i} - \overline{Q_{obs}}\ |} \tag{3}$$

$$R^2 = \left(\frac{\sum_{i=0}^{N} (Q_{obs,i} - \overline{Q_{obs}})(Q_{sim,i} - \overline{Q_{sim}})}{\sqrt{\sum_{i=0}^{N} (Q_{obs,i} - Q_{obs})^2} \sqrt{\sum_{i=0}^{N} (Q_{sim,i} - \overline{Q_{sim}})^2}} \right)^2 \tag{4}$$

where $Q_{obs,i}$ and $Q_{sim,i}$ are the observed and simulated spring discharges at time, *i,* while $\overline{Q_{obs}}$ and $\overline{Q_{sim}}$ represent the mean values of the corresponding variables.

Procedure for the Estimation of Recession Parameters

For the estimation of recession parameters of five karst springs in Austria, we used three REMs and four RAMs provided in the HYDRORECESSION software toolbox. The algorithms of the recession curve extraction and recession analysis methods are summarized in Table 2 and Table 3.

To comparatively assess the REMs, the recession coefficients were only calculated by the Maillet exponential formula as it is the most preferable recession analysis method for spring hydrograph recession analysis. Additionally, we used PFTs which are broadly used for the model fitting techniques over the hydrograph recession plot [log (- dQ/dt) vs. log (Q)].

For the sake of simplicity, we used the abbreviations of VG, BRU, and AWM for the Vogel Method, Brutsaert Method, and Aksoy and Wittenberg Method, respectively. Similarly, the abbreviations of MAI, BOU, and WIT were used for the Maillet (1905), Boussinesq (1904), and Wittenberg (1999) Methods. During the recession analysis, the PFTs including the linear regression, lower envelope, and data binning were also shortened by LR, LE, and BIN, reciprocally.

To refer to the dual combination of REMs and RAMs for the procedure of recession parameter estimation we used the mathematical symbol of the intersection "∩". For instance, we called VG ∩ MAI to combine the Vogel recession extraction method with the Maillet recession analysis method.

Table 2

Recession Extraction Methods (REMs)

Recession Extraction Methods	Criterion	Minimum Duration* (days)	Filter Criterion* (removed days)	Exclusion of Anomalous Recession Decline*
Vogel and Kroll, 1992	Decreasing 3-day moving average	10	First 30 %	$\frac{Q_i - Q_{i+1}}{Q_{i+1}} > 30\ \%$
Brutsaert and Nieber, 1977	$\frac{dQ}{dt} < 0$	6-7	First 3-4 Last 2	$\frac{Q_i - Q_{i+1}}{Q_{i+1}} > \frac{dQ}{dt}$
Aksoy and Wittenberg, 2011	$\frac{dQ}{dt} < 0$	5	First 2	$CV > 0.20$

*Editable features provided by the HYDRORECESSION Toolbox. We used the same properties as provided.

Table 3

Recession Analysis Methods (RAMs)

Recession Analysis Methods	Storage-Discharge Relationship	Recession Curve Equation	Parameter-Fitting Techniques
Maillet, 1905	$S = \frac{Q}{\alpha}$	$Q_t = Q_0 e^{-\alpha t}$	Mean Square Error
Boussinesq, 1904	$S = \int f(Q)dt$	$Q_t = \frac{Q_0}{(1+nt)^2}$	Least Squares
Coutagne, 1948	$\frac{dQ}{dt} = -\alpha Q^b$	$Q_t = \frac{1}{\left[{Q_0}^{1-b} - (1-b)\alpha t\right]^{1-b}}$	Linear Regression Lower Envelope Data Binning
Wittenberg, 1999**	$S = cQ^d$	$Q_t = Q_0 \left[1 + \frac{(1-d)Q_0}{cd}\right]^{\frac{1}{(d-1)}}$	Mean Square Error

**provided by the HYDRORECESSION Toolbox. In the paper, the recession parameters, c and d, of the Wittenberg Method are referred to a and b, respectively.

Incorporation of Hydro-Chemographs into Late-time Recession Curve Analysis

To reveal the late-time recession time and its duration based on the physicochemical response of the karstic aquifers we examined spring hydro-chemographs by relating this process-based knowledge with the aquifer internal flow dynamics. To do so, we considered the dynamic hydrological response of the karstic aquifer considering karst spring physicochemical data including electrical conductivity (*EC)* and water temperature (*T)* in response to variations of spring discharge (Q). This, therefore, allowed us to examine to what extent the late-time recession time and its duration could be captured by the hydro-chemograph analysis.

Results and Discussions

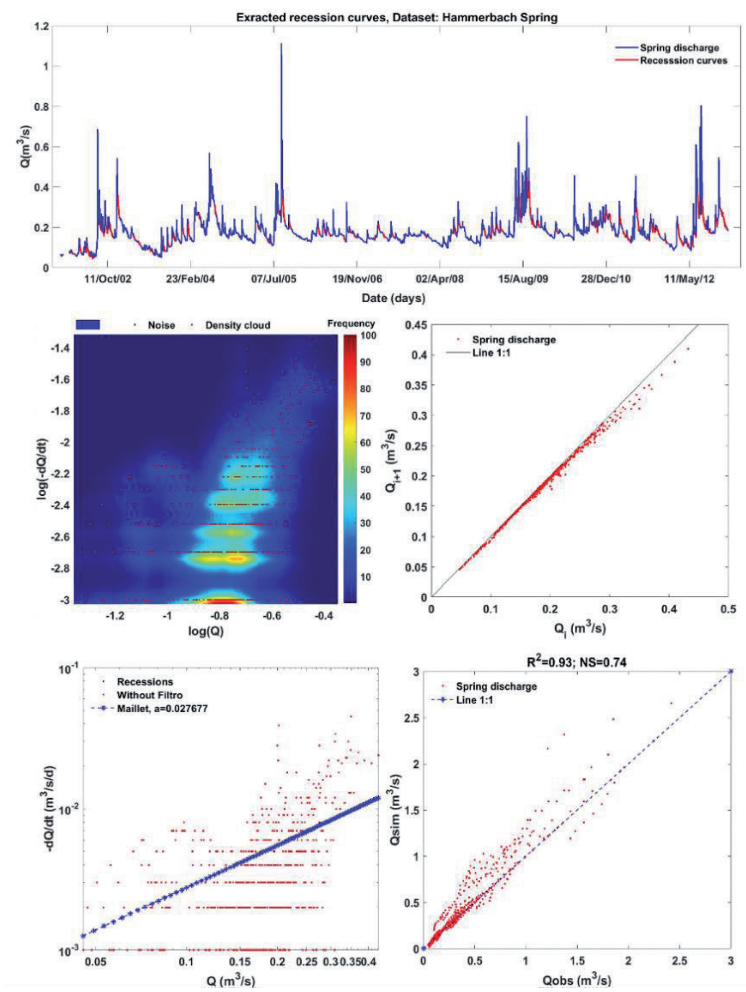
Extraction of Late-time Recession Curves

Figure 2 and Figure 3 demonstrate how an automated REM was implemented into the spring hydrograph recession curve analysis while collectively analyzing the recession parameters over the 10-year discharge data. In both figures, the spring hydrographs are also accompanied by the recession plots [log (- dQ/dt) vs. log (Q)] in which the aggregation of all candidate recession curves (hereinafter referred to the 'point cloud') and the plot of linearity of the selected points (Q_{i+1} vs. Q_i) are provided. Here, the model results for the Hammerbach spring and Gollinger Wasserfall are particularly given for the comparison of the main differences between the automated recession curve analysis, which are only resulted from the hydrological regimes of both springs. The BRU recession extraction method was used to exemplify the implementation procedure of the REMs in each spring while the Maillet method was selected as a reservoir model. Furthermore, Appendix Figure A1 demonstrates the model results for the Gollinger Wasserfall spring with the selection of three REMs over the period of 01/01/2002 – 31/12/2012.

Overall, the recession extraction procedure does not necessarily capture the late-time recession behavior of Hammerbach spring (Figure 2), instead mainly extracting the early-time recession characteristics. By comparison, the late-time recession segments are, in general, represented by the extracted curves from the Gollinger Wasserfall during the 10 years (Figure 3). Considering the recession plots – indicated as the point cloud in both figures – the point space for the Gollinger Wasserfall is more consistent, providing a denser space for the point cloud than that of Hammerbach spring. Overall, the dependency of the recession rate (- dQ/dt) on the discharge of the Gollinger Wasserfall is higher to obtain the storage-discharge relationship, thus defining the late-time flow characteristics. However, the extracted segments do not ensure a condensed space for the Hammerbach spring hydrograph, thereby resulting in a rather noise for the estimation of recession parameter, despite the better model performance (NS: 0.74 and R^2: 0.93).

Figure 2

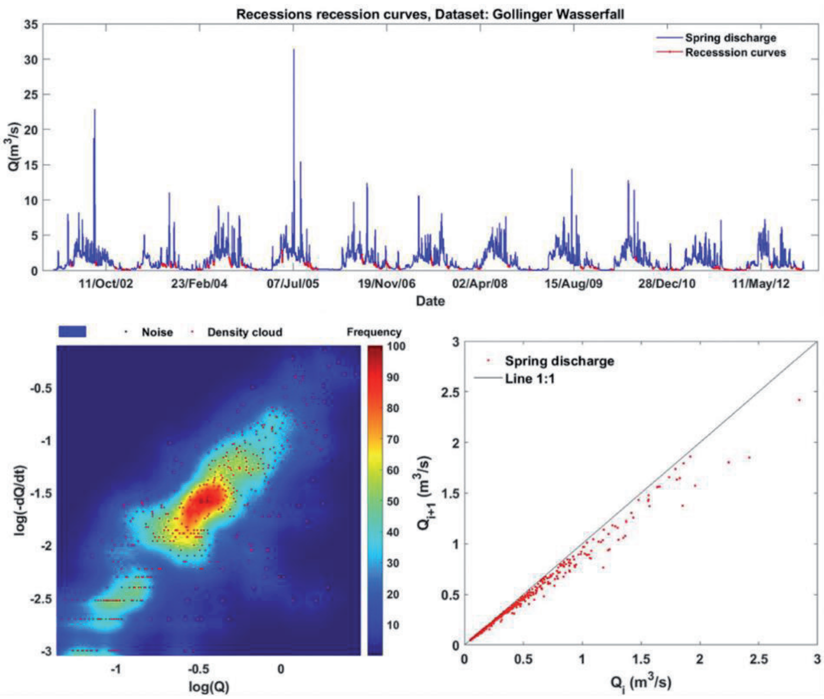
Model Results Obtained from HYDRORECESSION



Note. Upper Panel: Extracted recession curves of Hammerbach spring hydrograph over 10 years. Lower Panel (1): (Left) Hydrograph recession plot is demonstrated as a form of a cloud of points, which indicates the noise reduction from log (- dQ/dt) vs. log (Q). Lower Panel: (Right) Linearity of the dataset based on the simulation with the Maillet method. Lower Panel (2): (Left) Hydrograph recession plot is demonstrated as a form of a cloud of points, which indicates the noise reduction from log (-dQ/dt) vs. (Q). The Lowest Panel: (Right) Model performance on the late-time flow analysis based on the with the Maillet method, and the quality fit of the recession plots demonstrated by the model performance metrics.

Figure 3

Model Results obtained from HYDRORECESSION



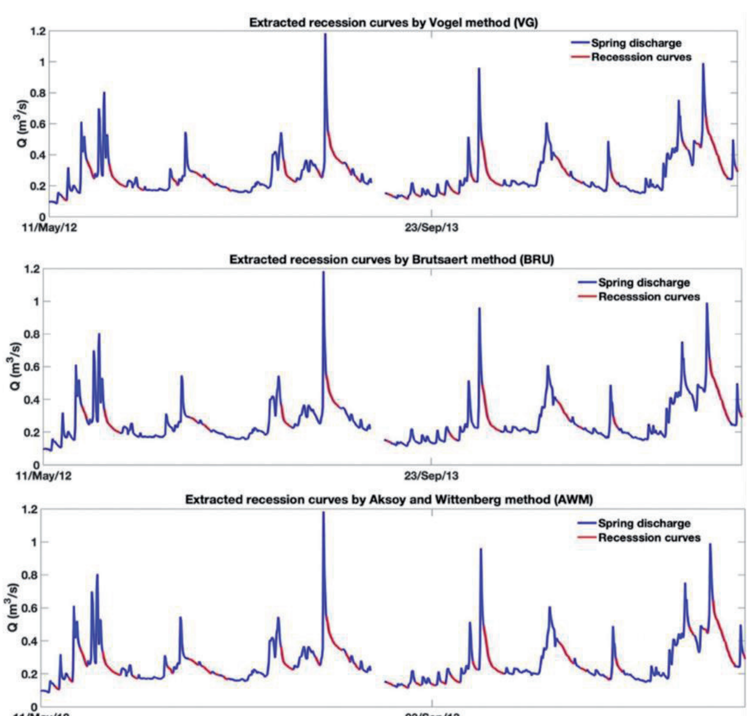
Note. Upper Panel: extracted recession curves of Gollinger Wasserfall for 10 years (01/01/2002 – 31/12/2012). Lower Panel: (Left) Hydrograph recession plot is demonstrated as a form of a cloud of points, indicating the noise reduction from log (- dQ/dt) vs. log (Q). *Here, the red points correspond to the clouds with higher frequency.* Lower Panel: (Right) Linearity of the dataset based on the model simulation with the Maillet method.

For the comparison of three REMs, Figure 4 exemplifies the extracted curves from the Hammerbach spring hydrograph over a one-year time-window. Overall, the BRU method extracted the 14 recession segments (a = 0.039 day^{-1}, NS = 0.71, R^2 = 0.91) during which the VG method extracts 27 segments (a = 0.035 day^{-1}, NS = 0.75, R^2 = 0.93) whereas the AWM method selected more segments with 31 segments, in total (a = 0.020 day^{-1}, NS = 0.89, R^2 = 0.98). Figure 4 also reveals that some extracted recession segments were interrupted by a sudden - but rather small – increment in the spring discharge, which limits to obtain the longer recession curves. This ultimately led to the extraction of a rather short recession segment. For that reason, it is not necessarily possible to cover a long recession curve when the recession rate

(- dQ/dt) is exposed to a small increment over the recession curve. Therefore, considering the flow characteristics of the extracted segment(s), the REMs procedures mainly captured the transition-time and late-time recession characteristics. This implies that all three methods do not necessarily correspond to the late-time recession characteristics in the spring hydrograph in each level of the recession curve analysis. Figure 4 confirms that the AWM method extracts more recession segments while capturing the recession variability much better, thereby providing with candidate recession segments for the spring hydrograph recession curve analysis.

Figure 4

The Comparison of the REMs during the Automated Recession Curve Extraction Procedure for the Hammerbach Spring

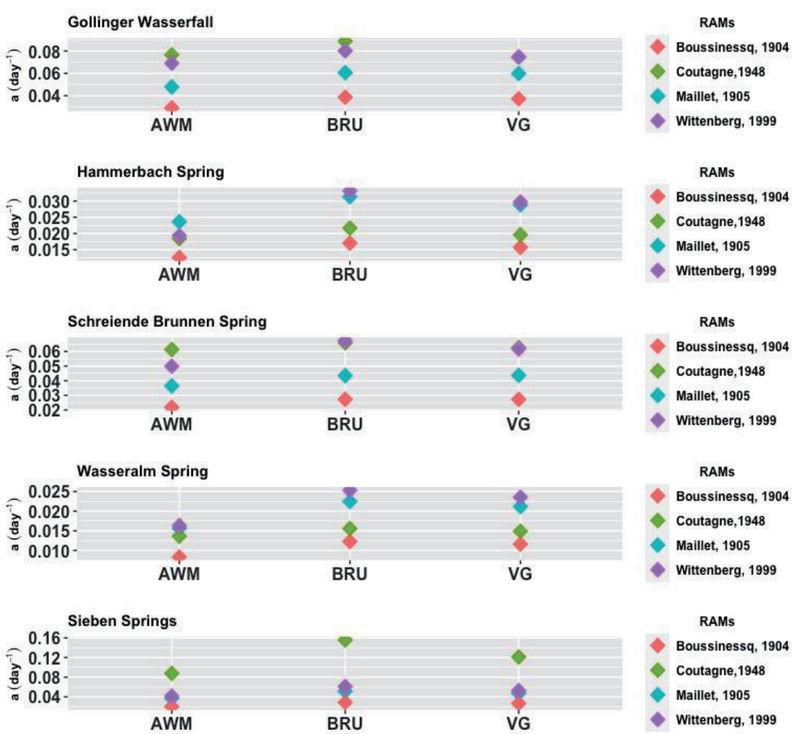


Variations in Obtained Recession Coefficients Due to the Implementation Procedure

The variations in the obtained late-time recession coefficients, a (day^{-1}) of the karstic springs are indicated in Figure 5. Overall, the estimated values were mainly influenced by which parameter estimation procedure was applied into the spring hydrograph analysis, thereby bringing in a wide range of parameter range. Of all dual combinations with RAMs, the AWM method consistently provided the lowest recession coefficients, followed by the VG and BRU methods, respectively. Similarly, as a reservoir model, the Boussinesq method ensured the estimation of the lowest recession coefficient by all dual combinations with three REMs regardless of which type and shape of the spring hydrograph were under the examination. Therefore, the results in Figure 5 are also supported by the previous findings about the streamflow hydrograph recession curve analysis (Stewart, 2015; Chen & Krajewski, 2016) such that the parameter estimation is strongly influenced by which procedure is implemented into the recession curve analysis. Yet here, the only exception would be the Boussinesq method which delivered the lowest values in the parameter range among all dual combinations by REMs and RAMs.

Figure 5

Variations of the Estimated Recession Coefficients for the Austrian Site Springs



Note. The recession parameters for the karstic springs are obtained from the 10-year spring discharge data records (01/01/2002 – 31/12/2012).

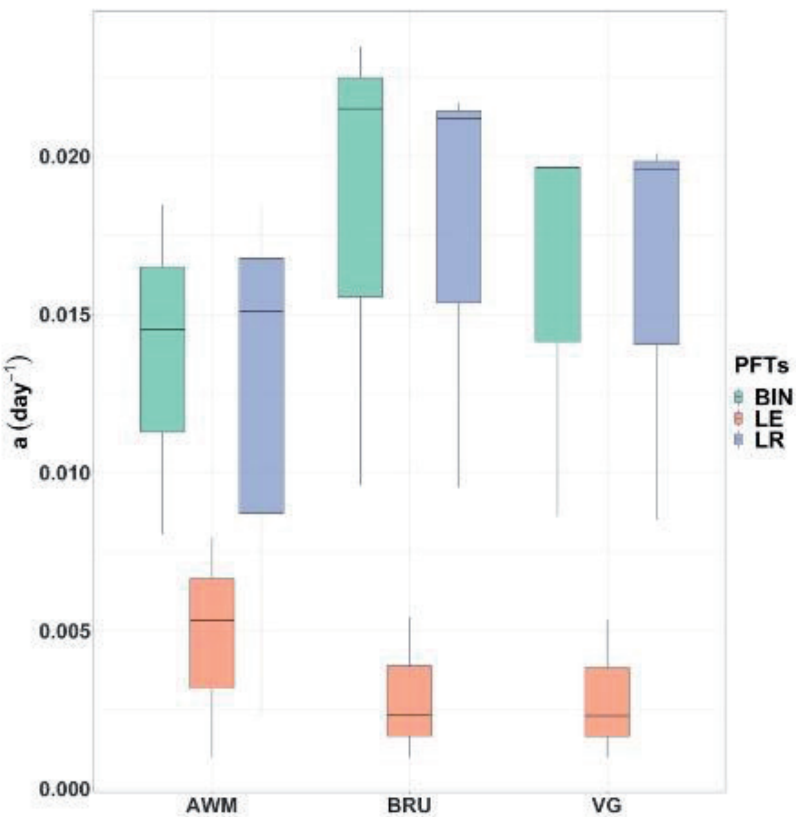
The box-and-whisker plots in Figure 6 demonstrate the sensitivity of the annual recession coefficient, *a* (day^{-1}) to the selection of the PFTs. The recession analysis results are also provided in Appendix Figure A2 for the Hammerbach spring over 01/01/2002 – 31/12/2012. Here, the Coutagne method was used to estimate the recession coefficients as it mainly gives insight into the non-linearity of a hydrological system due to the different *b* (–) values. Overall, the lower envelope (LE) with all REMs provided the lowest parameter values while ensuring a strictly confined parameter space with a lower interquartile range (IQR). This, therefore, gave less uncertain parameter estimation. Similarly, the AWM method enabled to obtain the lower recession coefficients as compared to the BRU and VG methods, particularly allowing a larger interquartile range with the linear regression (LR) method. As opposed to the AWM method, the higher values of the recession coefficient obtained by the combination of BRU ∩ LE and BRU ∩ LR amplified the parameter uncertainty, as indicated by the wider IQRs. Furthermore, the LE naturally contributed to the lower values with all dual combinations by each REM, thereby providing a less uncertain parameter range. Hence, Figure 6 confirms that the selection of PFTs inherently designs the estimation of the recession parameter(s) as much as the implementation of the REMs and RAMs procedures. From this point of view, it would be possible to infer that the dual combination procedure of the recession parameter estimation would serve to obtain different parameter ranges depending on the research target – preferably to estimate either lower or higher recession coefficients.

Incorporation of Hydro-Chemograph Analysis into Late-Time Recession Curve Analysis

Figure 7 and Figure 8 exemplify the main difference between the physicochemical response of Sieben springs and Hammerbach spring, accompanied by the spring hydrographs. In general, the temporal variations in *EC* for both springs did not have the same behavior during certain time-periods, particularly over the summer during which the young water came into the hydrological system via the storm event(s). The same hydrological response could also be tracked over the 10-year period (2002-2012) (see Appendix Figure A3-A4).

Figure 6

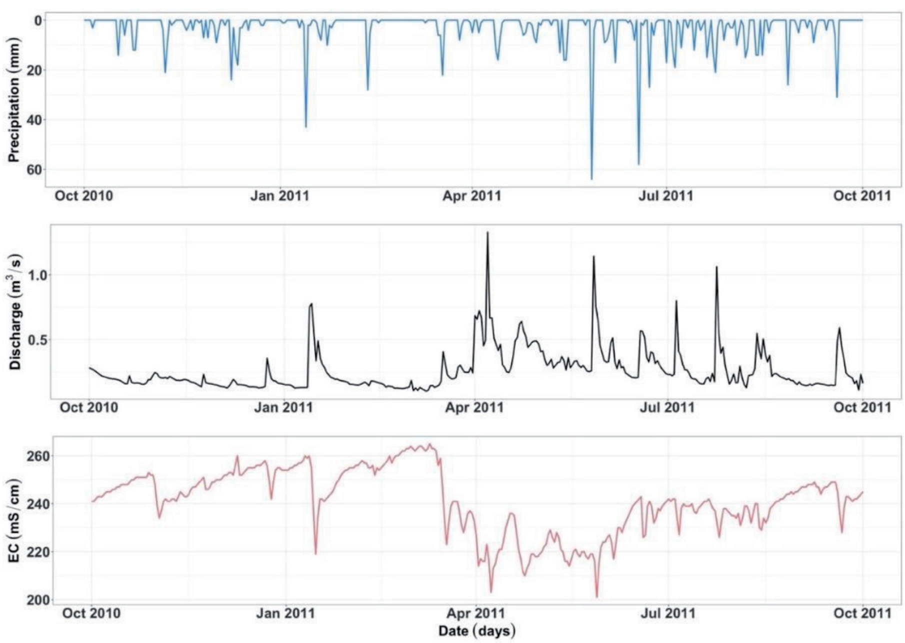
The Range of the Annual Recession Coefficients for the Hammerbach Spring



Note. Here, the Coutagne method was used as a RAM for the model parameterization. The boxplots indicate the 25th and 75th percentile of the annual recession coefficients. The black line in each plot demonstrates the median value of the estimated values.

Figure 7

A Time-Window for the Physicochemical Response of the Sieben Springs Over A One-Hydrological Year (01/10/2010 - 01/10/2011)



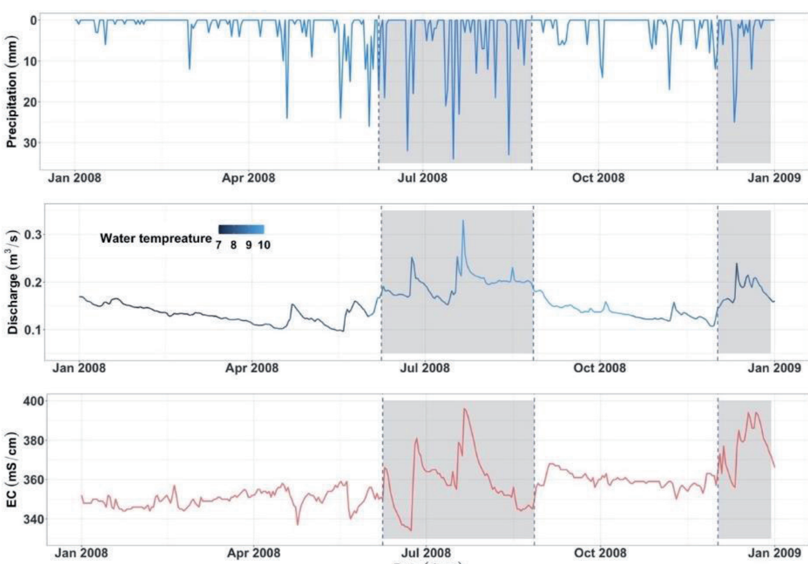
Note. Here, when the young water comes into the aquifer system by the storm (precipitation) event, the old water in storage is quickly mobilized by the propagation of hydraulic pulses, thereby resulting in a corresponding decline in EC.

In Figure 7, the physicochemical response of the hydrological system is mainly characterized by the system hydrological process in such that after the entrance of the new coming water into the karstic system, the stored old water is quickly mobilized by the propagation of hydraulic pulses. This process is primarily known as the 'piston effect' (Ford & Williams, 2013), typically seen in karstic aquifers, thereby resulting in a sudden decline in *EC* in response to the increment of spring discharge. To a lesser extent, this typical response of the karstic system entails the strong cross-correlation between storm events and spring discharge (Fiorillo & Guadagno, 2010; Ford & Williams, 2013).

As oppose to the physicochemical response of Sieben springs in Figure 7, the positive relationship between the Hammerbach spring discharge and *EC* can be observed during the period of late-May and mid-August in Figure 8. Here, the peaky behavior in the spring discharge and *EC* mainly overlapped each other during the shaded periods. Indeed, this is not a typical or common response observed in karstic hydrological system, which can be explained by the piston effect. Instead, the chemical response of the system in the shaded areas could be related to the hydrogeological settings of the karstic aquifer. For that reason, a reasonable explanation is that that the epikarst zone could consist of easily soluble evaporitic rocks such as gypsum, thus leading to a substantial increase in *EC* followed by a snowmelt period around May during which the snowmelt dominates the physicochemical response of Hammerbach spring. Therefore, the shaded time-periods in the Figure 8 are not necessarily informative about the karst spring hydrograph late-time recession characteristics, especially while resembling the late-time recession time and duration.

Figure 8

A Time-Window for the Physicochemical Responses of the Hammerbach Spring During the Period of 01/01/2008 - 01/01/2009



Note. The spring discharge hydrograph is classified considering the variations in the temperature. The hydrograph is also accompanied by the temporal variations in the *EC*.

As for the temporal relationship between Q and *T* for Hammerbach spring in Figure 8, the lowest values of *T* primarily characterize the matrix-dominated flow regimes whereas an increase in the spring discharge leads to a decline in *T* due to the dilution effect. More importantly, the time-period during which the discharge flow component (either conduit or matrix, or a combination of both) dominating the spring flow can be identified by the analysis of the temporal variation of *T* in karst spring. For instance, at the beginning of the time-period – indicated by the shaded areas – the spring flow still reflects the matrix-dominated discharge characteristics with the values of *T* varying between 7°C and 8°C. However, the substantial increase in the water temperature, *T* after mid-July could be explained by the domination of the conduit/fracture flow component on the spring hydrograph. Therefore, as compared to *EC*, the temporal variation in *T* is particularly important as it is more likely to bring the process-based system knowledge into the pure late-time recession curve/segment analysis.

Conclusions

Our research attempted to explore the applicability of the REMs procedures to the late-time recession analysis in the karst spring hydrograph. To do so, we comparatively evaluated the REMs by combining with four RAMs, while obtaining late-time recession parameters.

Our results confirmed that although the procedures of the REMs for the streamflow recession curve analysis is a convincingly systematic approach to objectively extract the candidate curves (– or segments), it is not necessarily possible to capture the late-time recession characteristics of karst spring hydrograph by the REMs during in our research. In fact, the primary problem encountered over five spring hydrograph analysis was that the candidate recession curves/segments did not necessarily reflect a certain type of flow characteristic in each hydrograph. Instead, the automated REMs extracted the different recession segments – mainly considering the decreasing discharge rate (- dQ/dt) – while reflecting the different sub-regimes in the karstic hydrological system. Therefore, since each REM procedure mainly ignores the early-time recession behavior to analyze the catchment baseflow characteristics from the streamflow hydrograph under the assumption that the early-time recession segment is more frequently influenced by the storm events – considering its intensity and duration –, it might not be possible to define the recession characteristics of the conduit-dominated flow mechanism, either. At this point, when an automated recession procedure is applied to the spring hydrograph analysis it would be wise to first decide upon which type of hydrograph (e.g., flashy, or steady) is under the examination, then to estimate the recession parameters by those recession segments.

To define the recession time and the duration for the spring recession curve analysis, it is also reasonable to couple spring hydrograph analysis with the hydro-chemograph analysis to reliably extract the candidate recession curve/segments based on the hydro(geo)logical process over the karstic system. This, therefore, leads to capturing the hydrological process knowledge of which distinctive flow mechanism could be more dominant on the recession curve.

Due to the hydrological and climatological characteristics of the karstic hydrological system, the automated recession curve selection approach is of great importance to reduce the potential uncertainties conveyed throughout the manually selected curve procedure – which is typically applied in the spring hydrograph analysis –. In this context, our research highlighted the fact that a framework of collectively and automatedly extracted recession curve analysis for karst spring hydrograph is essential to capture the recession variability while eliminating the subjectivity of the manually selected recession curve analysis. Therefore, there is an apparent need to develop an automated recession curve extraction procedure(s) to collectively analyze the recession behaviors of karst spring hydrographs, either characterizing the matrix-dominated flow or quantifying the conduit/fractured-dominated flow characteristics.

Our study provided a research direction to improve the automated recession curve extraction procedures, while drawing a conclusion that the hydro-chemograph analysis is a reasonable complementary technique when this knowledge is coupled by the automated recession curve analysis algorithm. Doing that, the recession time (initial discharge) and its duration (length) of a recession curve/segment could be captured by the physicochemical response of a karst spring, thereby constraining the parameter uncertainty in the recession analysis based on the system process-based knowledge.

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Appendix

Figure A1

Automated Recession Curve Analysis by HYDRORECESSION for the Gollinger Wasserfall Spring over the Period of 01/01/2002 - 31/12/2012

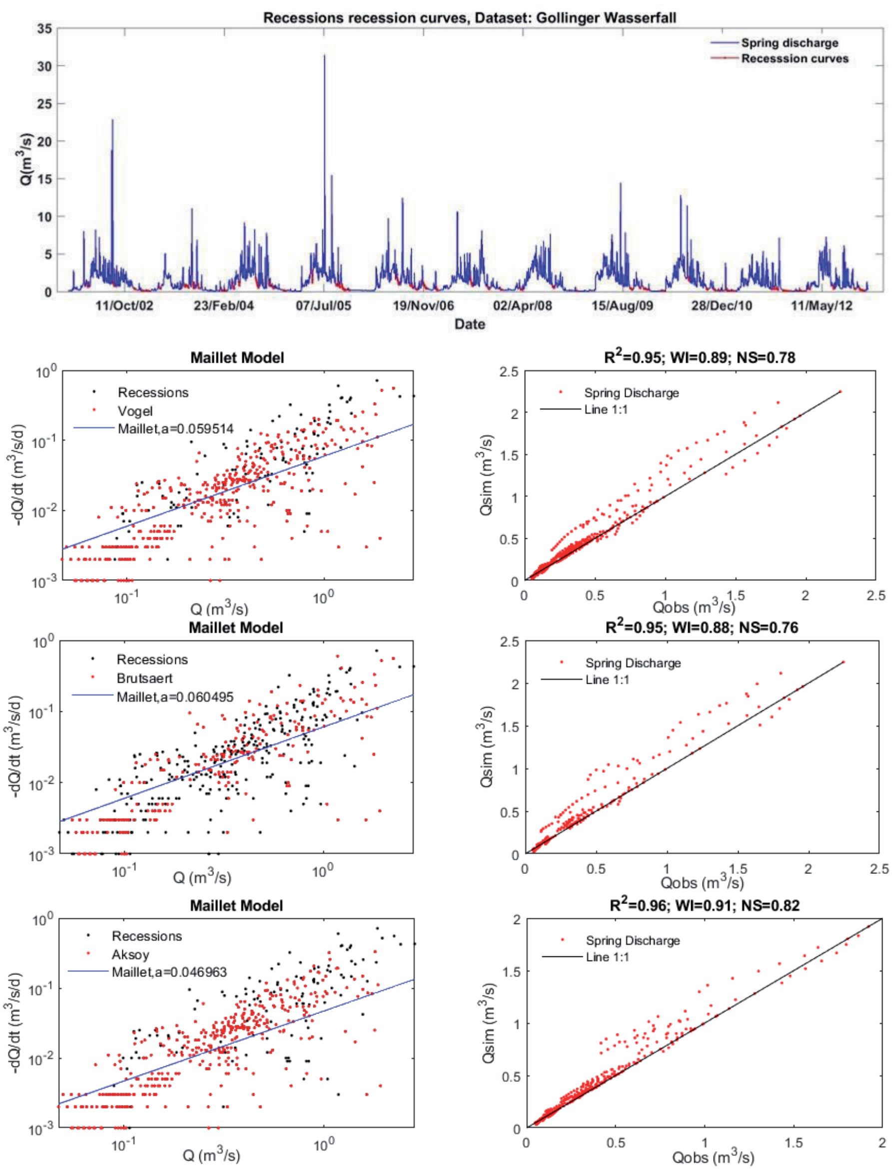
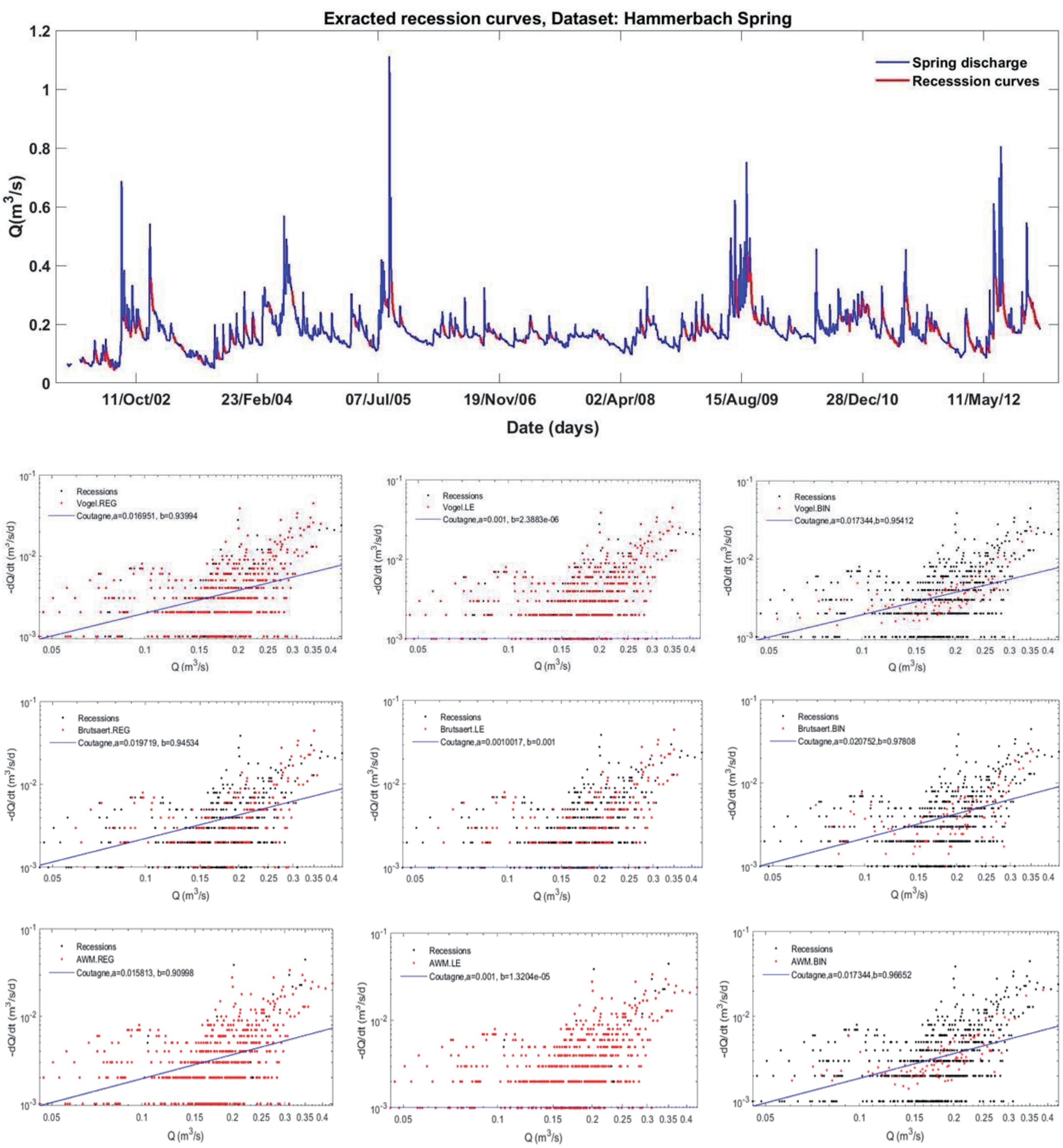


Figure A2

The Model Results Obtained by the Automated REMs for the Hammerbachquelle Spring



Note. REMs and PFTs were compared based on the Coutagne Method.

Figure A3

The Time-Series of the Precipitation, Discharge, and EC for the Sieben Springs Over the Period of 01/01/2002 – 31/12/2012

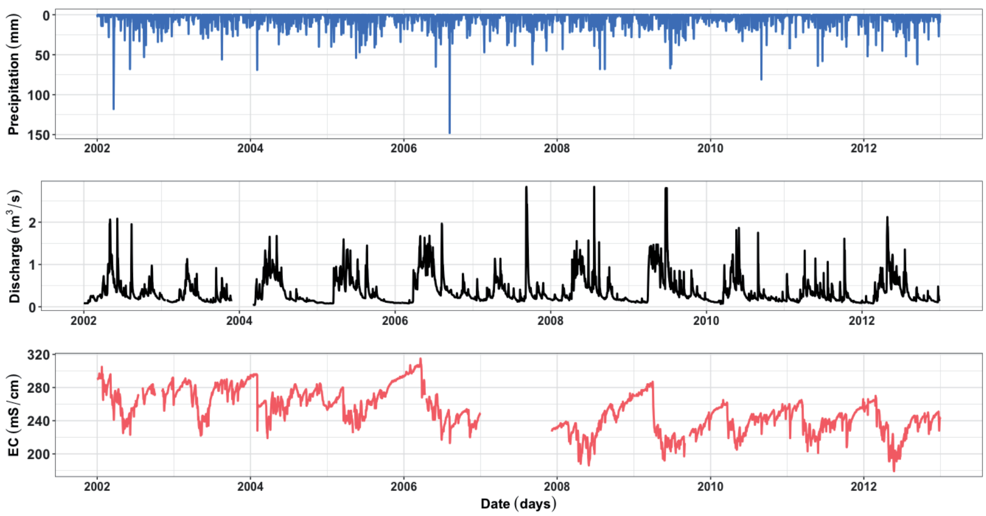
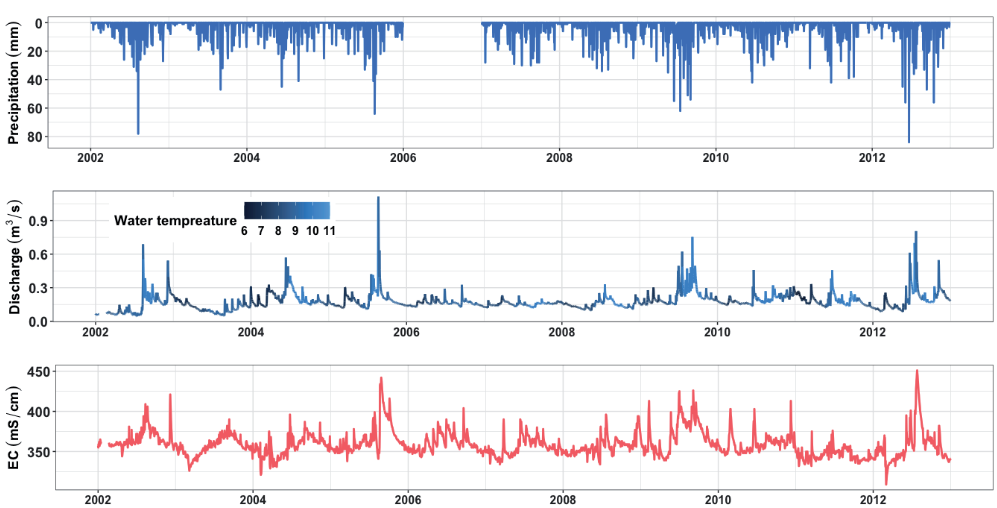


Figure A4

The Time-Series of the Precipitation, Discharge, Water Temperature, and Electrical Conductivity for the Hammerbach Spring Over the Period of 01/01/2002 – 31/12/2012



Extended Turkish Abstract
(Genişletilmiş Türkçe Özet)

Otomatik Çekilme Eğrisi Seçim Prosedürlerinin Karstik Kaynak Hidrograflarında Karşılaştırılmalı Değerlendirilmesi

Çekilme eğrisi analizleri, hidrolojik sistemlerin hidrodinamik özelliklerini tanımlamak amacıyla kullanılan temel ve yaygın analiz yöntemlerinden birisidir. Akım hidrografı çekilme eğrisi analizlerinden farklı olarak, karstik kaynak çekilme eğrisi analizlerinde bir çekilme eğrisi üzerinde birden fazla çekilme segmenti tanımlanmaktadır. Bu segmentler farklı boşalım rejimleri ile karakterize edilmekte olup birden fazla çekilme katsayısı belirlenmektedir. Örneğin, bir karstik kaynak hidrografi üzerinde geç-zaman çekilme eğrisi karstik sistemde taban akışını temsil eden taneli ortam akım özelliklerini yansıtmaktadır. Öte yandan, aynı çekilme eğrisi üzerindeki erken-zaman çekilme segmenti kırıklı-çatlaklı ortamları ve/veya karstik kanal boşalım koşullarını karakterize etmektedir.

Geleneksel bir yaklaşım olarak, karstik kaynak boşalım parametrelerinin belirlenmesinde temel adım, çekilme eğrilerinin başlangıç ve bitiş zamanlarının manuel olarak belirlenerek, ilgili çekilme segmentine bir matematiksel model eğrisinin uydurulması esasına dayanmaktadır. Bu yöntemin uygulanabilirliğinin önündeki en temel kısıtlayıcı unsur, aday bir çekilme eğrisinin araştırmacı tarafından sübjektif olarak seçimidir. Bununla birlikte, karstik sistemdeki boşalım özelliklerinin öznel olarak belirlenmiş tek bir çekilme eğrisine dayanarak türetilmesi de temsil edicilik açısından yeterli olmamaktadır. Bunun başlıca nedeni, çekilme eğrisinin yapısının bir hidrolojik olaydan diğerine önemli ölçüde değişmesidir. Bu temelde, bir karstik kaynak hidrografında çekilme eğrilerinin manuel olarak seçimi, ne tüm aday çekilme eğrilerini kapsayacak şekilde pratik bir tekniktir, ne de uzun dönemli bir kaynak hidrografı çekilme davranışındaki tüm hidrolojik/hidrodinamik çeşitliliğinin tanımlanmasına izin vermektedir. Buradan esasla, karstik kaynaklarda çekilme eğrisi analizleri, uzun bir veri kaydı dikkate alınarak akım dinamiklerinin hidrolojik değişkenliğini ve hidrolik özelliklerini yakalamak amacıyla toplu olarak yapılmalıdır.

Akarsu hidrograf analizi için otomatikleştirilmiş çekilme eğrisi belirleme prosedürlerinin çerçevesi, havza baz (temel) akış koşullarını (– veya geç dönem çekilme karakteristiklerini) betimlerken, çekilme eğrilerinin nesnel olarak tanımlanabilmesi için alternatif bir yaklaşım olarak dikkat çekmektedir. Buradan hareketle, otomatik çekilme eğrisi analiz teknikleri, manuel olarak seçilen çekilme eğrisi secim prosedürünün öznelliğinden kaynaklanan olası belirsizlikleri ve önyargıları ortadan kaldırmak için önemli bir fırsat sağlamaktadır.

Bu çalışma kapsamında, karstik kaynaklarda geç-dönem çekilme eğrisi analizlerinde otomatik çekilme eğrisi belirleme prosedürlerinin uygulanıp uygulanmayacağı araştırılmıştır. Bu kapsamda, üç adet otomatik çekilme eğrisi belirleme yöntemi (Vogel Metodu, Brutsaert Metodu, Aksoy ve Wittenberg Metodu) ile dört adet çekilme eğrisi analiz metodu (Maillet, 1905; Boussinesq, 1904; Coutagne, 1948; Wittenberg, 1999) birleştirilerek karstik kaynak hidrograflarında geç-dönem çekilme eğrisi parametreleri belirlenmiş ve ilgili yöntemler karşılaştırmalı olarak değerlendirilmiştir. Avusturya'da bulunan beş karstik kaynakta (Wasseralm spring, Sieben springs, Hammerbach spring, Gollingen Wasserfall, Schreniende Brunnen) kaynak çekilme eğrisi analizi uygulanarak otomatik çekilme eğrisi belirleme prosedürlerinin olası zayıf yönleri değerlendirilmiştir. Buna ilaveten, karstik kaynak suyu fizikokimyasal verileri (elektriksel iletkenlik ve yeraltı suyu sıcaklığı) kaynak çekilme eğrisinin çekilme başlangıcı ve çekilme süresinin belirlenebilmesi kapsamında değerlendirilmiş ve

karstik sistemdeki hidrodinamik süreçler ile ilişkilendirilerek çekilme eğrisinin tanımladığı boşalım koşulları karakterize edilmeye çalışılmıştır.

Çalışma metodolojisi sırasıyla dört temel adımı içermektedir: (1) otomatik çekilme eğrisi belirleme yönteminin seçimi (REM), (2) çekilme eğrisi analiz metodunun seçimi (RAM), (3) çekilme parametrenin belirlenmesinde eğri uydurma tekniklerinin seçimi ve (4) karstik kaynak hidro-kemografları ile çekilme eğrisi zamanı ve süresinin tahmini.

Çalışma sonucunda, akarsu akışı çekilme eğrisi analizi için geliştirilen REM prosedürlerinin, aday çekilme eğrilerini nesnel olarak çıkarmak/belirlemek için ikna edici sistematik yaklaşım olduğu desteklenmiştir. Ancak, ilgili metotların karstik kaynak boşalım hidrografının geç zaman çekilme özelliklerini yakalamada yeterli olmayabileceği görülmüştür. Bu temelde, çalışmada otomatik REM'ler ile gerçekleştirilen kaynak çekilme eğrisi analizlerinde beş adet karstik kaynakta karşılaşılan temel ve ortak sorun, aday çekilme eğrilerinin farklı kaynak hidrograflarında belirli bir boşalım karakteristiğini yansıtmaması olmuştur. Bunun yerine, otomatikleştirilmiş REM'ler, karstik akifer sistemdeki farklı alt boşalım rejimlerini yansıtmıştır. Burada ilgili yöntemler, kaynak boşalımının zamanla değişimini (- dQ/dt) dikkate alarak farklı çekilme eğrilerini çıkarmayı başarmıştır.

Bir karstik kaynak hidrografi üzerinde erken dönem çekilme segmentinin yağış girdilerinden doğrudan etkilendiği bir gerçektir. Bu durum özellikle kanal baskın boşalımın hâkim olduğu karstik akifer sistemlerinde önemli ölçüde sistem hidrodinamiğini etkilemektedir. Dolayısı ile bu kaynaklarda geç-dönem çekilme özelliklerini otomatik REM'ler ile tanımlamak mümkün olmayabilir. Bu nedenle, boşalım hidrograf analizine otomatik bir çekilme prosedürü uygulanmak istenildiğinde, önce hangi tip akım hidrografının (örneğin kanal ya da gözenek baskın akım) inceleme altında olduğuna karar verilmeli, ardından çekilme parametreleri bu esasa göre otomatik çekilme eğrisi secim metotları ile tahmin edilmelidir.

Karstik bir hidrolojik sistemin otomatik çekilme eğrisi seçimi yaklaşımı, geleneksel olarak uygulanan çekilme eğrisinin manuel olarak seçiminde karşılaşılan potansiyel belirsizlikleri azaltmak için büyük önem taşımaktadır. Bu bağlamda, araştırmamız, manuel olarak seçilen çekilme eğrileri için hesaplanan çekilme katsayıları sonuçları üzerindeki öznelliği ortadan kaldırmıştır. Çalışma ayrıca, karstik kaynak hidrograflarında çekilme değişkenliğini yakalamak amacıyla kaynak boşalım analizi için toplu ve otomatik olarak çıkarılan bir çekilme eğrisi analizi prosedürünün gerekliliğini vurgulamıştır. Çalışma sonucunda, karstik kaynak hidrograf-kemograf analizlerinin otomatik çekilme eğrisi analizleri ile birleştirildiğinde, makul bir tamamlayıcı teknik olduğu sonucuna varılmıştır. Buna ilaveten, bir çekilme eğrisinin çekilme süresinin bir karstik kaynağın fizikokimyasal tepkisi ile yakalanabildiği, böylece sistem sürecine dayalı çekilme analizindeki parametre belirsizliğinin sınırlandırılabileceği sonucuna ulaşılmıştır.

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Research Article

A Flexible Water Quality Modelling Simulator Based on Matrix Algebra

Matris Cebrine Dayanan Esnek Bir Su Kalitesi Simülatörü

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Abstract

For the sustainable management of aquatic ecosystems, an integrated approach is required. This is why watershed-based management is becoming an increasingly popular instrument for the improvement of water quality. Water quality models serve as a central part of the watershed management. Predictive water quality models are valuable tools, but they are usually complex infrastructures in terms of both operation and software development. The aim of this study is to develop the water quality simulator of a larger hydro-ecological modelling framework. Since the water quality problems are diverse, development of one water quality kinetics sub-model that would fit to all water quality problems would be an impossible task. This is the reason why; the water quality simulator software code was developed following the open source philosophy, implemented on a high level (yet high performance) programming language, and documented intensively in-line to enhance the code readability. The water quality simulator software, which is designed as a component of HIDROTURK integrated modelling platform, consists of a general transport sub-model, three water quality kinetics sub-models and utilities.
Keywords: *water quality modelling, eutrophication, open source code, generic pollutants*

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Öz

Su ekosistemlerinin sürdürülebilir yönetimi için entegre bir yaklaşım gereklidir. Bu nedenle havza bazlı yönetim, su kalitesinin iyileştirilmesi için giderek daha popüler bir araç haline gelmektedir. Su kalitesi modelleri havza yönetiminin merkezi bir parçasıdır. Su kalitesi tahminleri yapan modeller değerli araçlardır, ancak işletme ve yazılım geliştirme açısından genellikle karmaşık altyapı bileşenleridir. Bu çalışmanın amacı, daha büyük bir hidro-ekolojik modelleme çerçevesinin su kalitesi simülatörünü geliştirmektir. Su kalitesi sorunları çok çeşitli olduğundan, tüm su kalitesi sorunlarına uyacak tek bir su kalitesi kinetiği alt modelinin geliştirilmesi imkansızdır. Bu nedenle; su kalitesi simülatörü yazılım kodlaması, açık kaynak felsefesini takip ederek yüksek seviyede (ancak yüksek performanslı) bir programlama dilinde gerçekleştirilmiş ve kod okunabilirliğini artırmak için kod içi belgelendirmeye önem verilmiştir. Genel bir taşınım alt modeli, üç su kalitesi kinetiği alt modeli ve yardımcı programlardan oluşan temel su kalitesi simülatör yazılımı HİDROTÜRK entegre modelleme platformu için bir bileşen olarak tasarlanmıştır.

Anahtar sözcükler: *su kalitesi modelleme, ötrofikasyon, açık kaynak kodu, genel kirleticiler*

Introduction

Water quality models can be defined as idealized formulations that represent the response of a physical system to external forcing. The impact-effect relationship between loading and concentration depends on the physical, chemical, and biological characteristics of the receiving waterbodies. Predictive water quality models are important tools for water quality management for aquatic ecosystems. In aquatic science and environmental/water resources engineering, water quality models are used to evaluate the potential impacts of external forcings and to understand the functioning of the system (Thomann & Mueller, 1987; Chapra, 1997; Arhonditsis & Brett, 2004). They are useful tools to get a holistic picture of ecosystems, to fill in the gaps in field data or to forecast the systems responses to different external forcings.

Water quality models have been used throughout the history of environmental and water resources engineering. These models evolved from simple equations to sophisticated modelling software following the new concerns related to the problems in the aquatic ecosystems and consequently rising water quality management problems. The early modelling studies mostly focused on the urban waste load allocation problem. The model developed by Streeter and Phelps (1925) on the Ohio River was the first study in the field. The following studies provided the evaluation of dissolved oxygen levels in streams and estuaries (Velz, 1938; Velz, 1947;

O'Connor, 1960; O'Connor, 1967). Bacteria models were also developed (O'Connor, 1962).

In the 1960s, digital computers became available, which led to major advances in both the models and the ways they are applied. Computers allowed analysts to address more complicated system geometries, kinetics, and time variable simulations; however, dissolved oxygen was still the main focus. The computers also allowed a more comprehensive approach to water quality problems. A watershed could be analysed as an entire system, rather than focusing on local effects of single point sources. As tools developed originally in the field of operations, models were used to generate cost effective treatment alternatives (Thomann & Sobel, 1964; Deininger, 1965; Ravelle et al., 1967).

In the 1970s eutrophication was the principal water quality problem addressed. Consequently, more mechanistic representations of biological processes were included leading to the development of detailed nutrient and food chain models (Chen, 1970; Chen & Orlob, 1975; Di Toro et al., 1971; Canale et al., 1974; Canale et al., 1976) incorporating more feedback loops and nonlinear kinetics.

In the 1980s more detailed problems such as the food web and toxic substances were on concern, because they represented an important threat to human and ecosystem health. Solid matter in the transport and fate of toxicants were among the major modelling advances in this period (Thomann & Di Toro, 1983; Chapra & Reckhow, 1983; O'Connor, 1988).

Following the needs and developments in the world, many water quality codes became available. Some of them such as WASP 8.32 (United States Environmental Protection Agency, 2020), QUAL2K (Chapra et al., 2012), QUAL2Kw (Pelletier & Chapra, 2008), HEC-RAS 5 (Brunner, 2016), are free of charge, whereas others such as LAKE2K (Chapra & Martin, 2012), CE-QUAL-W2 4.2 (Wells, 2021), AQUATOX (Park & Clough, 2018) and DELWAQ (Deltares, 2020) are open sourced. These models are mainly developed and used for investigating the conventional water pollution and eutrophication problem.

The number of studies on nutrient based water pollution and eutrophication modelling in Turkey is substantially less than those in the United States and in the European Union. Yet, it is known that the first studies started in the 1980s (Artan, 1983). For river water quality modelling studies in Turkey QUAL2E (Barbaros, 1997; İnkayalı, 2001; Küçükballı, 2003; Özbayrak, 2003) and QUAL2K (Baysal, 2014) models have been used usually. QUAL2E/QUAL2K models take the primary producers in the ecosystem into account. Although the models meet the needs in many watershed management studies in terms of the processes that they consider, their non-dynamic nature can cause problems, especially if the natural drainage network in the studied watershed contains streams and stagnant waters. This situation restricts the applicability of these models in Turkey where many reservoirs have been built in recent years. Yüceer (2005) developed a fully dynamic version of QUAL2E model, but the relevant model is not suitable for dynamic simulations in river systems where there are deep and stratified lakes and reservoirs, since all segments are fully mixed reactors. One of the models, WQRRS (Water Quality for River and Reservoir Systems), developed by US Army Corps of Engineers, is suitable for the simulation of deep and stratified lakes. WQRRS (United States Army Corps of Engineers, 1978) model's reservoir module was adapted only for Turkey. The model, which assumes horizontally complete mixing, contains many vertical layers. It has been applied several times in Turkey (Öktem, 1996; Genç, 1998; Üstün, 1998). Hasanoğlu (2015) realized dissolved oxygen simulations of Borabey Reservoir by using the more advanced CE-QUAL-W2 model. Instead of using models developed specifically for rivers and standing waters, general-purpose water quality models (suitable for lakes, rivers and estuaries) are also used in the studies in Turkey. Among these models, WASP was used by Yenilmez (2007) and Ekdal (2008), whereas, AQUATOX was used by Karaaslan (2009) and Karami (2017). The main disadvantage of these models is that they allow very rough definition (completely mixed or two-layered) of stagnant water bodies. This makes the calibration of the relevant models inefficient and integration with optimization models become very difficult. PCLake, which was applied to Eymir Lake by Kuzyaka (2016), is a model suitable for shallow lake ecosystem simulations.

Other than the models described above, although some original modelling infrastructures were produced for streams (Yüceer, 2005) and lakes (Aydın, 1993; Davaslıgil, 1998; Koçal, 2006) in Turkey, development of these infrastructure for general purpose applications has not continued.

The aim of this study is to develop an independent water quality simulation code following the criteria as listed below. The next section of the paper will provide more details about their implementation as a water quality modelling software.

- **Criterion 1:** The model should be compatible with Turkey's condition in both: biogeographical diversity and data availability. Turkey's biogeographical condition necessitates a flexible water quality modelling framework so that it can make the application of multiple water quality kinetics sub-models possible. The model should also be able to cope with irregular data and missing data. It should also be scalable, from simple sub models to more complex ones making the use of simple model input data sets at first and then upgrading to the model inputs for more sophisticated water quality models without repeating the model inputs that define the simple versions of the final model.

- **Criterion 2:** The model should be applicable and operable based on the knowledge in water quality issues and field and laboratory methods as successfully applied in Turkey for decades. The application and operation of water quality models even for simple cases will necessitate a teamwork conducted by a team of field scientists trained in field methods, laboratory infrastructure with trained technicians and modelling experts with broad theoretical knowledge and computer skills.

- **Criterion 3:** The model should be applicable to different waterbodies such as streams and rivers, lakes, reservoirs, estuaries and coastal waters. The transport scheme should be designed considering this criterion. The model should also be aware of the waterbody type of each model box so that different sub-models or assumptions can be programmed for different waterbody types. The hydrodynamic variables (flow rates and turbulent diffusion/longitudinal dispersion if necessary) will be inputs provided by the users by all means necessary.

- **Criterion 4:** The model should contribute to the general knowledge of modelling for the academicians and institutions. This is an important issue, since water quality modelling software for general purposes are already available. However, specific applications of such modelling software with different kinetic sub models are not straightforward and sometimes impossible. In such a case further development of those modelling software may be necessary but can be impossible if the original developers do not have a source code sharing or developmental support policy.

- **Criterion 5:** The model code should be able to incorporate different water quality kinetic sub models and therefore should be easy to study and understand. As stated previously, Turkey's complex biogeography necessitates the application of different water quality kinetic sub models. To develop one infrastructure for making all of the different water quality sub models is a considerably difficult task, if possible at all. Therefore, the general code of the model should be easy to study and to extend.

Method

As discussed previously, the water quality simulation code consists of a transport code that is solving the transport equation by box model discretization. Three water quality sub-models: 1) a general water quality model for management applications targeting conventional pollutants and eutrophication, 2) an advanced water quality model for more detailed management applications and research, and 3) a generic water quality model with an unlimited number of state variables which are implemented as water quality sub-models.

The Main Transport Equation and Water Quality Kinetics Sub Models

The Transport Equation and its Discretization and its Solution

The transport equation is the advection diffusion equation (Equation 1) considering x and y as the coordinates of any point in lateral directions and z as the coordinate in vertical direction.

$$\frac{\partial C}{\partial t} = -u \cdot \frac{\partial C}{\partial x} + D_x \cdot \frac{\partial^2 C}{\partial x^2} - v \cdot \frac{\partial C}{dy} + D_y \cdot \frac{\partial^2 C}{\partial y^2} - w \cdot \frac{\partial C}{dz} + D_z \cdot \frac{\partial^2 C}{\partial z^2} - v_{set} \cdot \frac{\partial C}{dz} + \sum_{h=1}^{\substack{number\ of\\ external\\ sources/\\ sinks}} S_h + \sum_{k=1}^{\substack{number\\ of\\ kinetic\\ processes}} R_k$$

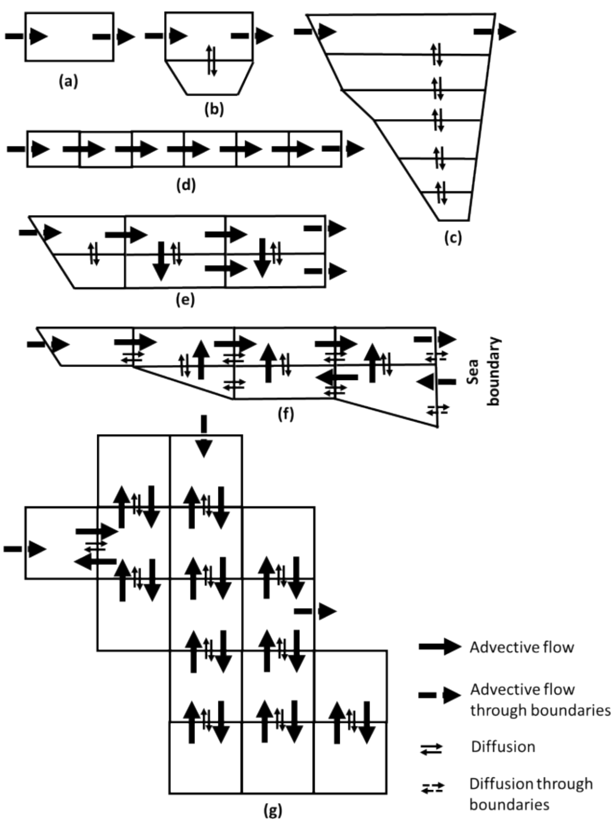
(Equation 1)

Where; C is the concentration of any state variable $[M \cdot T^{-3}]$; t is the time [T]; u, v and z are the velocities in x, y, z directions $[L \cdot T^{-1}]$; v_{set} is the settling velocity $[L \cdot T^{-1}]$; D_x, D_y and D_z are the diffusion coefficients in x, y, z directions $[L^2 \cdot T^{-1}]$; k is the index for the processes related to the particular state variable; R_k is the reaction rate of the process k related to the particular state variable $[L \cdot T^{-3} \cdot T^{-1}]$; h is the index for external mass inflows/withdrawals for a particular state variable; S_h is the inflow/withdrawal rate of the inflow/outflow for a particular state variable $[L \cdot T^{-3} \cdot T^{-1}]$.

As discussed previously in the introduction; the equation was discretized into spatial boxes, where each box was assumed to be a completely mixed reactor (Figure 1), which is dynamic in time but homogeneous in space.

Figure 1

The Spatial Discretization into Boxes



(a) A completely mixed pond/lake (b) A two-layer lake (c) A vertical one-dimensional and horizontally fully mixed lake/reservoir (d) A non-dispersive river (e) Two-layer scheme for a narrow and deep reservoir (f) Two-layer scheme for a narrow and deep estuary (g) Top view of a multi-dimensional general waterbody (Only top boxes shown each of the box seen could have several neighbouring boxes in vertical direction)

As seen in Figure 1, a model domain consists of boxes and the model domain consists of any number of boxes. Boxes can have any shape, and transfer water and through advection and diffusion processes. A box can be connected to any number of other boxes through links. The boxes are vertically grouped into segments. The boxes and segments can have any configuration and shape (Figure 3). The only limitation imposed is that a box cannot be a part of more than one segment.

Figure 2

Boxes and Links

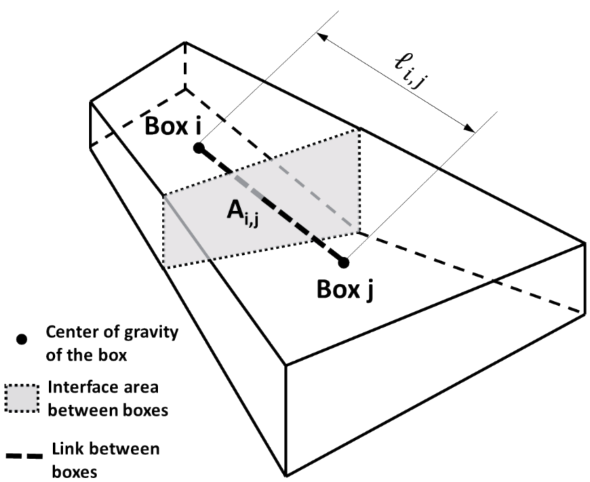
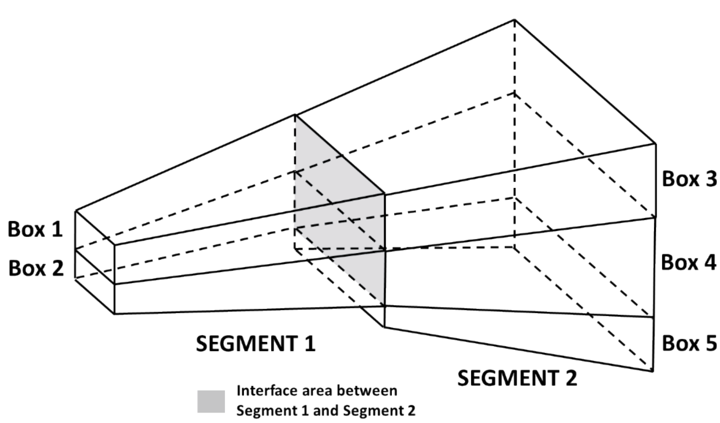


Figure 3

Boxes and Segments



The boxes can be of any shape and the model uses user-entered shape functions for the volume-depth relations of a segment, depth-surface area relations of a segment, depth-interface area relations between two segments and depth-mixing length areas between to segments. Basically, the model calculates the volume of each box and segment (as the sum of related boxes) each time step conducting a water mass balance. The volume is translated to depth and to water surface elevation. This information is then used for each box to calculate the surface area, the interface area and the mixing length between any other neighbouring boxes eliminating the need of user defined time series for these important model inputs. The algorithm conducting these calculations is fairly complex with many intermediate checks and is therefore not given in this paper.

The first step of a modelling study using the described model is to spatially discretize Equation 1 according to box model scheme is to integrate it over the box volumes, box interface areas and box distances. Equation 2 is an example of such an integration considering box i and its link to box j as illustrated in Figure 1.

$$\int_{V_i} \frac{\partial C}{\partial t} \cdot dV_i = \underbrace{\int_{A_{i,j}} \left(-u \cdot \int_{\ell_{x,i,j}} \frac{\partial C}{\partial x} \cdot dx - v \cdot \int_{\ell_{y,i,j}} \frac{\partial C}{\partial y} \cdot dy - w \cdot \int_{\ell_{z,i,j}} \frac{\partial C}{\partial z} \cdot dz \right) \cdot dA_{i,j}}_{ADVECTION} +$$

$$\underbrace{\int_{A_{i,j}} \left(D_x \cdot \int_{\ell_{x,i,j}} \frac{\partial^2 C}{\partial x^2} \cdot dx + D_y \cdot \int_{\ell_{y,i,j}} \frac{\partial^2 C}{\partial y^2} \cdot dy + D_z \cdot \int_{\ell_{z,i,j}} \frac{\partial^2 C}{\partial z^2} \cdot dz \right) \cdot dA_{i,j}}_{DIFFUSION} -$$

$$\underbrace{\int_{A_{i,j}} \left(v_{set} \cdot \int_{\ell_{z,i,j}} \frac{\partial C}{\partial z} \right) \cdot dA_{i,j}}_{SETTLING} + \underbrace{\sum_{h=1}^{\substack{number\ of\\ external\\ sources/sinks}} \int_{V_i} S_h \cdot dV_i}_{EXTERNAL\ SOURCES\ AND\ SINKS} +$$

$$\underbrace{\sum_{k=1}^{\substack{number\ of\\ kinetic\ processes\\ for\ the\ state\\ variable}} \int_{V_i} R_k \cdot dV_i}_{KINETIC\ PROCESSES}$$

(Equation 2)

Evaluating the advection terms

$$\int_{A_{i,j}} \left(-u \cdot \int_{\ell_{x,i,j}} \frac{\partial C}{\partial x} \cdot dx - v \cdot \int_{\ell_{y,i,j}} \frac{\partial C}{\partial y} \cdot dy - w \cdot \int_{\ell_{z,i,j}} \frac{\partial C}{\partial z} \cdot dz \right) \cdot dA_{i,j} =$$

$$-u \cdot \int_{A_{i,j}} \int_{\ell_{x,i,j}} \frac{\partial C}{\partial x^2} \cdot dx \cdot dA_{i,j} - v \cdot \int_{A_{i,j}} \int_{\ell_{y,i,j}} \frac{\partial C}{\partial y^2} \cdot dy \cdot dA_{i,j} -$$

$$w \cdot \int_{A_{i,j}} \int_{\ell_{z,i,j}} \frac{\partial C}{\partial z^2} \cdot dz \cdot dA_{i,j}$$

(Equation 3)

Since

$$u \cdot \int_{A_{i,j}} dA_{i,j} = Q_{x,i,j}; \quad v \cdot \int_{A_{i,j}} dA_{i,j} = Q_{y,i,j}; \quad w \cdot \int_{A_{i,j}} dA_{i,j} = Q_{z,i,j}$$

(Equation 4)

Equation 3 can be rewritten as

$$\int_{A_{i,j}} \left(-u \cdot \int_{\ell_{x,i,j}} \frac{\partial C}{\partial x} \cdot dx - v \cdot \int_{\ell_{y,i,j}} \frac{\partial C}{\partial y} \cdot dy - w \cdot \int_{\ell_{z,i,j}} \frac{\partial C}{\partial z} \cdot dz \right) \cdot dA_{i,j} =$$

$$-Q_{x,i,j} \cdot \int_{\ell_{x,i,j}} \frac{\partial C}{\partial x^2} \cdot dx - Q_{y,i,j} \cdot \int_{\ell_{y,i,j}} \frac{\partial C}{\partial y^2} \cdot dy - Q_{z,i,j} \cdot \int_{\ell_{z,i,j}} \frac{\partial C}{\partial z^2} \cdot dz$$

(Equation 5)

Considering that,

$$\int_{\ell_{x,i,j}} \frac{\partial C}{\partial x^2} \cdot dx = \frac{\partial C}{\partial x} \approx \frac{C_j - C_i}{\ell_{x,i,j}}; \int_{\ell_{y,i,j}} \frac{\partial C}{\partial y^2} \cdot dy = \frac{\partial C}{\partial y} \approx \frac{C_j - C_i}{\ell_{y,i,j}}; \int_{\ell_{z,i,j}} \frac{\partial C}{\partial z^2} \cdot dz = \frac{\partial C}{dz} \approx \frac{C_j - C_i}{\ell_{z,i,j}} \quad and$$

$$Q_{i,j} = Q_{x,i,j} + Q_{y,i,j} + Q_{z,i,j}$$

(Equation 6)

$$\int_{\ell_{x,i,j}} \frac{\partial C}{\partial x} \cdot dx = \int_{\ell_{y,i,j}} \frac{\partial C}{\partial y} \cdot dy = \int_{\ell_{z,i,j}} \frac{\partial C}{\partial z} \cdot dz = \left(C_i - C_j\right)$$

(Equation 7)

Equation 5 can be rewritten as

$$\int_{A_{i,j}} \left(-u \cdot \int_{\ell_{x,i,j}} \frac{\partial C}{\partial x} \cdot dx - v \cdot \int_{\ell_{y,i,j}} \frac{\partial C}{\partial y} \cdot dy - w \cdot \int_{\ell_{z,i,j}} \frac{\partial C}{\partial z} \cdot dz\right) \cdot dA_{i,j} = -Q_{i,j} \cdot \left(C_j - C_i\right)$$

(Equation 8)

Evaluating the diffusion terms

$$\int_{A_{i,j}} \left(D_x \cdot \int_{\ell_{x,i,j}} \frac{\partial^2 C}{\partial x^2} \cdot dx + D_y \cdot \int_{\ell_{y,i,j}} \frac{\partial^2 C}{\partial y^2} \cdot dy + D_z \cdot \int_{\ell_{z,i,j}} \frac{\partial^2 C}{\partial z^2} \cdot dz\right) \cdot dA_{i,j} =$$

$$D_x \cdot \int_{A_{i,j}} \int_{\ell_{x,i,j}} \frac{\partial^2 C}{\partial x^2} \cdot dx \cdot dA_{i,j} + D_y \cdot \int_{A_{i,j}} \int_{\ell_{y,i,j}} \frac{\partial^2 C}{\partial y^2} \cdot dy \cdot dA_{i,j} +$$

$$D_z \cdot \int_{A_{i,j}} \int_{\ell_{z,i,j}} \frac{\partial^2 C}{\partial z^2} \cdot dz \cdot dA_{i,j}$$

(Equation 9)

by considering Equation 6, Equation 7 and Equation 10; Equation 9 can be rewritten as

$$\int_{A_{i,j}} dA_{i,j} = A_{i,j} \quad and \quad \ell_{i,j} = \ell_{x,i,j} + \ell_{y,i,j} + \ell_{z,i,j} \quad and \ D_{i,j} = f\left(D_x, D_y, D_z\right)$$

(Equation 10)

where f is an arbitrary function depending on the size, shape and positions of the neighbouring boxes i and j

$$\int_{A_{i,j}} \left(D_x \cdot \int_{\ell_{x,i,j}} \frac{\partial^2 C}{\partial x^2} \cdot dx + D_y \cdot \int_{\ell_{y,i,j}} \frac{\partial^2 C}{\partial y^2} \cdot dy + D_z \cdot \int_{\ell_{z,i,j}} \frac{\partial^2 C}{\partial z^2} \cdot dz \right) \cdot dA_{i,j} = \frac{A_{i,j} \cdot D_{i,j}}{\ell_{i,j}} \cdot (C_j - C_i)$$

(Equation 11)

Considering the index j as all the neighbouring boxes of i, the advective and diffusive mass flow rate can be written as in Equations 12 and 13 respectively, where M is mass of any state variable. Now, they can be plugged in back to Equation 2 and Equation 14 is obtained.

$$\left(\frac{\partial M}{\partial t}\right)_{ADVECTION} = \sum_{j=1}^{\substack{Number\ of \\ neighbouring \\ boxes\ to\ box\ i}} Q_{j,i} \cdot C_j - \sum_{j=1}^{\substack{Number\ of \\ neighbouring \\ boxes\ to\ box\ i}} Q_{i,j} \cdot C_i$$

(Equation 12)

$$\left(\frac{\partial M}{\partial t}\right)_{DIFFUSION} = \sum_{j=1}^{\substack{Number\ of \\ neighbouring \\ boxes\ to\ box\ i}} \frac{A_{i,j} \cdot D_{i,j}}{\ell_{i,j}} \cdot (C_j - C_i)$$

(Equation 13)

$$\int_{V_i} \frac{\partial C}{dt} \cdot dV_i = \sum_{j=1}^{\substack{Number\ of \\ neighbouring \\ boxes\ to\ box\ i}} Q_{j,i} \cdot C_j - \sum_{j=1}^{\substack{Number\ of \\ neighbouring \\ boxes\ to\ box\ i}} Q_{i,j} \cdot C_i + \sum_{j=1}^{\substack{Number\ of \\ neighbouring \\ boxes\ to\ box\ i}} \frac{A_{i,j} \cdot D_{i,j}}{\ell_{i,j}} \cdot (C_j - C_i)$$

$$\underbrace{\int_{A_{i,j}} \left(v_{set} \cdot \int_{\ell_{z,i,j}} \frac{\partial C}{\partial z} \right) \cdot dA_{i,j}}_{SETTLING} + \underbrace{\sum_{h=1}^{\substack{number\ of \\ external \\ sources/sinks}} \int_{V_i} S_h \cdot dV_i}_{EXTERNAL\ SOURCES\ AND\ SINKS} +$$

$$\underbrace{\sum_{k=1}^{\substack{number\ of \\ kinetic\ processes \\ for\ the\ state \\ variable}} \int_{V_i} R_k \cdot dV_i}_{KINETIC\ PROCESSES}$$

(Equation 14)

Considering that the settling term as an advective inflow from overlaying box (indexed as obox) and outflow to underlying box (indexed as ubox) outflow, and taking all the integrals in Equation 14; Equation 15 is obtained. Dividing both sides of Equation 1 by the volume of box i, the final form of spatially discretized transport equation is obtained.

$$\frac{\partial M_i}{dt} = \sum_{j=1}^{\substack{Number\ of \\ neighbouring \\ boxes\ to\ box\ i}} Q_{j,i} \cdot C_j - \sum_{j=1}^{\substack{Number\ of \\ neighbouring \\ boxes\ to\ box\ i}} Q_{i,j} \cdot C_i + \sum_{j=1}^{\substack{Number\ of \\ neighbouring \\ boxes\ to\ box\ i}} \frac{A_{i,j} \cdot D_{i,j}}{\ell_{i,j}} \cdot (C_j - C_i) +$$

$$v_{set,obox,i} \cdot A_{i,obox} \cdot C_{obox} - v_{set,obox,i} \cdot A_{i,ubox} \cdot C_i +$$

$$\sum_{h=1}^{\substack{number\ of\\ external\\ sources\ and\ sinks\\ sinks}} S_h \cdot V_i + \sum_{k=1}^{\substack{number\ of\\ kinetic\ processes}} R_k \cdot V_i$$

(Equation 15)

$$\frac{\partial C_i}{dt} = \sum_{j=1*}^{\substack{Number\ of\\ neighbouring\\ boxes\ to\ box\ i}} \frac{Q_{j,i}}{V_i} \cdot C_j - \sum_{j=1}^{\substack{Number\ of\\ neighbouring\\ boxes\ to\ box\ i}} \frac{Q_{i,j}}{V_i} \cdot C_i + \sum_{j=1}^{\substack{Number\ of\\ neighbouring\\ boxes\ to\ box\ i}} \frac{A_{i,j} \cdot D_{i,j}}{\ell_{i,j} \cdot V_i} \cdot (C_j - C_i) +$$

$$\frac{v_{set,obox,i} \cdot A_{i,obox}}{V_i} \cdot C_{obox} - \frac{v_{set,obox,i} \cdot A_{i,ubox}}{V_i} \cdot C_i + \sum_{h=1}^{\substack{number\ of\\ external\\ sources\ and\\ sinks}} S_h + \sum_{k=1}^{\substack{number\ of\\ kinetic\ processes\\ for\ the\ state\\ variable}} R_k$$

(Equation 16)

If Equation 16 is written for each model box and state variable a system of equations is obtained. The transport related terms related to different state variables are independent from each other; however, the water quality kinetics related terms couple the entire equation system over the state variables. The system of equations can be written in matrix form as shown in Equation (17).

$$\frac{\partial}{dt}\begin{bmatrix} C \end{bmatrix}_{m\,x\,n} = \left(\left(\begin{bmatrix} Transport \\ Matrix \end{bmatrix}_{m\,x\,m} x \begin{bmatrix} C \end{bmatrix}_{m\,x\,n}\right) + \begin{bmatrix} TEI \end{bmatrix}_{m\,x\,n}\right) + \begin{bmatrix} R \end{bmatrix}_{m\,x\,n}$$

(Equation 17)

where m is the number of boxes, n is the number of state variables, $\begin{bmatrix} C \end{bmatrix}$ is the concentration of each box of each state variable [ML^{-3}], $\begin{bmatrix} Transport \\ Matrix \end{bmatrix}_{m\,x\,m}$ is the transport matrix for each box [T^{-1}], $\begin{bmatrix} TEI \end{bmatrix}_{m\,x\,n}$ is the total external inflows matrix for all boxes and state variables[$ML^{-3}T^{-1}$] and $\begin{bmatrix} R \end{bmatrix}_{m\,x\,n}$ is the kinetic rates for each box and state variable [$ML^{-3}T^{-1}$] handled by one of the water quality kinetics sub models in the following sections. The transport matrix has the following rules that help developing an algorithm to generate it:

- All outflows are located on the diagonal of the transport matrix, where the row and column indexes should be equal to the box number.
- If there is an advective inflow, then the inflow will be located on the receiving boxes row and on the boxes column from which inflow is received.
- A diffusive mass transfer is considered as two advective inflows one from the relevant box to its neighbour as an outflow and one from the neighbour of the relevant box as an inflow. Since both boxes in this case will get one inflow and one outflow, four elements of the transport matrix will be occupied. Considering i as the relevant box and j as its neighbour, matrix elements Row:i, Column:i, Row:j, Column:j; Row:i, Column:j and Row:j, Column:i will be occupied by the diffusive transport term.

The transport equation is handled in the same way for all of the state variables. The kinetic sub models are represented by the kinetics matrix $\left[\ R\ \right]_{m\,x\,n}$ plugged into Equation 17. The contents of the kinetics matrix are given in the next three subsections.

The numerical solution of Equation 17 is straightforward using the simple Euler scheme as shown in Equation 18,

$$\left[\ C^{t+\Delta t}\ \right]_{m\,x\,n} = \left[\ C^{t}\ \right]_{m\,x\,n} + \left(\left(\left(\begin{bmatrix} Transport^{t} \\ Matrix \end{bmatrix}_{m\,x\,m} x \left[\ C^{t}\ \right]_{m\,x\,n}\right) + \left[\ TEI^{t}\ \right]_{m\,x\,n}\right) + \left[\ R^{t}\ \right]_{m\,x\,n}\right) \cdot \Delta t$$

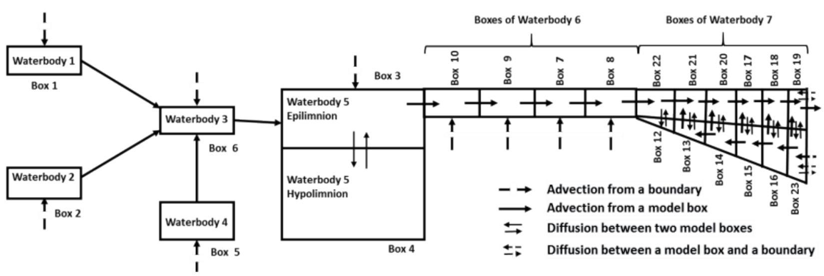
(Equation 18)

where the superscript $t + \Delta t$ represents the next time step and t represents the present time step.

The box model discretization approach provides great flexibility when building a model. Since boxes can be of any shape and configuration, all the waterbodies of an entire watershed could be incorporated into a single model domain (Figure 4).

Figure 4

An Example Model Domain Incorporating Several Waterbodies



In such a configuration, some of the segments (with one or more vertical boxes) could correspond to one waterbody, whereas other segments with their boxes could be assigned into single waterbodies that need to be simulated in more detail. As seen in this Figure, a box could be considered as important as an entire waterbody (such as the boxes 1 to 4 in Figure 4), or a relatively insignificant part of a waterbody (such as box 12 in Figure 4). To provide this flexibility; defining the box shapes is a complex task and the water quality modelling software developed in this study includes the infrastructure that enable to track:

- The water surface elevation of each box depending on its volume
- The horizontal interface area and mixing length of each box pairs depending on their surface and bottom elevations
- The vertical interface area of each box pairs being same segments

Water Quality Sub Model 1

Water quality kinetics sub model 1 (Figure 4) is a conventional water quality model intends for general water pollution problems and eutrophication oriented but not limited to environmental engineering related water quality studies. The model includes 10 state variables and is configured for 4 different complexity levels (Table 1).

As seen in Table 1, the model is designed to be scalable for managing different water quality problems. The interactions among the state variables are illustrated in Figure 5. State variables salinity and total solids are kinetically non-reactive however they are subjected to transport, where salinity is conservative and total suspended solids are subjected to settling. Salinity is used to calculate the saturation concentration of dissolved oxygen and can be used as a conservative tracer. Total suspended solids are used as a common conventional water quality parameter for water quality classification, can be used as ecological state indicator and is usually measured in most of the monitoring campaigns.

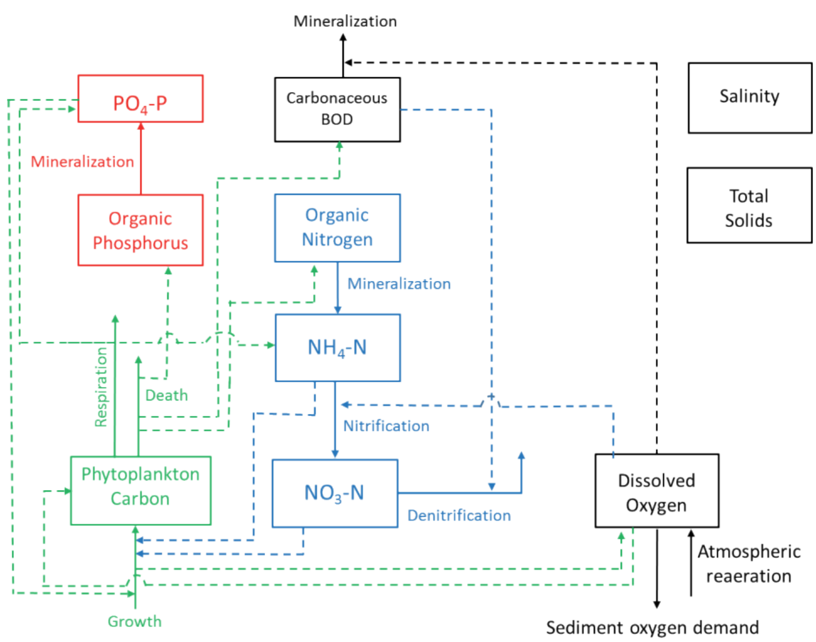
Table 1

State Variables and Complexity Levels of Water Quality Kinetics Sub Model 1

State Variable No	State Variable	Representation in Model	Complexity Level 1	Complexity Level 2	Complexity Level 3	Complexity Level 4
1	Salinity	SALT	✓	✓	✓	✓
2	Total Suspended Solids	TSS	✓	✓	✓	✓
3	Carbonaceous BOD	CBOD	✓	✓	✓	✓
4	Dissolved Oxygen	DOXY	✓	✓	✓	✓
5	Non-algal Organic Nitrogen	ORGN		✓	✓	✓
6	Ammonia Nitrogen	NH4N		✓	✓	✓
7	Nitrate Nitrogen	NO3N		✓	✓	✓
8	Non-algal Organic Phosphorus	ORGP			✓	✓
9	Phosphate Phosphorus	PO4P			✓	✓
10	Phytoplankton Carbon	PHYC				✓

Figure 5

Water Quality Model 1 State Variable Kinetic Interactions



Colour codes: Black: State variables active in model complexity 1, Blue: Additional state variables for model complexity 2, Red: Additional state variables for model complexity 3, Green: Additional state variables for model complexity 4.

The kinetic rates of model state variables are given in the Equations (19-26). Salinity is considered to be conservative and total suspended solids undergo settling that is handled during the solution of the main transport equation only.

$$\frac{dCBOD}{dt} = \underbrace{\left(a_{O2:C,PHYTO} \cdot R_{PHYTO,DEATH}\right)}_{\substack{Gain\ of\ CBOD\ due\ to\ death\ and \\ respiration\ of\ phytoplankton. \\ Only\ active\ if\ complexity\ level>3}} - \underbrace{R_{CBOD,MINER}}_{\substack{Loss\ of\ CBOD\ due\ to \\ aerobic\ mineralization \\ of\ organic\ matter}} - \underbrace{\left(\frac{5}{4} \cdot \frac{32}{14} \cdot R_{DENIT}\right)}_{\substack{Loss\ of\ CBOD\ due \\ to\ denitrification.\ Only \\ active\ if\ complexity\ level>2}}$$

(Equation 19)

$$\frac{dDOXY}{dt} = \underbrace{R_{REAR}}_{\substack{Gain\ of\ DOXY \\ due\ to\ atmospheric \\ reaeration}} - \underbrace{R_{CBOD,MINER}}_{\substack{Loss\ of\ DOXY\ due\ to \\ aerobic\ mineralization \\ of\ organic\ matter}} - \underbrace{\left(a_{O2:C,PHYTO} \cdot R_{PHYTO,RESP}\right)}_{\substack{Loss\ of\ DOXY\ due\ to\ the\ respiration \\ of\ phtoplankton.\ Only\ active\ if \\ complexity\ level > 3}} +$$

$$\underbrace{\left(a_{o2:C,PHYTO} \cdot pref_{NH4N} \cdot R_{PHYTO,GROWTH}\right)}_{\substack{Generation\ of\ DOXY\ due\ to\ primary\ production\ where \\ NH4N\ was\ uptaken\ as\ DIN.\ Only\ active\ if \\ complexity\ level>3}} +$$

$$\underbrace{\left((1 - pref_{NH4N}) \cdot R_{PHYTO,GROWTH} \cdot \left(a_{o2:C,PHYTO} + \frac{48}{14} \cdot a_{N:C,PHYTO}\right)\right)}_{\substack{Generation\ of\ DOXY\ due\ to\ primary\ production\ where\ NO3N\ was\ uptaken\ as\ DIN, \\ considering\ the\ effect\ of\ three\ oxygen\ atoms\ in\ nitrate.\ Only\ active\ if \\ complexity\ level > 3.}} -$$

$$\underbrace{\frac{SOD}{H}}_{\substack{Loss\ of\ DOXY \\ due\ to\ SOD}} - \underbrace{\frac{64}{14} \cdot R_{NITR}}_{\substack{Loss\ of\ DOXY\ due\ to\ nitrificaiton. \\ Only\ active\ if\ complexity\ level>1}}$$

(Equation 20)

$$\frac{dORGN}{dt} = \underbrace{\left(f_{ON} \cdot a_{N:C,PHYTO} \cdot R_{PHYTO,DEATH}\right)}_{\substack{Generation\ of\ ORGN\ due\ to\ phytoplankton \\ death\ and\ respiration.\ Only\ active\ if \\ complexity\ level > 3.}} - \underbrace{R_{ORGN,MINER}}_{\substack{Loss\ of\ ORGN\ due\ to\ organic \\ matter\ mineralization}}$$

(Equation 21)

$$\frac{dNH4N}{dt} = \underbrace{\left((1 - f_{ON}) \cdot a_{N:C,PHYTO} \cdot R_{PHYTO,RESP}\right)}_{\substack{Generation\ of\ NH4N\ due\ to\ the \\ release\ by\ phytoplankton.\ Only \\ active\ if\ complexity\ level > 3.}} -$$

$$\underbrace{\left(a_{N:C,PHYTO} \cdot pref_{NH4N} \cdot R_{PHYTO,GROWTH}\right)}_{\substack{Loss\ of\ NH4N\ due\ to\ the\ uptake\ by\ phytoplankton. \\ Only\ active\ if\ complexity\ level > 3.}} +$$

$$\underbrace{R_{ORGN,MINER}}_{\substack{Generation\ of\ NH4N\ due\ to\ organic \\ matter\ (ORGN)\ mineralization}} - \underbrace{R_{NITR}}_{\substack{Loss\ of\ NH4N\ due\ to \\ nitrification}}$$

(Equation 22)

$$\frac{dNO3N}{dt} = \underbrace{R_{NITR}}_{\substack{Generation\ of\ NH4N \\ due\ to\ nitrification}} - \underbrace{\left((1 - pref_{NH4N}) \cdot a_{N:C,PHYTO} \cdot R_{PHYTO,GROWTH}\right)}_{\substack{Loss\ of\ NO3N\ due\ to\ the\ uptake\ by\ phytoplankton. \\ Only\ active\ if\ complexity\ level > 3.}} - \underbrace{R_{DENITR}}_{\substack{Generation\ of\ NO3N \\ due\ to\ denitrification}}$$

(Equation 23)

$$\frac{dORGP}{dt} = \underbrace{\left(f_{OP} \cdot a_{P:C,PHYTO} \cdot R_{PHYTO,DEATH}\right)}_{\substack{Generation\ of\ ORGP\ due\ to\ phytoplankton \\ death\ and\ respiration.\ Only\ active\ if \\ complexity\ level > 3.}} - \underbrace{R_{ORGP,MINER}}_{\substack{Loss\ of\ ORGP\ due\ to\ organic \\ matter\ mineralization}}$$

(Equation 24)

$$\frac{dPO4P}{dt} = \underbrace{\left((1 - f_{OP}) \cdot a_{P:C,PHYTO} \cdot R_{PHYTO,RESP}\right)}_{\substack{Generation\ of\ PO4P\ due\ to\ the \\ release\ by\ phytoplankton.\ Only \\ active\ if\ complexity\ level > 3.}} - \underbrace{a_{P:C,PHYTO} \cdot R_{PHYTO,GROWTH}}_{\substack{Loss\ of\ PO4P\ due\ to\ the\ uptake\ by \\ phytoplankton. \\ Only\ active\ if\ complexity\ level > 3.}} + \underbrace{R_{ORGP,MINER}}_{\substack{Generation\ of\ PO4P\ due\ to\ organic \\ matter\ (ORGP)\ mineralization.}}$$

(Equation 25)

$$\frac{dPHYC}{dt} = \underbrace{R_{PHYTO,GROWTH}}_{\substack{Generation\ of\ PHYC\ due\ to \\ the\ growth}} - \underbrace{R_{PHYTO,DEATH,RESP}}_{\substack{Loss\ of\ PHYC\ due\ to \\ the\ death\ and\ respiration}}$$

(Equation 26)

The process rates and their auxiliary variables were given in Table 2. The dissolved oxygen saturation ($DOXY_{SAT}$) and reaeration rate constant (k_A) formulas and calculation procedures were too long to be placed into the Table 2 and were therefore given in Appendix. Water Quality Model 1 does not distinguish between the dissolved and particulate parts of state variables. Instead, it allows the user to associate each state variable on each box with a settling velocity time series. In case of salinity and dissolved oxygen; if a non-zero settling velocity occurs, the effect of the settling will still be set back to zero as a failsafe. All of the other state variables

Table 2

Process Rates and Auxiliary Variables of Water Quality Kinetics Sub Model 1

Process Rate	Description	Equation and Auxiliary Variables
$R_{CBOD,MINER}$	Mineralization of CBOD	$k_{CBOD,MINER,20} \cdot \theta_{CBOD,MINER}^{(TEMP-20)} \cdot CBOD \cdot \frac{DOXY}{DOXY + k_{HS,MINER,DOXY}}$
R_{REAR}	Atmospheric reaeration	$k_A \cdot (DOXY_{SAT} - DOXY)$
$R_{ORGN,MINER}$	Mineralization of ORGN	$k_{ORGN,MINER,20} \cdot \theta_{ORGN,MINER}^{(TEMP-20)} \cdot ORGN$
R_{NITR}	Nitrification	$k_{NITR,20} \cdot \theta_{NITR}^{(TEMP-20)} \cdot NH4N \cdot \frac{DOXY}{DOXY + k_{HS,NITR,DOXY}}$
R_{DENIT}	Denitrification	$k_{DENITR,20} \cdot \theta_{DENITR}^{(TEMP-20)} \cdot NO3N \cdot \frac{K_{HS,DENITR,DOXY}}{DOXY + K_{HS,DENITR,DOXY}}$
$R_{ORGP,MINER}$	Mineralization of ORGP	$k_{ORGP,MINER,20} \cdot \theta_{ORGP,MINER}^{(TEMP-20)} \cdot ORGP$
$R_{PHYTO,GROWTH}$	Phytoplankton growth	$k_{PHYTO,GROW,20} \cdot \theta_{PHYTO,GROW}^{(TEMP-20)} \cdot lim_{LIGHT} \cdot lim_{NUT} \cdot PHYC$ Light limitation factor $lim_{LIGHT} = \frac{2.718 \cdot f_{DAY}}{k_E \cdot H} \cdot \left(exp\left(-\frac{I_A}{I_S} \cdot exp(-1 \cdot k_E \cdot H)\right) - exp\left(-\frac{I_A}{I_S}\right)\right)$ Light extinction coefficient $k_E = k_{B,E} + (8.8 \cdot 10^{-3} \cdot CHLA) + \left(5.4 \cdot 10^{-2} \cdot CHLA^{2/3}\right)$ Chlorophyll-A $a_{CHLA:C,PHYTO} \cdot PHYC \cdot 1000$ Nutrient limitation factor $lim_{NUT} = min\left(\frac{NH4N + NO3N}{k_{HS,N} + NH4N + NO3N}, \frac{PO4P}{k_{HS,P} + PO4P}\right)$
$R_{PHYTO,DEATH}$	Phytoplankton death	$k_{PHYTO,DEATH,20} \cdot \theta_{PHYTO,DEATH}^{(TEMP-20)} \cdot PHYC$
$R_{PHYTO,RESP}$	Phytoplankton death	$k_{PHYTO,RESP,20} \cdot \theta_{PHYTO,RESP}^{(TEMP-20)} \cdot PHYC$

Table 3

Derived Variables of Water Quality Kinetics Sub Model 1

Derived Variable No	Derived Variable	Representation in Model	Unit	Derivation
1	Conductivity	COND	µS/cm	UNESCO (1983) is used reversely*
2	Dissolved inorganic nitrogen	DIN	g/m^3	NH4N + NO3N
3	Total Organic Nitrogen	TON	g/m^3	For complexity levels 2 and 3 ORGN For complexity level 4 ORGN + (PHYC · $a_{N:C,PHYTO}$)
4	Total Kjeldahl Nitrogen	TKN	g/m^3	NH4N + TON
5	Total Nitrogen	TN	g/m^3	DIN + TON
6	Total Organic Phosphorus	TOP	g/m^3	For complexity level 3 ORGP For complexity level 4 ORGP + (PHYC · $a_{P:C,PHYTO}$)
7	Total Phosphorus	TP	g/m^3	PO4P + TOP
8	Chlorophyll-a	CHLA	µg/L	For complexity level 4 PHYC · $a_{CHLA:C,PHYTO}$ · 1000
9	Ultimate CBOD	CBODU	g/m^3	For complexity levels 1, 2, 3 CBOD For complexity level 4 CBOD + (PHYC · $a_{O2:C,PHYTO}$)
10	5 days Biochemical Oxygen Demand	BOD5	g/m^3	If the bottle BOD decay rate constant is available CBODU · (1 - exp(-$k_{d,bott}$ ·5)) otherwise CBODU · (1 - exp(-k_{dc} ·5))

* The original method is for the calculation of salinity based on conductivity. Future versions may include other options.

were allowed to settle assuming that the related decision was a user initiative. As discussed previously, the water quality kinetics sub model 1 could derive additional variables listed in Table 3 using the state variables and output them.

Water Quality Sub Model 2

Water quality kinetics sub model 2 is a detailed and aquatic science oriented model where building a more realistic model was intended, at the cost of simplicity for scientific studies where a higher complexity and a more detailed representation of the aquatic ecosystem could be desired. The main aim of the water quality kinetics sub model 2 is the analysis of dynamics of eutrophication, an important problem of many developed and developing countries. Unlike the water quality kinetic sub model 1, this kinetics sub model does not have any complexity level and must be fully used without any simplifications. It includes 14 state variables (Table 4).

Table 4

State Variables and Complexity Levels of Water Quality Kinetics Sub Model 2

State Variable No	State Variable	Representation in the Model
1	Phytoplankton Carbon	PHYC
2	Detrital Particulate Organic carbon	FPOC
3	Detrital Particulate Organic Nitrogen	FPON
4	Detrital Particulate Organic Phosphorus	FPOP
5	Dissolved Organic Carbon	DOC
6	Dissolved Organic Nitrogen	DON
7	Dissolved Organic Phosphorus	DOP
8	Ammonium Nitrogen	NH4N
9	Nitrate Nitrogen	NO3N
10	Soluble Reactive Phosphorus	SRP
11	Particulate Inorganic Phosphorus	PIP
12	Dissolved Oxygen	DOXY
13	Inorganic Suspended Solids	ISS
14	Salinity	SALT

The interactions among the state variables were illustrated in Figure 6. The kinetic rates of model state variables were given in the Equations (27-38). Salinity is considered to be conservative and total suspended solids undergo settling that is handled during the solution of the main transport equation only.

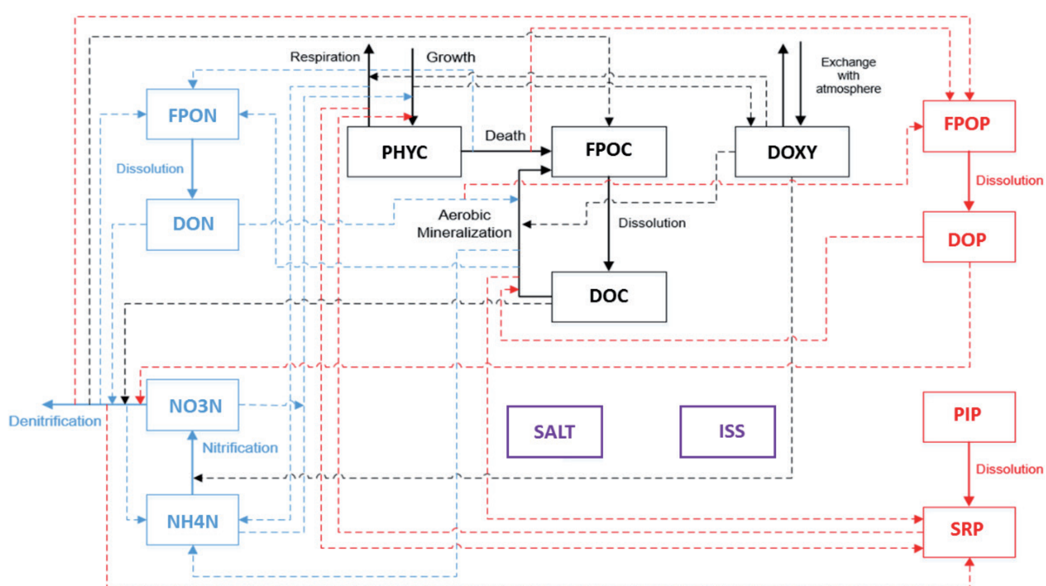
Water Quality Sub Model 2 has a state variable that was not common in water quality models, namely the particulate inorganic phosphorus. This state variable represents the phosphorus that is in inorganic form, however not readily available as soluble reactive phosphorus. It is incorporated to simulate the delayed effects of adsorbed or mineral incorporated phosphorus after dissolution to SRP. Its sources are

- Attached to eroded soil/sediments, especially during wet season. It is known that the erosion rates in Turkey are relatively high
- Resuspension of suspended sediments in stormy weather

As Water Quality Sub Model 2 was further developed, these physical processes would also be realistically incorporated.

Figure 6

Water Quality Model 2 State Variable Kinetic Interactions



Colour codes: Black: Carbon and oxygen related state variables, Blue: Nitrogen related state variables, Red: Phosphorus related state variables, Violet: Other state variables.

$$\frac{dPHYC}{dt} = \underbrace{R_{PHYTO,GROWTH}}_{\substack{Generation\ of\ PHYC\ due\ to \\ the\ growth}} - \underbrace{R_{PHYTO,RESP}}_{\substack{Loss\ of\ PHYC\ due\ to \\ the\ respiration}} - \underbrace{R_{PHYTO,DEATH}}_{\substack{Loss\ of\ PHYC\ due\ to \\ the\ death}}$$

(Equation 27)

$$\frac{dFPOC}{dt} = \underbrace{R_{PHYTO,DEATH}}_{\substack{Generation\ of \\ FPOC\ due\ to \\ the \\ growth \\ of \\ phytoplankton}} + \underbrace{Y_{AERHET} \cdot R_{DOC,MINER}}_{\substack{Generation\ of\ FPOC\ due\ to\ the \\ gain\ of\ the\ aerobic\ heteretroph \\ bacteria\ biomass\ because \\ of\ the\ consumption\ of\ DOC\ as \\ carbon\ source\ during \\ organic\ matter\ mineralization}} + \underbrace{a_{C:N,DENITR} \cdot Y_{DENITR} \cdot R_{DENITR}}_{\substack{Generation\ of\ FPOC\ due\ to\ the \\ gain\ of\ the\ denitrifying \\ bacteria\ biomass\ because \\ of\ the\ consumption\ of\ DOC\ as \\ carbon\ source\ during \\ denitrification}} - \underbrace{R_{FPOC,DISS}}_{\substack{Loss\ of\ FPOC \\ due\ to\ the \\ dissolution\ of \\ particulate \\ organic\ matter}}$$

(Equation 28)

$$\frac{dFPON}{dt} = \underbrace{a_{N:C,PHYTO} \cdot R_{PHYTO,DEATH}}_{\substack{Generation\ of \\ FPON\ due\ to \\ the \\ growth \\ of \\ phytoplankton}} + \underbrace{Y_{AERHET} \cdot R_{DON,MINER}}_{\substack{Generation\ of\ FPON\ due\ to\ the \\ gain\ of\ the\ aerobic\ heteretroph \\ bacteria\ biomass\ because \\ of\ the\ consumption\ of\ DON\ as \\ nitrogen\ source\ during \\ organic\ matter\ mineralization}} + \underbrace{a_{C:N,DENITR} \cdot Y_{DENITR} \cdot R_{DENITR} \cdot \frac{DON}{DOC}}_{\substack{Generation\ of\ FPOC\ due\ to\ the \\ gain\ of\ the\ denitrifying \\ bacteria\ biomass\ because \\ of\ the\ consumption\ of\ DOC\ as \\ carbon\ source\ during \\ denitrification}} - \underbrace{R_{FPON,DISS}}_{\substack{Loss\ of\ FPON \\ due\ to\ the \\ dissolution\ of \\ particulate \\ organic\ matter}}$$

(Equation 29)

$$\frac{dFPOP}{dt} = \underbrace{a_{P:C,PHYTO} \cdot R_{PHYTO,DEATH}}_{\substack{\text{Generation of} \\ \text{FPON due to} \\ \text{the} \\ \text{growth} \\ \text{of} \\ \text{phytoplankton}}} + \underbrace{Y_{AERHET} \cdot R_{DOP,MINER}}_{\substack{\text{Generation of FPOP due to the} \\ \text{gain of the aerobic heteretroph} \\ \text{bacteria biomass because} \\ \text{of the consumption of DOP as} \\ \text{nitrogen source during} \\ \text{organic matter mineralization}}} + \underbrace{a_{C:N,DENITR} \cdot Y_{DENITR} \cdot R_{DENITR} \cdot \frac{DOP}{DOC}}_{\substack{\text{Generation of FPOC due to the} \\ \text{gain of the denitrifying} \\ \text{bacteria biomass because} \\ \text{of the consumption of DOC as} \\ \text{carbon source during} \\ \text{denitrification}}} - \underbrace{R_{FPOP,DISS}}_{\substack{\text{Loss of FPOP} \\ \text{due to the} \\ \text{dissolution of} \\ \text{particulate} \\ \text{organic matter}}}$$

(Equation 30)

$$\frac{dDOC}{dt} = \underbrace{R_{FPOC,DISS}}_{\substack{\text{Generation of DOC} \\ \text{due to the} \\ \text{dissolution of} \\ \text{FPOC}}} - \underbrace{R_{DOC,MINER}}_{\substack{\text{Loss of DOC due to the} \\ \text{organic matter} \\ \text{mineralization} \\ \text{by the aerobic} \\ \text{heterotrophic} \\ \text{bacteria}}} - \underbrace{a_{C:N,DENITR} \cdot R_{DENITR}}_{\substack{\text{Loss of DOC due to the} \\ \text{organic matter} \\ \text{mineralization} \\ \text{by the} \\ \text{denitrifying} \\ \text{bacteria}}}$$

(Equation 31)

$$\frac{dDON}{dt} = \underbrace{R_{FPON,DISS}}_{\substack{\text{Generation of DON} \\ \text{due to the} \\ \text{dissolution of} \\ \text{particulate} \\ \text{FPON}}} - \underbrace{R_{DON,MINER}}_{\substack{\text{Loss of DON due to the} \\ \text{organic matter} \\ \text{mineralization} \\ \text{by the aerobic} \\ \text{heterotrophic} \\ \text{bacteria}}} - \underbrace{a_{C:N,DENITR} \cdot R_{DENITR} \cdot \frac{DON}{DOC}}_{\substack{\text{Loss of DON due to the} \\ \text{organic matter} \\ \text{mineralization} \\ \text{by the} \\ \text{denitrifying} \\ \text{bacteria}}}$$

(Equation 32)

$$\frac{dDOP}{dt} = \underbrace{R_{FPOP,DISS}}_{\substack{\text{Generation of DOP} \\ \text{due to the} \\ \text{dissolution of} \\ \text{particulate} \\ \text{FPOP}}} - \underbrace{R_{DOP,MINER}}_{\substack{\text{Loss of DOP due to the} \\ \text{organic matter} \\ \text{mineralization} \\ \text{by the aerobic} \\ \text{heterotrophic} \\ \text{bacteria}}} - \underbrace{a_{C:N,DENITR} \cdot R_{DENITR} \cdot \frac{DOP}{DOC}}_{\substack{\text{Loss of DOP due to the} \\ \text{organic matter} \\ \text{mineralization} \\ \text{by the} \\ \text{denitrifying} \\ \text{bacteria}}}$$

(Equation 33)

$$\frac{dNH4N}{dt} = \underbrace{a_{N:C,PHYTO} \cdot R_{PHYTO,RESP}}_{\substack{Generation\ of\ NH4N\ due\ to \\ the\ respiration\ of \\ phytoplankton}} - \underbrace{a_{N:C,PHYTO} \cdot pref_{NH4N} \cdot R_{PHYTO,GROWTH}}_{\substack{Loss\ of\ NH4N\ due\ to \\ the\ growth\ of \\ phytoplankton}} - \underbrace{R_{NITR}}_{\substack{Loss\ of\ NH4N\ due\ to \\ nitrification}} + \underbrace{(1 - Y_{AERHET}) \cdot R_{DON,MINER}}_{\substack{Generation\ of\ NH4N\ due\ to\ the \\ internal\ respiration\ of\ the \\ aerobic\ heteretroph \\ bacteria\ biomass\ because \\ of\ the\ consumption\ of\ DON\ as \\ nitrogen\ source\ during \\ organic\ matter\ mineralization}} + \underbrace{a_{C:N,DENITR} \cdot (1 - Y_{DENITR}) \cdot R_{DENITR} \cdot \frac{DON}{DOC}}_{\substack{Generation\ of\ NH4N\ due\ to\ the \\ internal\ respiration\ of\ the\ denitrifying \\ bacteria\ biomass\ because \\ of\ the\ consumption\ of\ DOC\ as \\ nitrogen\ source\ during \\ denitrification}}$$

(Equation 34)

$$\frac{dNO3N}{dt} = \underbrace{R_{NITR}}_{\substack{Generation\ of\ NO3N \\ due\ to\ nitrification}} - \underbrace{a_{N:C,PHYTO} \cdot (1 - pref_{NH4N}) \cdot R_{PHYTO,GROWTH}}_{Loss\ of\ NH4N\ due\ to\ the\ respiration\ of\ phytoplankton} - \underbrace{R_{DENITR}}_{\substack{Loss\ of\ NO3N\ due\ to \\ denitrification}}$$

(Equation 35)

$$\frac{dSRP}{dt} = \underbrace{a_{N:C,PHYTO} \cdot R_{PHYTO,RESP}}_{\substack{Generation\ of\ PO4P\ due\ to \\ the\ respiration\ of \\ phytoplankton}} - \underbrace{a_{P:C,PHYTO} \cdot R_{PHYTO,GROWTH}}_{\substack{Loss\ of\ PO4P\ due\ to \\ the\ growth\ of \\ phytoplankton}} + \underbrace{R_{PIP,DISS}}_{\substack{Generat.on\ of\ PO4P\ due\ to \\ the\ dissolution\ of\ PIP}} + \underbrace{(1 - Y_{AERHET}) \cdot R_{DON,MINER}}_{\substack{Generation\ of\ PO4P\ due\ to\ the \\ internal\ respiration\ of\ the \\ aerobic\ heteretroph \\ bacteria\ biomass\ because \\ of\ the\ consumption\ of\ DOP\ as \\ phosphorus\ source\ during \\ organic\ matter\ mineralization}} + \underbrace{a_{C:N,DENITR} \cdot (1 - Y_{DENITR}) \cdot R_{DENITR} \cdot \frac{DOP}{DOC}}_{\substack{Generation\ of\ PO4P\ due\ to\ the \\ internal\ respiration\ of\ the\ denitrifying \\ bacteria\ biomass\ because \\ of\ the\ consumption\ of\ DOP\ as \\ phosphorus\ source\ during \\ denitrification}}$$

(Equation 36)

$$\frac{dPIP}{dt} = - \underbrace{R_{PIP,DISS}}_{\substack{Loss\ of\ PIP\ due\ to \\ the\ dissolution}}$$

(Equation 37)

$$\frac{dDOXY}{dt} = \underbrace{R_{REAR}}_{\substack{Gain\ of\ DOXY \\ due\ to\ atmospheric \\ reaeration}} - \underbrace{R_{CBOD,MINER}}_{\substack{Loss\ of\ DOXY\ due\ to \\ aerobic\ mineralization \\ of\ organic\ matter}} - \underbrace{a_{O2:C,PHYTO} \cdot R_{PHYTO,RESP}}_{\substack{Loss\ of\ DOXY\ due\ to\ the \\ respiration\ of\ phtoplankton}} + \underbrace{a_{O2:C,PHYTO} \cdot R_{PHYTO,GROWTH}}_{\substack{Generation\ of\ DOXY\ due\ to \\ primary\ production}} - \underbrace{\frac{64}{14} \cdot R_{NITR}}_{\substack{Loss\ of\ DOXY\ due\ to \\ nitrificaiton}}$$

(Equation 38)

The process rates and their auxiliary variables are given in Table 5. The dissolved oxygen saturation ($DOXY_{SAT}$) and reaeration rate constant (k_A) formula and calculation procedures are too long to be placed into Table 5 and are therefore given in Appendix.

Unlike the Water Quality Model 1, Water Quality Model 2 includes state variables that are intended to be of dissolved or particulate form. However, all the state variables, no matter dissolved or particulate are associated to with a settling velocity time series, assuming that the related decision is a user initiative. There is no fail-safe as in Water Quality Model 1. Moreover, Water Quality Model 2 is designed to consider the bacterial loop by assuming the non-algal organic carbon, nitrogen and phosphorus as detritus, the non-living organic matter plus the bacteria associated to them. Hence, the nutrient and organic matter feedback loops are more detailed than the Water Quality Model 1. Additional variables listed in Table 6 can be derived using the state variables in Water Quality Model 2.

Table 5

Process Rates and Auxiliary Variables of Water Quality Kinetics Sub Model 2

Process Rate	Description	Equation and Auxiliary Variables
$R_{DOC,MINER}$	Mineralization of DOC	$k_{DOC,MINER,20} \cdot \theta_{DOC,MINER}^{(TEMP-20)} \cdot DOC \cdot min\left(\left(1 - lim_{O2,DENITR}\right), \frac{DOC}{DOC + k_{HS,DOC,MINER}}\right)$
R_{REAR}	Atmospheric reaeration	$k_A \cdot (DOXY_{SAT} - DOXY)$
$R_{ORGN,MINER}$	Mineralization of DON	$k_{DON,MINER,20} \cdot \theta_{DON,MINER}^{(TEMP-20)} \cdot DON \cdot min\left(\left(1 - lim_{O2,DENITR}\right), \frac{DON}{DON + k_{HS,DON,MINER}}\right)$
R_{NITR}	Nitrification	$k_{NITR,20} \cdot \theta_{NITR}^{(TEMP-20)} \cdot NH4N \cdot \frac{NH4N}{NH4N + k_{HS,NITR,NH4N}} \cdot \frac{DOXY}{DOXY + k_{HS,NITR,DOXY}}$
R_{DENIT}	Denitrification	$k_{DENITR,20} \cdot \theta_{DENITR}^{(TEMP-20)} \cdot NO3N \cdot min\left(lim_{NO3N,DENITR}, min\left(lim_{O2,DENITR}, lim_{DOC,DENITR}\right)\right)$ Nitrate Limitation Factor $lim_{NO3N,DENITR} = \frac{NO3N}{NO3N + k_{HS,DENITR,NO3N}}$ Dissolved Oxygen Limitation (Inhibition) Factor $lim_{O2,DENITR} = \frac{k_{HS,DENITR,DOXY}}{DOXY + k_{HS,DENITR,DOXY}}$ Dissolved Organic Carbon Limitation Factor $lim_{DOC,DENITR} = \frac{DOC}{DOC + k_{HS,DENITR,DOC}}$
$R_{ORGP,MINER}$	Mineralization of DOP	$k_{DOP,MINER,20} \cdot \theta_{DOP,MINER}^{(TEMP-20)} \cdot DOP \cdot min\left(\left(1 - lim_{O2,DENITR}\right), \frac{DOP}{DOP + k_{HS,MINER,DOP}}\right)$

Table 5

(Continued)

Process Rate	Description	Equation and Auxiliary Variables
$R_{PHYTO,GROWTH}$	Phytoplankton growth	$k_{PHYTO,GROW,20} \cdot \theta_{PHYTO,GROW}^{(TEMP-20)} \cdot min(lim_{LIGHT}, lim_{NUT}) \cdot PHYC$ Light limitation factor $lim_{LIGHT} = \frac{2.718 \cdot \mathrm{f_{DAY}}}{\mathrm{k_E \cdot H}} \cdot \left(exp\left(-\frac{I_A}{I_S} \cdot exp(-1 \cdot k_E \cdot H)\right) - exp\left(-\frac{I_A}{I_S}\right)\right)$ Light extinction coefficient $\mathrm{k_E = k_{E,W} + (5.2 \cdot 10^{-2} \cdot ISS) + (1.74 \cdot 10^{-1} \cdot DVSS) + (8.8 \cdot 10^{-3} \cdot CHLA) + (5.4 \cdot 10^{-2} \cdot CHLA^{2/3})}$ Chlorophyll-a $CHLA = a_{CHLA:C,PHYTO} \cdot \mathrm{PHYC} \cdot 1000$ Detrital Volatile Suspended Solids $DVSS = a_{VSS:C,FPOC} \cdot \mathrm{FPOC}$ Nutrient limitation factor $lim_{NUT} = min\left(\frac{[NH4N]+[NO3N]}{k_{HS,N}+[NH4N]+[NO3N]}, \frac{[SRP]}{k_{HS,P}+[SRP]}\right)$
$R_{PHYTO,DEATH}$	Phytoplankton death	$k_{PHYTO,DEATH,20} \cdot \theta_{PHYTO,DEATH}^{(TEMP-20)} \cdot PHYC$
$R_{PHYTO,RESP}$	Phytoplankton respiration	$k_{PHYTO,RESP,20} \cdot \theta_{PHYTO,RESP}^{(TEMP-20)} \cdot PHYC$
$R_{FPOC,DISS}$	Dissolution of non-algal particulate organic carbon	$k_{FPOC,DISS,20} \cdot \theta_{FPOC,DISS}^{(TEMP-20)} \cdot FPOC \cdot \frac{FPOC}{FPOC + k_{HS,DISS,FPOC}}$

Table 5

(Continued)

Process Rate	Description	Equation and Auxiliary Variables
$R_{FPON,DISS}$	Dissolution of non-algal particulate organic nitrogen	$k_{FPON,DISS,20} \cdot \theta_{FPON,DISS}^{(TEMP-20)} \cdot FPON \cdot \frac{FPON}{FPON + k_{HS,DISS,FPON}}$
$R_{FPOP,DISS}$	Dissolution of non-algal particulate organic phosphorus	$k_{FPOP,DISS,20} \cdot \theta_{FPOP,DISS}^{(TEMP-20)} \cdot FPOP \cdot \frac{FPOP}{FPOP + k_{HS,DISS,FPOP}}$
$R_{PIP,DISS}$	Dissolution of non-algal particulate inorganic phosphorus	$k_{PIP,DISS,20} \cdot \theta_{PIP,DISS}^{(TEMP-20)} \cdot PIP \cdot \frac{PIP}{PIP + k_{HS,DISS,PIP}}$

Water Quality Sub Model 3

Water quality kinetics sub model 3 is a generic water quality model considering the decay of multiple and unlimited number of pollutants. Calculations can be made using the desired order kinetics. Interactions of all pollutants with each other and with the by-products formed are not considered. Differential equation for generic pollutants is provided in Equation 39.

$$\frac{dC}{dt} = -k_{20} \cdot \theta_{20}^{(T-20)} \cdot C^n$$

(Equation 39)

where,

C : Generic pollutant concentration [g/m^3]

T : Temperature [°C]

θ : Arrhenius temperature correction factor [-]

k_{20} : Loss rate at 20°C [1/day]

n : Reaction order [-]

Table 6

Derived Variables of Water Quality Kinetics Sub Model 2

Derived Variable No	Derived Variable	Representation in Model	Unit	Derivation
1	Conductivity	COND	µS/cm	UNESCO (1983) is used reversely*
2	Dissolved Inorganic Nitrogen	DIN	g/m^3	NH4N + NO3N
3	Total Organic Nitrogen	TON	g/m^3	FPON + DON + (PHYC · N:C ratio)
4	Total Kjeldahl Nitrogen	TKN	g/m^3	NH4N + TON
5	Total Nitrogen	TN	g/m^3	DIN + TON
6	Total Organic Phosphorus	TOP	g/m^3	FPOP + DOP + (PHYC · P:C ratio)
7	Total Phosphorus	TP	g/m^3	SRP + PIP + TOP
8	Chlorophyll-a	CHLA	µg/L	CHLA
9	Ultimate Carbonaceous BOD	CBODU	g/m^3	FPOC + (DOC · O_2:C ratio) + (PHYC · O_2:C ratio)
10	5 day Biochemical Oxygen Demand	BOD5	g/m^3	If the bottle BOD decay rate constant is available CBODU· $(1 - \exp(-K_{d,bott} \cdot 5))$ otherwise CBODU · $(1 - \exp(-K_{dc} \cdot 5))$
11	Biochemical Oxygen Demand for Dissolved Organic Carbon	DCBOD	g/m^3	DOC · O_2:C ratio
12	Biochemical Oxygen Demand for Particulate Organic Carbon	PCBOD	g/m^3	(PHYC · O_2:C ratio) + (DOC · O_2:C ratio)
13	Carbonaceous Biochemical Oxygen Demand (Carbonaceous BOD representing non-living organic matter)	CBOD	g/m^3	(FPOC + DOC) · (O_2:C_{DOC} ratio)
14	Suspended Solids	TSS	g/m^3	(PHYC · (PHYC:VSS coefficient)) + (FPOC · (FPOC:VSS coefficient)) + ISS
15	Volatile Suspended Solids	VSS	g/m^3	(PHYC · (PHYC:VSS coefficient)) + (FPOC · (FPOC:VSS coefficient))
16	Phosphate Phosphorus	PO4P	g/m^3	SRP + PIP

* The original method is for the calculation of salinity based on conductivity. Future versions may include other options.

Software Development

Programming Language

Since the transport equation is solved using a matrix method, a matrix-oriented programming language was considered as a logical choice. MATLAB (MATrix LABoratory) was chosen for the following reasons:

- It is a high-level language, which is popular and used by many scientists and engineers. A high-level programming language such as MATLAB, Python, R and Julia provides automatic services for the programmer including automatic initialization of variables, preventing memory leaks and automatic garbage collection once the life cycle of a program object is over. High level programming languages usually run on an interpreter sub system and therefore ease the software development process. On the other hand, programmers that choose lower level programming languages (such as Fortran, C, C++, Pascal) must deal with most of these jobs that would have been provided by high level programming languages themselves. The advantage of lower level programming languages is faster code that needs less memory to execute since no interpreter is involved, however at a cost of high and complex software development time and a more complex debugging process. Most high-level language-based environments make use of precompiled libraries (called packages, plugins, etc.) and just in time compilers that precompile the code lines into a virtual machine code and call that machine code when the relevant code section is repeatedly called (such as a time loop in a water quality model) without losing time for interpretation.

- MATLAB code is easy to read and understand. With a few lines of code, it is possible to solve many programming tasks that would result in tens of lines of code using other popular programming languages (such as Fortran, C, C++) in water quality modelling arena.

- MATLAB is an interpreted language; however, it is not as slow as the traditional interpreted programming languages. With a smart way of programming, code that would create a comparable performance to compiled languages can be developed. Software development using an interpreted programming language is more productive than using a compiled programming language, because of easier and faster debugging during the development stage. After the development stage, code could easily be compiled and distributed with MATLAB runtime.

- MATLAB has access to many standard and commonly used file formats (such as spreadsheet files).

- MATLAB has standard library functions that ease many common programming tasks. Development of such utility functions in other programming languages would have taken months of work.

Because MATLAB is a proprietary commercial software and not everybody may obtain a legal copy; Octave (Eaton et.al, 2021) that is highly compatible with MATLAB was considered for parallel development. Most of the water quality code is designed to run in both environments identically. Some issues that would be not compatible in both interpreters are separately written. Therefore, the water quality includes routines that check the interpreter from which it was called so that the water quality simulator is aware of its environments and use different alternatives for specific tasks, which should be conducted differently in MATLAB and Octave.

Input/Output Organization

Since most of the model, inputs are tabular; spreadsheet workbooks are used for input. Spreadsheets provide a comfortable environment for creating model inputs. The model requires xlsx files that are native to Microsoft Excel. For the users, which cannot obtain a legal copy of Microsoft Excel several free spreadsheet software that can also read and write xlsx files (such as Libre Office Calc) exist. Another alternative to assemble model input data sets is to develop a graphical user interface, either a general one for the individual user or a specific one for institutional users and/or institutes with more focused tasks. The model outputs are generated as comma separated value (csv) files for state and derived variables. The model generates three basic outputs:

- Daily outputs for each box

- Diurnal outputs for user specified dates where output is given for each time step in that day

- Spatially and temporally averaged monthly results, which are averaged over one or several box groups prescribed by the user

Modules of the Code

The water quality simulation code consists of the following modules

- The pre-processor and its utilities that assemble the model data structures using the model input datasets.

- The geometry module that conducts the geometrical computations at each time step considering the water budget and the possibly irregular shape of this box.

- The advection-diffusion-reaction module that solves the main transport equation and conducts book-keeping for each box and state variable at each time step

- The water quality kinetic sub model codes that calculate the reaction rates and interact with the advection-diffusion-reaction module

Results

The result of this study is a water quality simulation system that could be applied for a wide range of applications. The model can be used under MATLAB and Octave interpreters and can be distributed as an executable as well.

Under MATLAB, the model can be executed at a satisfying performance, a typical simulation with 30-40 model boxes takes around one minute on a standard pc based laptop. Under Octave, the performance is less satisfactory, 5 to 10 times slower, since Octave is not equipped with a JIT (Just In Time) compiler like MATLAB. However, since Octave is a completely free software, the water quality code can be run or experimented with for free from the software environment point of view. This makes the legal use of the model in low income countries possible.

The water quality simulation software developed in this study has many options to compensate if some of the model inputs are missing. Moreover, it contains three water quality kinetics sub models and can therefore address a wide range of problems in different type of aquatic ecosystems.

Water quality kinetics sub model 1 can be run under four different complexity levels. It can be used for both addressing basic water quality problems and more advanced studies such as planning the program of measures against more complicated water quality and ecology problems such as eutrophication. As the model complexity increases, the data requirements will increase as well. It is also possible to start from complexity level 1 and increase the complexity level as more data become available. Another approach is that any user can start from complexity level 1 and can progress slowly watching out for his or her user mistake and progress into the higher complexity levels once the complexity level being worked on is proven to be user mistake free.

Water quality kinetics sub model 2 is tailored for more complex problems and is research oriented. It is designed to be more realistic than conventional water quality models and is heavily focusing on the investigation of the eutrophication dynamics. Unlike the water quality kinetics sub model 1, it does not have complexity levels. It is structured to be easily expandable for further development in academic studies, so that it could be considered as the "complexity level 1" template of more advanced eutrophication models that would be upcoming for both generally advanced studies or ecosystem specific studies. Future enhancements may include inorganic carbon-alkalinity-pH modelling, simulation of multiple phytoplankton groups, benthic algae and organisms that are on the higher level of the aquatic foodweb (zooplankton, fish, etc.).

Water quality kinetics sub model 3 is intended for initial studies related to a wide range of pollutants except the dissolved gasses, nutrients and semi-natural pollutants such as (organic carbon, BOD and COD). It will be more useful for screening approach in cases where multiple pollutants usually of type synthetic organics, toxics such as pesticides, and hazardous materials in aquatic environment are of concern. Some of those pollutants are of course more complex, so that a generic model would not be suitable to simulate their fate in the aquatic environment thoroughly. However, in watershed management studies especially related to water framework directive related studies, where a need to simulate the behaviour of priority and specific pollutants may arise. In such studies, many pollutants are considered first, but not all of them will be important enough to be modelled. To decide which pollutant should be modelled first, a screening approach simulating all of the potential pollutants is necessary. Then the water quality simulation software developed in this study can be extended with additional subprograms that would include all of the relevant details of case study specific pollutants. Another application of the water quality sub model 3 would be the simulation of several bacterial and viral pathogens in waterbodies since it can simulate the degradation process of any generic pollutant assuming temperature dependence and any order of reaction.

Discussion and Suggestion for Future Work

A software for water quality modelling simulation was developed in Turkey with the collaboration of multiple institutions (universities, state agencies and techno-parks) based on 5 criteria aiming the end product to be compatible and can be integrated with the water quality modelling needs in Turkey.

Considering the Criteria 1 and 5; the model code should be developed in a way that the transport scheme should be general for all water quality variables and several water quality kinetic codes should exist to: (i) serve as templates and (ii) be ready to allow saleable and general applications of the model. In other words, a decoupled model development strategy with a transport code and several water quality codes that should easily plugged into the transport code is encouraged.

Considering Criterion 2, the models state variables or derived variables should be included in the standard variables in water quality analysis. For example, the most of the water quality models use ultimate carbonaceous BOD as state variable, but the most of the laboratories measure BOD_5. Since BOD is used for dissolved oxygen dynamics as a state variable, the model should be able to produce BOD_5 as result. Phytoplankton carbon is another example. Using phytoplankton carbon is more convenient than Chlorophyll-a as a model state variable. However, most of the laboratories have experts to measure Chlorophyll-a, and to measure phytoplankton carbon is very tedious and needs experts that can identify different phytoplankton species, count them and convert each of the species to phytoplankton carbon. However, it is worth to note that some of the already available models such as AQUATOX (Park & Clough, 2018), CE-QUAL-W2 (Wells, 2021) and WASP (United States Environmental Protection Agency, 2020) contain similar derived variables.

Considering the Criterion 3, a box model is the best option for spatial discretization and the transport scheme. The spatial resolution can be easily varied using a box model as well. Each box can represent a waterbody or just a computational element according to the needs that arise in any study.

Criteria 4 and 5 necessitate a readable code written in a high-level language that is easy to study and is supported with standard functions that keeps the programmer away from common programming tasks. The experts that would study and modify the source code can then concentrate on model development and not on the less productive tasks such as developing subroutines that read some data from a spreadsheet file, where the binary format of the spreadsheet file should be parsed to retrieve the data.

The result of this study is the successful implementation of a water quality box model on a high-level programming language styled as an easy-to-read computer code with acceptable performance. The computer code developed is easily extendible and can be run on a computer environment with free software if desired. Its inputs can be generated by using popular spreadsheet applications (Microsoft Excel and Libre Office Calc). Details are given in the results section.

This is the first time for Turkey, that a water quality simulation software was developed at such a level of collaboration and was delivered to the Minister of Agriculture and Forestry of the Republic of Turkey. It was already integrated into the HIDROTURK modelling platform-water quality and ecology module, which is under continuous development and consists of the box model described in this paper (a general transport sub-model, three water quality kinetics sub-models as components), a water ecology module based on Product Unit Neural Networks (PUNN), several utilities for assisting model input generation and a GIS based graphical user interface developed in Python. All of these components of HIDROTURK, including the box model core described in this paper are under continuous development. Additional water quality model cores (such as the one described in this paper) as well as new utilities may be added into the HIDROTURK water quality and ecology module.

To develop a universally accepted and industry style water quality model takes decades and therefore the product of this study should be further developed. This stage of development could be considered as a working prototype and should be further tested for Quality Control and Quality Assurance (QC/QA).

The software was implemented on the high-level programming language -MATLAB, an easy to read and understand coding style- and was also run under Octave, the free software environment. As it was delivered to the Minister of Agriculture and Forestry, it could be released after the QC/QA in several forms; open source that can run both under Octave and MATLAB or as an executable that would run seamlessly without installing any integrated development environment. The software could also be integrated to other water resources management-oriented software development projects such as water allocation models.

Suggestions for the future work are:

- The model developed in this study does not contain any algorithms to calculate the flow fields and turbulent diffusion and accepts user provided time series files for these important inputs. In the future utilities that use the outputs from several hydrodynamic models can be developed. Such type of hydrodynamic linkage will help to speed up the water quality modelling process considerably. Hydrodynamic models can also provide information for the water temperature that can directly be used by the water quality model.

- The only numerical solution algorithm to solve the system of differential equations implemented in the model is the simple Euler method. In the future more advanced numerical solution schemes can be implemented, that the user can trade-off which numerical algorithm so choose.

- The modelling software developed in this study consists of a main model code only. Utilities to support steps of the modelling processes such as the model calibration, sensitivity analysis and uncertainty analysis could be further developed. To support these tools; a database of model constants which can produce probability density functions of each relevant model constant would be useful.

- Development of a suit of post processing tools that calculate several indexes such as water quality classes, eutrophication indexes or ecological processes such as primary production, respiration or nutrient mass balance information would be beneficial as well. For these purposes, the model outputs should be extended to include process-based outputs.

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Appendix A

Dissolved Oxygen Saturation for Water Quality Model 1 and Water Quality Model 2

Dissolved oxygen saturation concentration given below is a function of temperature, salinity and elevation. Here the equation given by American Public Health Association (1992) has been utilized.

$$\mathrm{DOXY_{SAT}} = \left(1 - (0{,}1148 \cdot \mathrm{ELEV})\right) \cdot \exp\left(-139.34411 + \frac{157570.1}{T_K} - \frac{66423080}{T_K^2} + \frac{12438000000}{T_K^3} - \frac{862194900000}{T_K^4} - SALT \cdot \left(0.017674 - \frac{10.754}{T_K} + \frac{2140.7}{T_K^2}\right)\right)$$

(Equation 40)

where,

T_K : Temperature in degrees Kelvin (°K)
SALT : Salinity (ppt)
ELEV : Elevation above the mean sea level (km)

Reaeration Rate Constant for Water Quality Model 1

Reaeration rate constant is a function of average water velocity, depth, wind speed and temperature. In the model, the user can define a single reaeration rate constant, define variable reaeration rate constants depending on location, or instruct the model to calculate the variable reaeration constants depending on flow (KAHYDRA subroutine) or wind (KAWIND subroutine). Bigger reaeration rate constant calculated by flow or by the wind is used by the model.

Reaeration by Hydraulics

Model calculates the reaeration rate that occurs with the effect of flow by Covar Method (Covar, 1976). This method calculates reaeration rate as a function of velocity and depth using formula by Owens (Equation 41), Churchill (Equation 42) or O'Connor Dobbins (Equation 43).

$$k_{qj}(20\,°C) = 5{,}349^{0{,}67}_{vj} \cdot D_j^{-1.85} \quad \text{(Equation 41)}$$

$$k_{qj}(20\,°C) = 5{,}049^{0{,}97}_{vj} \cdot D_j^{-1.67} \quad \text{(Equation 42)}$$

$$k_{qj}(20\,°C) = 3{,}93^{0{,}5}_{vj} \cdot D_j^{-1.50} \quad \text{(Equation 43)}$$

where,

k_{qj} : flow-induced reaeration rate coefficient at 20°C [1/day]

v_j : average water velocity in model box j [m/s]

D_j : Average model box depth [m]

Owens formula is used for model boxes with a depth of less than 60 cm. For model boxes with a depth of more than 60 cm, O'Connor Dobbins or Churchill formula is chosen by taking depth and velocity into consideration. For deep and slow flowing rivers O'Connor Dobbins Formula, while relatively shallow and fast flowing rivers Churchill formula is used. Depending on model box temperature, k_{qj} (20°C) coefficient is corrected by using Equation 44.

$$k_{qj}(T) = k_{qj}(20\,°C) \cdot \theta_a^{T-20} \quad \text{(Equation 44)}$$

where,

T : water temperature [°C]

$k_{qj}(T)$: Reaeration rate constant at model box temperature [day^{-1}]

Θ_a : Temperature correction factor for reaeration rate constant [-]

Reaeration by Wind

Wind-induced reaeration is determined by O'Connor (1983). This method calculates reaeration as a function of wind speed, air and water temperatures and depth using either Equation 45 or Equation 46 or Equation 47.

$$k_{wj} = \frac{86400}{100\,D_j} \cdot \left(\frac{D_{OW}}{v_W}\right)^{2/3} \cdot \frac{\kappa^{1/3}}{\Gamma} \cdot \sqrt{\frac{\rho_a}{\rho_W} \cdot C_d} \cdot 100 \cdot W$$

(Equation 45)

$$k_{wj} = \frac{86400}{100\,D_j} \cdot \left[\frac{1}{\left(\frac{D_{OW}}{v_W}\right)^{2/3} \cdot \frac{\kappa^{1/3}}{\Gamma} \cdot \sqrt{\frac{\rho_a}{\rho_W} \cdot C_d} \cdot 100 \cdot W} + \frac{1}{\sqrt{\frac{D_{OW}}{\kappa\, z_0} \cdot \frac{\rho_a v_a}{\rho_W v_W}} \cdot \sqrt{C_d} \cdot 100 \cdot W}\right]^{-1}$$

(Equation 46)

$$k_{wj} = \frac{86400}{100 \cdot D_j} \cdot \sqrt{\frac{D_{OW}}{\kappa\, z_e} \cdot \frac{\rho_a v_a}{\rho_W v_W}} \cdot \sqrt{C_d} \cdot 100 \cdot W$$

(Equation 47)

where,

k_{wj}	: wind-induced reaeration rate coefficient [1/day]
W	: time-varying wind speed at 10 m above surface [m/s]
T_a	: air temperature [°C]
T	: water temperature [°C]
ρ_a	: density of air as a function of T_a [g/cm^3]
ρ_w	: density of water [1.0 g/cm^3]
v_a	: viscosity of air as a function of T_a [cm^2/s]
v_W	: viscosity of water as a function of T [cm^2/s]
D_{OW}	: diffusivity of oxygen in water as a function of T [cm^2/s]
κ	: von Karman's coefficient, 0.4
v_t	: transitional shear velocity, set to 9, 10, and 10 for small, medium and large scale [cm/s]

v_c : critical shear velocity, set to 22, 11, and 11 for small, medium and large scales [cm/s]

z_e : equivalent roughness, set to 0.25, 0.35, and 0.35 for small, medium and large scales [cm]

z_0 : effective roughness as a function of, z_e, Γ, C_d, v_t, v_a, and W [cm]

λ : inverse of Reynold's number, set to 10, 3, and 3 for small, medium and large scales

Γ : nondimensional coefficient, set to 10, 6.5, and 5 for small, medium and large scales [-]

Γ_u : nondimensional coefficient as function of, Γ, v_c, C_d, and W [-]

C_d : drag coefficient as a function of z_e, Γ, v_a, κ, v_t ve W [-]

The model uses Equation 45 for wind speeds of up to 6 m/s, where interfacial conditions are smooth and viscous forces dominate momentum transfer. Equation 46 is used by the model for wind speed over 20 m/s, where interfacial conditions are rough and momentum transfer is dominated by turbulent eddies. Equation 47 is used for wind speeds between 6 and 20 m/s, and represents a transition zone in which the diffusional sublayer decays and the roughness height increases.

Reaeration Rate Constant for Water Quality Model 2

Reaeration coefficient k_A is defined to the module by the user. If this value is negative, it is estimated by the model, through a subroutine that calculate reaeration rate.

If $k_A < 0$,

$$k_A = k_l / H \qquad \text{(Equation 48)}$$

where

k_A: reaeration rate coefficient [1/day]

k_l: dissolved oxygen interfacial transfer coefficient [m/day]

Reaeration Options for River Type Segment

Option 1 (O'Connor & Dobbins, 1958):

$$k_1 = 3.93 \cdot \frac{U^{0.5}}{H^{1.5}} \cdot H$$ (Equation 49)

Option 2 (Churchill et al., 1962):

$$k_1 = 5.026 \cdot \frac{U}{H^{1.67}} \cdot H$$ (Equation 50)

Option 3 (Owens et al., 1964):

$$k_1 = 5.32 \cdot \frac{U^{0.67}}{H^{1.85}} \cdot H$$ (Equation 51)

Option 4 (Langbein & Drum, 1967):

$$k_1 = 5.13 \cdot \frac{U}{H^{1.33}} \cdot H$$ (Equation 52)

Units given in Equation 49-Equation 52 are as follows: ka (d^{-1}), U(mps), H(m)

Reaeration Options for Lake/Reservoir Type Segment

Option 1 (Broecker et al., 1978):

$$k_1 = 0.864 \cdot U_{W,10}$$ (Equation 53)

Option 2 (Banks, 1975; Banks & Herera, 1977):

$$k_1 = \left(0.728 \cdot \sqrt{U_{W,10}}\right) - \left(0.317 \cdot U_{W,10}\right) + \left(0.0372 \cdot U_{W,10}^2\right)$$ (Equation 54)

Option 3 (Wanninkhof et al., 1991):

$$k_1 = 0.864 \cdot U_{W,10}^{1.64} \qquad \text{(Equation 55)}$$

Reaeration Options for Estuary Type Segment

Homann & Fitzpatrick, 1982

$$k_1 = 5.32 \cdot \frac{U^{0,5}}{H^{1,5}} \cdot H + 0.728 \cdot U^{0,5} - 0.317 \cdot U + 0.0372 \cdot U^2 \qquad \text{(Equation 56)}$$

If $k_a > 0$,

$$k_A = k_A / H \qquad \text{(Equation 57)}$$

where,

$U_{W,10}$: wind speed at 10 m above water surface (m/s)

k_A : Reaeration rate constant (1/day)

H : Depth (m)

Appendix B
Kinetic and Stoichiometric Constants for Water Quality Kinetics Sub Model 1

Appendix B
Model Constants That Can Be Spatially Variable

Model constant	Unit	Relevant State Variable	Description
$k_{NITR,20}$	1/day	NH4N	Nitrification rate at 20°C
θ_{NITR}	-	NH4N	Arrhenius temperature correction factor for nitrification rate constant
$k_{HS,NITR,DOXY}$	mg O_2/L	NH4N	Monod half saturation concentration of dissolved oxygen for nitrification
$k_{DENITR,20}$	1/day	NO3N	Denitrification rate at 20°C
θ_{DENITR}	-	NO3N	Arrhenius temperature correction factor for denitrification rate constant
$K_{HS,DENITR,DOXY}$	mg O_2/L	NO3N	Reversed Monod half saturation concentration of dissolved oxygen for denitrification
$k_{PHYTO,GROW,20}$	1/day	PHYC	Phytoplankton growth rate constant at 20°C
$\theta_{PHYTO,GROW}$	-	PHYC	Arrhenius temperature correction factor for phytoplankton growth rate constant
$k_{HS,N}$	mg N/L	PHYC	Monod half saturation concentration for phytoplankton nitrogen uptake
$k_{HS,P}$	mg P/L	PHYC	Monod half saturation concentration for phytoplankton phosphorus uptake
$k_{PHYTO,DEATH,20}$	1/day	PHYC	Phytoplankton death rate constant at 20°C
$\theta_{PHYTO,DEATH}$	-	PHYC	Arrhenius temperature correction factor for phytoplankton death rate constant
$k_{PHYT,RESP,20}$	1/day	PHYC	Phytoplankton respiration rate constant at 20°C
$\theta_{PHYTO,RESP}$	-	PHYC	Arrhenius temperature correction factor for phytoplankton respiration rate constant
$k_{HS,AMM,PREF,NH4N}$	mg N/L	PHYC	Monod half saturation concentration for ammonia preference of phytoplankton
$k_{CBOD,MINER,20}$	1/day	CBOD	Carbonaceous BOD deoxygenation rate constant at 20°C
$\theta_{CBOD,MINER}$	-	CBOD	Arrhenius temperature correction factor for CBOD deoxygenation rate constant
$k_{HS,MINER,DOXY}$	mg O_2/L	CBOD	O_2 half saturation concentration for CBOD mineralization
k_A	1/day	DOXY	Reaeration rate at 20°C If equal to zero ⇨calculated by model
$k_{ORGN,MINER,20}$	1/day	ORGN	Organic nitrogen mineralization rate at 20°C
$\theta_{ORGN,MINER}$	-	ORGN	Arrhenius temperature correction factor for mineralization rate constant for organic nitrogen
$k_{ORGP,MINER,20}$	1/day	ORGP	Organic phosphorus mineralization rate at 20°C
$\theta_{ORGP,MINER}$	-	ORGP	Arrhenius temperature correction factor for mineralization rate constant for organic phosphorus
$k_{d,bott}$	1/day	*	Bottle BOD decay rate constant

* This constant is related with derived parameter BOD_5.

Appendix C
Kinetic and Stoichiometric Constants for Water Quality Kinetics Sub Model 2

Table C2

Model Constants That Can Be Spatially Variable

Model constant	Unit	Relevant State Variable	Description
$k_{PHYTO,GROW,20}$	1/day	Phytoplankton Carbon	Phytoplankton growth rate constant at 20°C (1/day)
$\theta_{PHYTO,GROW}$	-	Phytoplankton Carbon	Arrhenius temperature correction factor for phytoplankton growth
$k_{PHYTO,RESP,20}$	1/day	Phytoplankton Carbon	Phytoplankton respiration rate constant at 20°C
$\theta_{PHYTO,RESP}$	-	Phytoplankton Carbon	Arrhenius temperature correction factor for phytoplankton respiration
$k_{PHYTO,DEATH,20}$	1/day	Phytoplankton Carbon	Phytoplankton death rate constant at 20°C
$\theta_{PHYTO,DEATH}$	-	Phytoplankton Carbon	Arrhenius temperature correction factor for phytoplankton death
$k_{HS,AMM,PREF,NH4N}$	-	Phytoplankton Carbon	Ammonia preference of phytoplankton as DIN source
$k_{HS,N}$	g N/m^3	Phytoplankton Carbon	Monod half saturation concentration for phytoplankton DIN uptake
$k_{HS,P}$	g P/m^3	Phytoplankton Carbon	Monod half saturation concentration for phytoplankton SRP uptake
I_S	Langley	Phytoplankton Carbon	Saturation light intensity
$k_{FPOC,DISS,20}$	1/day	Particulate Organic Carbon	Particulate organic carbon dissolution rate constant at 20°C
$\theta_{FPOC,DISS}$	-	Particulate Organic Carbon	Arrhenius temperature correction factor for fine particulate organic carbon dissolution
$k_{HS,FPOC,DISS,20}$	gC/m^3	Particulate Organic Carbon	Monod half saturation concentration of fine particulate organic carbon dissolution
$k_{FPON,DISS,20}$	1/day	Particulate Organic Nitrogen	Particulate organic nitrogen dissolution rate constant at 20°C
$\theta_{FPON,DISS}$	-	Particulate Organic Nitrogen	Arrhenius temperature correction factor for fine particulate organic nitrogen dissolution
$k_{HS,PON,DISS}$	g N/m^3	Particulate Organic Nitrogen	Monod half saturation concentration of fine particulate organic nitrogen dissolution
$k_{FPOP,DISS,20}$	1/day	Particulate Organic Phosphorus	Particulate organic phosphorus dissolution rate constant at 20°C

Table C2

Model Constants That Can Be Spatially Variable (continued)

Model constant	Unit	Relevant State Variable	Description
$\theta_{FPOP,DISS}$	-	Particulate Organic Phosphorus	Arrhenius temperature correction factor for fine particulate organic phosphorus dissolution
$k_{HS,POP,DISS}$	g P/m^3	Particulate Organic Phosphorus	Monod half saturation concentration of fine particulate organic phosphorus dissolution
Y_{AERHET}	g C/g C	Dissolved Organic Carbon	Bacterial biomass yield of DOC mineralization
$k_{DOC,MINER,20}$	1/day	Dissolved Organic Carbon	Dissolved organic carbon mineralization rate constant at 20°C
$\theta_{DOC,MINER}$	-	Dissolved Organic Carbon	Arrhenius temperature correction factor for dissolved organic carbon mineralization
$k_{HS,DOC,MINER}$	g C/m^3	Dissolved Organic Carbon	Monod half saturation concentration of dissolved organic carbon mineralization
$k_{DON,MINER,20}$	1/day	Dissolved Organic Nitrogen	Dissolved organic nitrogen mineralization rate constant at 20°C
$\theta_{DON,MINER}$	-	Dissolved Organic Nitrogen	Arrhenius temperature correction factor for dissolved organic nitrogen mineralization
$k_{HS,DON,MINER}$	g N/m^3	Dissolved Organic Nitrogen	Monod half saturation concentration of dissolved organic nitrogen mineralization
$k_{DOP,MINER,20}$	1/day	Dissolved Organic Phosphorus	Dissolved organic phosphorus mineralization rate constant at 20°C
$\theta_{DOP,MINER}$	-	Dissolved Organic Phosphorus	Arrhenius temperature correction factor for dissolved organic phosphorus mineralization
$k_{HS,DOP,MINER}$	g P/m^3	Dissolved Organic Phosphorus	Monod half saturation concentration of dissolved organic phosphorus mineralization
$k_{NITR,20}$	1/day	Ammonium Nitrogen	Nitrification rate constant at 20°C
θ_{NITR}	-	Ammonium Nitrogen	Arrhenius temperature correction factor for nitrification
$k_{HS,NITR,NH4N}$	g N/m^3	Ammonium Nitrogen	Monod half saturation concentration of ammonia nitrogen for nitrification
$k_{HS,NITR,DOXY}$	g O_2/m^3	Ammonium Nitrogen	Monod half saturation concentration of dissolved oxygen for nitrification

Table C2

Model Constants That Can Be Spatially Variable (continued)

Model Constant	Unit	Relevant State Variable	Description
Y_{DENITR}	g C/g N	Nitrate Nitrogen	Bacterial biomass yield of denitrification
$k_{DENITR,20}$	1/day	Nitrate Nitrogen	Denitrification rate constant at 20°C
θ_{DENITR}	-	Nitrate Nitrogen	Arrhenius temperature correction factor for denitrification
$k_{HS,DENITR,NO3N}$	g N/m^3	Nitrate Nitrogen	Monod half saturation concentration of nitrate nitrogen for denitrification
$k_{HS,DENITR,DOXY}$	g O_2/m^3	Nitrate Nitrogen	Monod half saturation concentration of dissolved oxygen for denitrification
$k_{HS,DENITR,DOC}$	g C/m^3	Nitrate Nitrogen	Monod half saturation concentration of dissolved organic carbon for denitrification
$k_{PIP,DISS,20}$	1/day	Particulate Inorganic Phosphorus	Particulate inorganic phosphorus dissolution rate constant at 20°C
$\theta_{PIP,DISS}$	-	Particulate Inorganic Phosphorus	Arrhenius temperature correction factor for particulate inorganic phosphorus dissolution rate constant
$k_{\mathrm{HS,PIP,DISS}}$	g P/m^3	Particulate Inorganic Phosphorus	Monod half saturation concentration of particulate inorganic phosphorus dissolution
k_{ew}	1/m	Phytoplankton Carbon	Background light extinction parameter
k_{A}	1/day	Dissolved Oxygen	Reaeration rate constant

Table C1

Model Constants That Cannot Be Spatially Variable

Model constant	Unit	Relevant State Variable	Description
$a_{O2:C,PHYTO}$	mg O_2/mg C	PHYC	32/12 mg O_2/mg C in phytoplankton
$a_{P:C,PHYTO}$	mg P/mg C	PHYC	P:C ratio in phytoplankton
$a_{N:C,PHYTO}$	mg N/mg C	PHYC	N:C ratio in phytoplankton
$a_{CHLA:C,PHYTO}$	mg C/mg Chl-a	PHYC	Chlorophyll-a to Carbon ratio

Table C3

Model Constants That Cannot Be Spatially Variable

Model constant	Unit	Relevant State Variable	Description
$a_{CHLA:C,PHYTO}$	mg Chl-a/mg C	Phytoplankton Carbon	Stoichiometric Chlorophyll-a to phytoplankton carbon ratio
$a_{N:C,PHYTO}$	mg N/mg C	Phytoplankton Carbon	Stoichiometric nitrogen to phytoplankton carbon ratio
$a_{P:C,PHYTO}$	mg P/mg C	Phytoplankton Carbon	Stoichiometric phosphorus to phytoplankton carbon ratio
$a_{O2:C,PHYTO}$	mg O_2/mg C	Phytoplankton Carbon	Stoichiometric oxygen to phytoplankton carbon ratio
$a_{VSS:C,FPOC}$	mg C/mg C	Non-algal Particulate Organic carbon	Stoichiometric volatile suspended solids to Non-algal Particulate Organic carbon ratio
$a_{O_2:C,\ DOC}$	mg O_2/mg C	Dissolved Organic Carbon	Stoichiometric oxygen to dissolved organic carbon ratio for dissolved organic carbon
$a_{C:N,DENITR}$	mg C/mg N	Nitrate Nitrogen	Stoichiometric carbon to nitrogen ratio for denitrification

Water Quality Model 1 – Expression for Ammonia Preference Factor

$$pref_{NH4N} = \frac{NH4N \cdot NO3N}{\left(k_{HS,AMM,PREF,NH4N} + NH4N\right) \cdot \left(k_{HS,AMM,PREF,NH4N} + NO3N\right)} + \frac{k_{HS,AMM,PREF,NH4N} \cdot NH4N}{\left(NH4N + NO3N\right) \cdot \left(k_{HS,AMM,PREF,NH4N} + NO3N\right)}$$

Water Quality Model 2 – Expression for Ammonia Preference Factor

$$pref_{NH4N} = \frac{NH4N}{k_{HS,AMM,PREF,NH4N} + NH4N}$$

Extended Turkish Abstract
(Genişletilmiş Türkçe Özet)

Matris Cebrine Dayanan Esnek Bir Su Kalitesi Simülatörü

Su ekosistemlerinin sürdürülebilir yönetimi için bütünleşik bir yaklaşım uygulanması gereklidir. Bu nedenle, havza esaslı yönetim, su kalitesinin iyileştirilmesi için giderek daha popüler bir araç haline gelmektedir. Su kalitesi modelleri havza yönetiminin merkezi bir parçasıdır. Su kalitesi tahminleri yapan modeller su kütlesine dışarıdan gelen etkileri değerlendirmek, sistemin işleyişini anlamak, verilerdeki boşlukları doldurmak ve senaryo analizleri yapabilmek açısından değerli araçlardır ancak hem işletme hem de yazılım geliştirme açısından genellikle karmaşık altyapıya sahiptirler. Dünyadaki ihtiyaç ve gelişmeleri takiben, bazıları ücretsiz, bazıları açık kaynak kodlu olmak üzere konvansiyonel su kirliliği ve ötrofikasyon problemini araştırmak için birçok su kalitesi kodu geliştirilmiştir. Türkiye'de besin elementi kökenli su kirliliği ve ötrofikasyon modelleme çalışmalarının sayısı ABD ve Avrupa Birliği ile karşılaştırıldığında fark edilir ölçüde azdır. Bu çalışmada Türkiye'de aşağıda listelenen kriterleri sağlayacak HİDROTÜRK modeline alt modül olarak bağımsız bir su kalitesi simülasyon kodunun geliştirilmesi hedeflenmiştir.

Kriter 1. Model, biyocoğrafik çeşitlilik ve veri kullanılabilirliği açısından Türkiye şartlarına uygun olmalıdır. Birden çok su kalitesi kinetiği alt modelinin uygulanmasını mümkün kılabilmesi için esnek bir su kalitesi modelleme çerçevesi gerektirmektedir. Model ayrıca düzensiz veriler ve eksik verilerle başa çıkabilmelidir. Ayrıca, basit alt modellerden daha karmaşık alt modellere geçiş yapılabilmelidir.

Kriter 2. Model, Türkiye'de yıllardır başarılı bir şekilde uygulanmakta olan su kalitesi konuları ile saha ve laboratuvar yöntemleri hakkındaki bilgilere dayanılarak uygulanabilir ve işletilebilir olmalıdır. Su kalitesi modellerinin basit durumlar için bile uygulanması ve işletilmesi, saha yöntemleri konusunda eğitilmiş sahada çalışan bilim insanları ekibi, eğitimli teknisyenlerle laboratuvar altyapısı ve geniş teorik bilgi ve bilgisayar becerilerine sahip modelleme uzmanları tarafından yürütülen bir ekip çalışmasını gerektirecektir.

Kriter 3. Model, akarsular, göller, rezervuarlar, haliçler ve kıyı suları gibi farklı su kütleleri için kullanılabilmelidir. Modelin taşınım şeması bu kriter dikkate alınarak tasarlanmalıdır.

Kriter 4. Model, akademisyenler ve kurumlar için genel modelleme bilgisine katkıda bulunmalıdır. Genel amaçlı su kalitesi modelleri zaten mevcut olduğundan, bu önemli bir konudur. Bununla birlikte, farklı kinetik alt modellerle spesifik uygulamalar yapmak mümkün olsa da basit değildir.

Kriter 5. Model kodu, farklı su kalitesi kinetik alt modellerini içerebilmeli ve bu nedenle incelenmesi ve anlaşılması kolay olmalıdır. Daha önce belirtildiği gibi, Türkiye'nin karmaşık biyocoğrafyası farklı su kalitesi kinetik alt modellerinin uygulanmasını gerektirmektedir. Tüm farklı su kalitesi alt modellerini yapmak için bir altyapı geliştirmek mümkün olmakla birlikte oldukça zor bir iştir. Bu nedenle, modelin genel kodunun incelenmesi ve genişletilmesi kolay olmalıdır.

Bu çalışmanın amacı, daha büyük bir hidro-ekolojik modelleme çerçevesinin ana su kalitesi simülatörünü geliştirmektir. Su kalitesi sorunları çok çeşitli olduğundan, tüm su kalitesi sorunlarına uygulanabilecek tek bir su kalitesi kinetiği alt modelinin geliştirilmesi imkânsızdır. Bu nedenle; su kalitesi simülatörü yazılım kodu, açık kaynak felsefesini takip ederek geliştirilerek yüksek seviyede (ancak yüksek performanslı) bir programlama dilinde yazılmış ve kod okunabilirliğini artırmak için iyi bir şekilde belgelendirilmiştir. Matris cebrine dayalı olarak geliştirilen temel su kalitesi simülatör yazılımı, genel bir taşınım alt modeli, üç su kalitesi kinetiği alt modeli ve yardımcı programlardan oluşmaktadır.

Taşınım alt modelinde her bir su kütlesi tam karışımlı olduğu varsayılan kontrol hacimlerine ayrılmakta ve kontrol hacimlerinin her biri için diğer kontrol hacimleri ile veya su kütlesinin temasta olduğu coğrafi bileşenler ile su ve madde alışverişlerini dikkate alan zamana göre dinamik su kütle dengeleri kurulmaktadır. Taşınım, adveksiyon ve difüzyon denklemi ile hesaplanmaktadır. Su kütleleri öncelikle yatay yönde segmentlere ayrılmakta, her segment ise düşeyde kutulara ayrılmaktadır. Segmentler birbirleriyle ve sınır koşullarıyla arayüz alanları üzerinden madde alışverişi yapmaktadır. Her kütle dengesinin taşınım dışında tam karışımlı reaktör içindeki madde dönüşümlerini temsil eden kinetik bileşeni de bulunmaktadır. Bu yaklaşım “Kutu Modeli” olarak bilinmektedir. Bu yaklaşımın üstünlüğü kutuların herhangi bir şekil ve büyüklük kısıtı olmaksızın tanımlanabilmelerinin mümkün olmasıdır. Kutuların dizilimlerine göre tek, iki veya üç boyutlu modellerinin oluşturulması mümkündür. Böylece aynı yaklaşımla akarsular (seri olarak dizilmiş tam karışımlı reaktörler olarak), sığ göller (yan yana dizilmiş tam karışımlı reaktörler olarak) ve derin göl ve baraj gölleri (kısmen yan yana, kısmen alt alta dizilmiş tam karışımlı reaktörler olarak) modellenebilmektedir.

Genel su kalitesi kinetiği alt modeli verilerin veya sistem ile ilgili bilgilerin kısıtlı olduğu durumlarda konvansiyonel kirleticiler ve ötrofikasyon problemi için kullanılmak üzere tasarlanmıştır. 10 adet durum değişkeni (Tuzluluk, Toplam Katı Madde, Karbonlu BOİ, Çözünmüş Oksijen, Canlı Olmayan Organik Azot, Amonyum Azotu, Nitrat Azotu, Canlı Olmayan Organik Fosfor, Fosfat Fosforu, Fitoplankton Karbonu) modellenebilmektedir. Model ayrıca 10 adet türetilmiş değişkeni (İletkenlik, Çözünmüş İnorganik Azot, Toplam Organik Azot, Toplam Kjeldahl Azotu, Toplam Azot, Toplam Organik Fosfor, Toplam Fosfor, Klorofil-a, Nihai Karbonlu BOİ, 5 Günlük Biyokimyasal Oksijen İhtiyacı) için sonuç vermektedir. Durum değişkenleri ve türetilmiş değişkenler model tarafından her zaman adımında ve her kutuda hesaplanmaktadır. Model karmaşıklık seviyeleri (1. Seviye, 2. Seviye, 3. Seviye, 4. Seviye) kullanıcı tarafından seçilebilmektedir. Seçilen model karmaşıklığı seviyesine göre modellenen durum değişkeni sayısı farklılık göstermektedir.

Detaylı yönetim uygulamaları için geliştirilen ileri su kalitesi kinetiği alt modelinde 14 adet durum değişkeni (Fitoplankton Karbonu, Partikül Haldeki Organik Karbon, Partikül Haldeki Organik Azot, Partikül Haldeki Organik Fosfor, Çözünmüş Organik Karbon, Çözünmüş Organik Azot, Çözünmüş Organik Fosfor, Amonyum Azotu, Nitrat Azotu, Çözünmüş Reaktif Fosfor, Partikül Haldeki İnorganik Fosfor, Çözünmüş Oksijen, İnorganik Askıda Katı Madde, Tuzluluk) mevcut olup bu değişkenlerin tamamı modellenmektedir. Model ayrıca 16 adet türetilmiş değişkeni [İletkenlik, Çözünmüş İnorganik Azot, Toplam Organik Azot, Toplam Kjeldahl Azotu, Toplam Azot, Toplam Organik Fosfor, Toplam Fosfor, Klorofil-a, Nihai Karbonlu BOİ, 5 Günlük Biyokimyasal Oksijen İhtiyacı, Çözünmüş Organik Karbon Esaslı Biyokimyasal Oksijen İhtiyacı, Partiküler Organik Karbon Esaslı Biyokimyasal Oksijen İhtiyacı, Karbonlu Biyokimyasal Oksijen İhtiyacı (Cansız organik maddeye karşılık gelen karbonlu BOİ), Askıda Katı Madde, Uçucu Askıda Katı Madde, Fosfat Fosforu] için sonuç vermektedir.

Genel kirletici kinetiği alt modelinde istenilen sayıda genel kirletici için istenilen mertebe kinetiği kullanılarak hesaplama yapılabilmektedir. Bütün kirleticilerin birbirleri ile ve oluşan yan ürünler ile etkileşimleri dikkate alınmamaktadır.

Model yazılım, okunması ve anlaşılması kolay bir kodlama stili kullanılarak üst düzey programlama dili MATLAB ile yapılmıştır. Ayrıca Octave ücretsiz yazılım ortamında da çalıştırılabilmektedir.

Sonuç olarak, Türkiye'de çok sayıda enstitünün (üniversiteler, devlet kurumları ve teknoparklar) işbirliği ile bir su kalitesi modelleme simülasyon yazılımı geliştirilmiştir. Türkiye için ilk kez, böyle bir işbirliği düzeyinde bir su kalitesi simülasyon yazılımı geliştirilmiş ve TC Tarım ve Orman Bakanlığı'na teslim edilmiştir. Dünyaca kabul gören ve endüstri tarzı bir su kalitesi modeli geliştirmek on yıllar almaktadır ve bu nedenle bu çalışmanın ürünü daha da geliştirilmeli, Kalite Güvence (QA) & Kalite Kontrol (QC) testlerinden geçirilmelidir.

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Research Article

The Influence of Marina Characteristics on Invasive Non-native Colonization

Marina Özelliklerinin İstilacı Yerli Olmayan Türlerin Kolonileşmesi Üzerindeki Etkisi

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Abstract

The marina properties, such as; salinity range, tidal fluctuations, depth, and ascidian presence influence the distribution of invasive non-natives. Transport vectors, aquaculture and fishery practices also have an important role in the distribution of invasive non-native species during the bioinvasion process. We focused on 10 invasive non-native species of marine invertebrates in the UK. This study explained the distribution and ecology of five invasive non-native species (*Styela clava* (Ascidia), *Didemnum vexillum* (Ascidia), *Caprella mutica* (Crustacea), *Crepidula fornicata* (Gastropoda), *Watersipora subtorquata* (Bryozoa)) and the marina characteristics of five invasive non-native species colonization (*Austrominius modestus* (Arthropoda), *Ciona intestinalis* (Ascidia), *Botrylloides violaceus* (Ascidia), *Tricellaria inopinata* (Bryozoa), *Bugula neritina* (Bryozoa)) in Wales coasts. The study site deployed settlement tiles in the inner and the outer tile as two positions in each marina: An equal number of vertical and horizontal tiles was deployed at each site with half-sampled at two weeks (very early colonization) and at eight weeks (later colonization). The results showed that some variations in marina characteristics affected the distribution of invasive non-native species. Invasive non-native species abundance, diversity and multivariate structure of the assemblages were very high. The colonization of tiles varied between locations at the entrance and within the marina, but not in any way, and the effect of tile orientation was surprisingly low.
Keywords: *distribution, invasive non-native species, marine invertebrates, transport vectors, Wales marinas*

Öz

Marinalarda tuzluluk aralığı, gelgit dalgalanmaları, derinlik, ve ascidan varlığı gibi faktörler istilacı yerli olmayan türlerin dağılımını etkiler. Ayrıca, taşıma vektörleri, su ürünleri yetiştiriciliği ve balıkçılık uygulamaları da tür istilası sürecinde istilacı yerli olmayan türlerin dağılımında önemli bir role sahiptir. Birleşik Krallıktaki 10 istilacı yerli olmayan deniz omurgasız türüne odaklandık. Bu çalışma, Büyük Britanya'daki beş istilacı yerli olmayan türün (*Styela clava* (Ascidia), *Didemnum vexillum* (Ascidia), *Caprella mutica* (Crustacea), *Crepidula fornicata* (Gastropoda), *Watersipora subtorquata*

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(Bryozoa)) dağılımını ve ekolojisini ayrıca, Galler kıyılarında beş istilacı yerli olmayan türün (*Austrominius modestus* (Arthropoda)*, Ciona intestinalis* (Ascidia)*, Botrylloides violaceus* (Ascidia)*, Tricellaria inopinata* (Bryozoa)*, Bugula neritina* (Bryozoa)) kolonileşmesi üzerinde marina karakterlerinin etkilerini açıklamaktadır. Çalışma bölgesindeki her bir marinanın girişi ve dışında belirlenen yüzeylere plakalar yerleştirdi. Örneklemeler iki haftada (çok erken kolonizasyon) ve sekiz haftada (sonraki kolonizasyon) alındı. Elde edilen sonuçlar, marina özelliklerindeki bazı değişikliklerin istilacı yerli olmayan türün dağılımını etkilediğini göstermiştir. Bu türlerin bolluğu, çeşitliliği ve çok değişkenli yapısının oldukça yüksek olduğu ve plakaların kolonizasyonu marina girişi ve içindeki konumlar arasında farklılık gösterdiği, fakat bu değişkenliğin tutarlı bir şekilde olmayıp marinalardaki plaka konumlarının etkisinin de şaşırtıcı derecede düşük olduğu belirlenmiştir.

Anahtar sözcükler: *dağılım, istilacı yerli olmayan türler, deniz omurgasızları, taşıyıcı vektörler, Galler marinaları*

Introduction

The Distribution of Invasive Non-Native Species

The Identification of Invasive Non-Natives and the Invasion Process

Invasive non-native species (INNS) are species that have been introduced to an area outside their previous natural distribution and have an adverse environmental, economic, or social impact on the new location (Corrales et al., 2020).

The stages of the invasion process include an introduction, colonization, and expansion phases. As the process mainly impacts invasive non-native biodiversity, so it is referred to as bioinvasion (Corrales et al., 2020), and bioinvasion is determined according to the features of invasive non-native species and the characteristics of the recipient community according to propagule pressure. The term 'propagule pressure' generally refers to the abundance or quantity of introduced species and the frequency that they arrive in the location (Ros et al., 2013). Invasive non-native species pass through all stages of the invasion process, from moving to a new environment to distribution after their establishment (Powell-Jennings & Callaway, 2018).

The Effects of Invasive Non-Natives on the Distribution of Marine Species

Invasive non-native species pose a significant threat to the world's oceans (Willis et al., 2009) because they may harm native species, through competition, predation or the transmission of viruses (Chan & Briski, 2017). These species can also affect the distribution of other invasive non-native species because of a damaged ecosystem. In this case, an invasive non-native species cannot spread around invasion areas, even though they are still considered invaders (Orlando-Bonaca et al., 2019).

The extreme density of invasive non-natives may have negative impacts on native and aquaculture species such as “cultured mussels” through competition for space and food. For example, *Didemnum vexillum* (Kott, 2002) can replace mussels as the dominant species in fouling communities. This event is a major functional habitat change because mussels provide a year-round substrate for the settlement of other organisms (National Estuarine and Marine Exotic Species Information System [NEMESIS], 2020). However, the ecological and economic impacts of *D. vexillum* on the biology of sea scallops and fishing are yet unknown (Commonwealth Agricultural Bureaux International [CABI], 2020). The solitary ascidian *Styela clava* (Herdman, 1882) is a sessile filter feeder, an abundant species, and often the dominant species fouling organisms in harbours. In English coasts, the growing population of *S. clava* is paralleled by a decrease in *Ciona intestinalis* because it excludes other organisms while also providing a secondary substrate for other fouling organisms (NEMESIS, 2020). *Crepidula fornicata* (Linnaeus, 1758) has been a very successful invader. They affect the growth of other bivalves by competing in a variety of ways as well as altering habitats in European waters (NEMESIS, 2020). For example, attached limpets increase the hydrodynamic stress on mussels; therefore, the mussels shift energy resources to increase the production of byssus threads (NEMESIS, 2020). *Watersipora subtorquata* (d’Orbigny, 1852) can form very large colonies (Global Invasive Species Database [GISD], 2020). Their colonies provide non-toxic points of attachment for other organisms by allowing a diverse fouling community to develop which can adversely affect the speed and efficiency of ships. Their colonies often develop elevated leaf-like folds rising above the substrate and create additional space for colonization by other organisms (NEMESIS, 2020).

The Ecological Features of Invasive Non-Native Species

The Characteristics of Invaders in Relation to Invasion Success

The adaptation of the species can influence their abundance and indicates the success of the invasion if they cannot successfully adapt to the new location due to their characteristics (Osman & Whitlatch, 2007). A growth layer of some invasive non-native species can serve to protect other invasive non-natives from predation, showing that they are opportunistic species (Foster et al., 2016). In addition, the invasion success of an invasive non-native species in experimental fouling communities is dependent on the presence of large amounts of unoccupied space (CABI, 2020). For example, the dense mats of *D. vexillum* are considered as physical barriers because this species influence the geochemical cycling of nutrients/elements and the circulation of dissolved oxygen, contributing to indirect shifts in benthic ecosystems (Zhan et al., 2015).

Invasive Non-Native Species Resistance to Stress Factors

The success of invasion in marine habitats is determined by the interaction between species-specific adaptations and site-specific environmental features (Lenz et al., 2011). Fluctuating habitats (i.e. intertidal habitats) are more likely to have successful invaders, as species growing in these environments are pre-adapted to variations in abiotic variables such as temperature, salinity, light intensity, and oxygen availability. Adaptation should be predestined by tolerating stressful situations during transport and after introduction to a new habitat (Lenz et al., 2011). Communities of invasive non-native species are less stress-tolerant in their area than in their introduced area. This difference can derive from another feature of successful invaders: the capacity to respond rapidly to new challenges (Lenz et al., 2011). As yet, however, it has not been possible to clarify how these factors on marine invasive non-native species impact the primary mechanisms that enable new invasive species to become established (Foster et al., 2016).

The Characteristics of the Recipient Community and Niche Availability

Invasive non-natives occupy significant structural and functional areas within the invaded ecosystems (Ojaveer et al., 2018). According to Zhan et al. (2015), once the ascidian species is established, it often spreads locally and regionally through 'stepping-stone' introductions associated with a variety of human-mediated vectors, including movement. Ascidians such as *S. clava* and *D. vexillum* are long-term dominant resident ascidians in many harbors along the New England coasts, but they do not spread more open coastal sites since communities with low-diversity have fewer competitors for any newly-settled recruits and juvenile life-stages. This indicates that their invasion success has been limited by competitors, e.g. predators (Osman & Whitlatch, 2007).

The increased bio-fouling complexity in habitats can allow for the introduction of additional species because this complexity can offer suitable habitats, food, and sheltered niche areas (Ulman et al., 2019b). However, if there is not enough environmental niche availability for colonization and there is greater biological resistance in the form of predators and competitors, the invasive non-native species establishment, secondary distribution and colonization may not be possible (Afonso et al., 2020).

The Role of Transport Vectors on Invasive Non-Native Species Distribution

Invasions are caused largely by marine traffic, by the fouling species transport on vessel hulls, and through ballast water released into the area (Ulman et al., 2019b).

Aquaculture facilities are also essential vectors for the movements of invasive non-natives. For example, *D. vexillum* was introduced in the Gulf of Maine when Pacific oysters (*Crassostrea gigas*) were transported for aquaculture purposes (Zhan et al., 2015). Recreational traffic is another important vector for these species in bigger locations where human activity is more prevalent (Afonso et al., 2020).

For example, *S.clava* is native to the Northwest Pacific shores of Asia and Russia (NEMESIS, 2020). This species have probably been introduced to Great Britain (GB) from Korea as well as the seaboards of North America, Australia, and New Zealand (GISD, 2020). *D. vexillum* is native to eastern Japan and first recorded in the marina in North Wales in autumn 2008 (GISD, 2020). The specific vectors for introduction are largely unknown, though international shipping, local boat traffic, and transport of aquaculture species are likely sources (CABI, 2020). *C. mutica* is native to the Northwest Pacific. These caprellids have been introduced to the East (Delaware-Newfoundland) and West Coasts (California-Alaska) of North America, Europe (from Spain to Norway and Germany), and New Zealand (NEMESIS, 2020).

C. fornicata is native from Point Escuminac, Canada, and can be along the East Coast of America, down to the Caribbean (GISD, 2020). In the English Channel, *C. fornicata* has spread from east to west. They were transported into Europe along with American oysters (*Crassostrea virginica*) dredged from oyster beds of Atlantic estuaries (CABI, 2020). At the end of the nineteenth century, these invasive non-natives were accidentally introduced in Europe where they found suitable conditions to settle and develop free surfaces of sandy, coarse sediment, an optimal water temperature range, abundant suspended organic matter as food and no predators. These factors allow for rapid growth and reproductive success (CABI, 2020). *W. subtorquata* is an encrusting bryozoan widely distributed around the world. These invasive non-natives have been introduced to the Northeast Pacific, much of the coast of Australia, New Zealand, and the Atlantic coast of France (NEMESIS, 2020). Their native range has not been determined, but they are becoming common in various regions worldwide on cool temperate coasts. *W. subtorquata* was first detected in 2008 in marinas in Plymouth (Devon) and Poole (Dorset), and their transport to GB is likely to have taken place by recreational craft (GISD, 2020). Their appearances in France are related to the culture of *C. gigas* (NEMESIS, 2020).

Hull Fouling and Ballast Water

The risk of invasion differs with the global shipping trend of vessels and the various coastal environments (Jägerbrand et al., 2019). For remote dispersal, the hulls and sea chests of ships represent major vectors, primarily for the transportation of juvenile and/or adult ascidians. In general, ballast water is not understood to be the key vector for remote dispersion, primarily owing to the limited survival period of free-swimming larvae, but the soil, ground and/or internal surfaces within ballast tanks can distribute harbour ascidians (Zhan et al., 2015). Therefore, three decades, the ballast water effect (via commercial shipping) has become widely understood to be a key vector of species introductions to coastal ecosystems. Despite enhanced global attempts to reduce the risk of ballast water-mediated invasions, ballast water remains a potent vector of invasive non-native aquatic species introductions (Darling et al., 2018).

The ballast water is used to stabilise vessels on the sea and contains suspended matter that can establish as sediment within the ballast tanks. The quantity of the sediment load becomes greater if the ballast water is loaded in shallow waters, rivers and estuaries. These physical factors mean that when a species is released into a new area, it does not automatically establish itself as a viable community (Jägerbrand et al., 2019). Also, the sediments must be removed from the ballast tanks, but this process interferes with the ship's activities (Maglić et al., 2019) because of a significant osmotic shock impact owing to the exposure of high salinity ocean water, so some species remain in the ballast water tanks. This process is called Ballast Water Exchange (BWE) and is greater when connected with transit between marine ports (Gray et al., 2007). In addition, recreational shipping has important effects via hull-fouling, whereas aquaculture and fishery practices have a role in culture of species and transfer of material (Ojaveer et al., 2018).

Recreational Boating

While recreational boats may not be a significant vector for the dispersal of invasive non-native species over great distances, they may play an important role in successful introductions on a local scale (Ros et al., 2013). This means of invasive non-native distribution can be successful due to short and relatively slow voyages undertaken by recreational craft (Foster et al., 2016), although recreational vessels are usually considered low-risk vectors for the same reason (Dafforn et al., 2009). They are connected with highly invaded systems as they occupy smaller marinas for invasive non-natives distribution (Foster et al., 2016). These areas allow for the occurrence of introductory vectors such as marine ports (fouling and ballast water), marinas and aquaculture facilities in recreational areas (Afonso et al., 2020). In commercial shipping the expulsion of ship ballast water/sediments and hull fouling occur because ship travel is increasingly faster, leading to an increase in the survival rate of the species in ballast tanks (Ulman et al., 2019b).

Marina Features As Hotspots for Invasive Non-Native Species Distribution

With their sheltered habitat and high propagule pressure, marinas have become 'hotspots' for the distribution of invasive non-natives (Ulman et al., 2019b). Marinas are called as 'secured islands' for underwater invasive non-native species and provide an entry point for non-natives via recreational yachts or through a network of appropriate ecosystems (Ashton, 2006). Marinas have diverse and complicated natural ecosystems, as well as offering a strong point of connectivity among marine systems. These systems have hydrographic boundaries at greater spatial scales than terrestrial ecosystems, and therefore the distribution of marine species is harder to evaluate than in coastal environments (Ros et al., 2013). In this case, the characteristics of marinas can be used to identify the influence of Non-native Invasion Species (NIS) prevalence (Foster et al., 2016).

Non-natural benthic surfaces (known as artificial substrates) such as floating docks, pilings, and boat hulls have been identified as places for the first arrival and establishment of many introduced marine species, as distributed by anthropogenic vectors (Lambert, 2019). For example, in all parts of its native and introduced range, *S. clava* is more frequently reported on anthropogenic structures than natural surfaces (NEMESIS, 2020). Hard-bottom communities, especially within artificial hard substrates such as docks and pilings, allow to the colonization of invasive non-natives within bays and estuaries (Ros et al., 2013). In particular, invasive non-natives in enclosed habitats are correlated with the artificial hard substrate, especially with fouling communities in harbours and marinas (Ulman et al., 2019a).

The number of berths contributes to the higher invasive non-native species quantity as they increase vessel traffic in a marina (Ulman et al., 2019a). Larger marina sizes also impact this distribution through transport vectors, as marina length becomes one of the most effective factors for invasive non-native species distribution in artificial environments over natural ecosystems (Orlando-Bonaca et al., 2019). Additionally, in marinas, the presence of floating pontoons has a significant role because these species has larger distribution in shallower areas than in their deeper equivalents owing to their distance from the seafloor (Ulman et al., 2019a). Pontoon length represents the availability of a hard infrastructure for invasive non-native species colonization, but is also an indirect measurement of the size of marinas (Dafforn et al., 2009).

The primary aim of the study is to research the ecology and distribution of important invasive non-natives in the UK and show how marina characteristics affect the diversity of invasive non-native species in Welsh coasts. Key factors in this study are therefore marina identity; position within marina; orientation of tile; and the period for colonization. This study evaluates each of these factors to analyze how they affect non-native species diversity and abundance. The study aims test the following hypotheses:

- Invasive non-native species diversity increases with marina size;
- Invasive non-native species diversity is greater within enclosed areas of the marina compared to marina entrances;
- Invasive non-native species diversity is greater on horizontally oriented surfaces than vertical.

In this study, as there was a lot of boat traffic in the largest marina, it was determined that the non-native species diversity was the highest in this marina. The effect of position and orientation varies among marinas, but there was no consistency in the difference between inner and outer in the abundance of some invasive non-native species. In addition, as colonisation period increases, the non-natives have more preferred to distribute vertically some enclosed marinas while some of them have preferred to horizontally distribute in some marina entrance.

Materials and Methods

Study Area

This study was conducted on eight marinas in Wales: Pwllheli (Pwll), Holyhead (HH), Burry Port(BP), Deganwy (DG) and Milford Haven (MH), Swansea (SW), Neyland (NY) and Victoria Dock (VD) in August and September 2009 (Figure 1).

Implementation of Design

The study design consists of an ecological field experiment to assess how marina characteristics in eight marinas affect invasive non-native colonization. The study site has deployed settlement tiles at two positions in each marina: the inner tile, which is well within the marina, and the outer tile at the marina entrance. An equal number of vertical and horizontal tiles was deployed at each site with half sampled at two weeks (very early colonization) and half sampled at eight weeks (later colonization) (Figure 2).

Figure 1

Locations of Marinas in Wales Surveyed for Invasive Non-Native Species (DiGSBS250K, 2011)

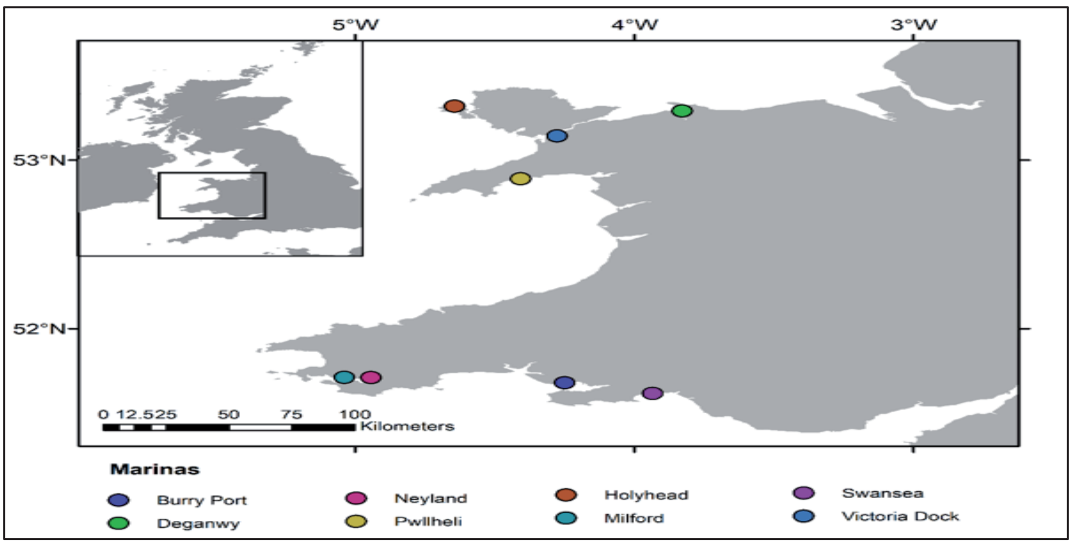


Figure 2

Samplings for Two Weeks and Eight Weeks in Eight Marinas on the Welsh Coasts

Note. N=Panel numbers

In this study, five panels were vertically and horizontally spaced inner and outer of each marina, with 160 samplings taken, totalling 320 samplings across all sites for two weeks and eight weeks colonization. The settlement tiles, which were made from black 'Correx'(approximately 4mm wide) (Figure 3) to provide a vertical and horizontal component, were deployed at a two-meter depth (Figure 4) using a thin nylon cord tied to marina pontoons and weighted with a small fishing weight.

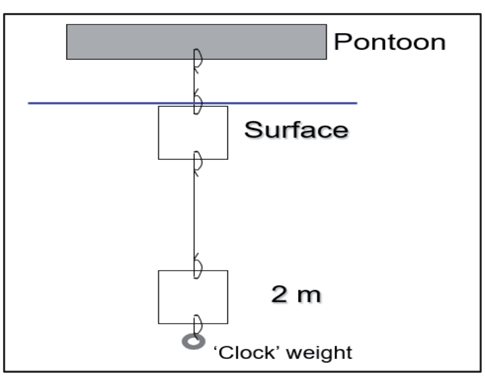
After the tiles were removed, at two weeks and eight weeks, they were scored in the laboratory under a dissecting microscope using a grid to aid the determination of % cover (in colonial organisms) or the number of organisms (for solitary organisms). All recognizable organisms were identified to the lowest possible resolution. After identification, the organisms were characterized as taxa.

Figure 3

Black 'Correx' and Eight-Week Panel Taken in the Survey Area

Figure 4

The Depth Level (Two-Meter) for Both Colonization Period



Data Analysis

The analysis was run separately for two-week data (very early colonization) and eight-week data (later colonization). Firstly, univariate analysis was conducted by using a combination of Excel and R studio to determine the effects of each marina position (inner, outer) and orientation (vertical, horizontal) among marinas on univariate response variables such as community diversity [Simpson's diversity index

(1-ג) and Margalef species richness index (d)], native species and abundance of invasive non-native *Austrominius modestus* (Darwin, 1854), *Ciona intestinalis* (Linnaeus, 1767)*, Botrylloides violaceus* (Oka, 1927)*, Bugula neritina* (Linnaeus, 1758), *Tricellaria inopinata* (D'Hondt & Ambrogi, 1985).

Secondly, raw data (late colonization) was used and three-way mixed-model ANOVA, which is fully factorial, was run to examine an interaction effect between three independent variables [marina, position (inner/outer), and orientation (vertical/horizontal)] on invasive non-native species diversity, richness, and abundance by calculating variance homogeneity. This allows for the application of a crossed model with a marina, position in the marina, and orientation. Cochran's test was used to determine homogeneity (P-value<0.05=heterogeneity or P-value>0.05 = homogeneity).

Finally, a multivariate analysis was conducted. The biological similarity matrix (Bray-Curtis) was calculated using square root-transformed abundance data, and PERMANOVA Analysis was conducted using the same three-way model described above. Following PERMANOVA analysis, multi-dimension scaling (MDS) plots were created to explore significant interactions in each marina in PRİMER.

Results

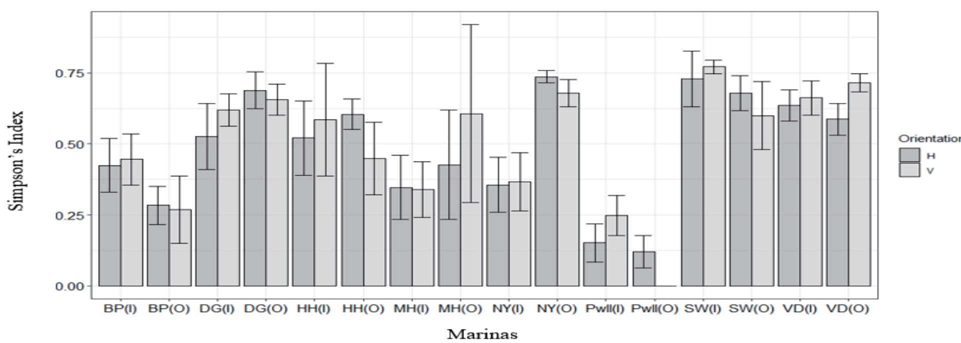
Invasive Non-Native Species Composition

In the survey, there were six species of solitary ascidian, eight species of colonial ascidians, two species of Mollusca, five species of worms, 13 species of Bryozoan, three species of Barnacle, four species of Hydroids, four species of Sponges, one species of Shrimp, one species of Macroalgae, and egg mass for both two weeks and eight weeks colonization.

Diversity indices of benthic communities; Simpson's index (1-ג) and Margalef species richness index (d) in each marina; the results of univariate analysis and three-way mixed factorial model ANOVA have been shown respectively (Figures 5**,** 6 and Table 1**).** According to Cochran's test results, the data had heterogeneous variance (P<0.05). After log transforming, Cochran's test showed no heterogeneity (P>0.05). There was a significant difference in Simpson diversity index among marinas [F (7,128) =17, 34, P-value<0.001], and significant interaction between marina and position [F (7,128) =3.63, P-value<0.05]. This means the effect of position varied among marinas. Investigating further using a Student–Newman–Keuls (SNK) post hoc test, there was no consistency in the difference between inner and outer in Simpson diversity (Table 1).

Figure 5

Simpson Diversity in Eight Marinas for Early Colonization



Note. BP: Burry Port, DG: Deganwy, HH: Holyhead, MH: Milford Haven, NY: Neyland, Pwll: Pwllheli, SW: Swansea, VD: Victoria Dock in two positions (I): Inner or (O): Outer through two orientation tiles H: Horizontal and V: Vertical in the survey area.

Table 1

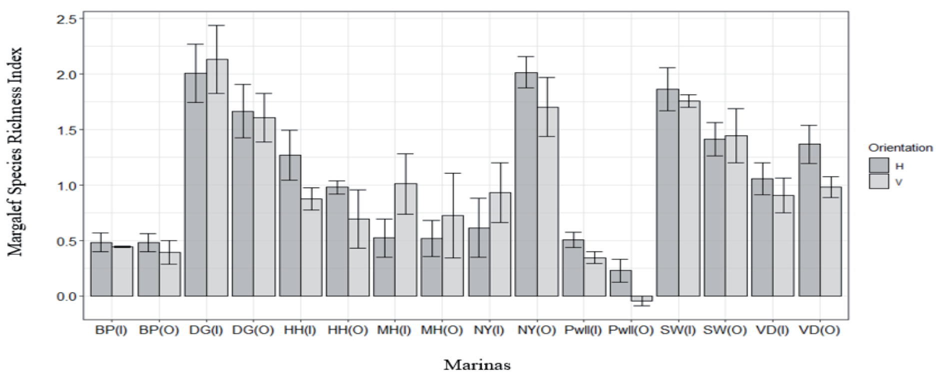
The Results of Three-Way Mixed Model ANOVA Showing Differences in Simpson Diversity among Responsible Variables (Level of Significance P-Value<0.05)

Source	df	Mean Squares (MS)	F-value	P-value
Marina	7	0.3592	17.34	0.0000
Position	1	0.0038	0.05	0.8285
Orientation	1	0.0021	0.35	0.5723
Marina*Position	7	0.0752	3.63	0.0013
Marina*Orientation	7	0.0059	0.28	0.9591
Position*Orientation	1	0.0273	1.93	0.2075
Marina*Position*Orientation	7	0.0142	0.68	0.6853
RES	128	0.0207		
TOTAL	159			

For species richness, according to Cochran's test results, the data had homogeneous variance ($P>0.05$). After log-transforming, the result of Cochran's test did not change. There was a significant difference in species richness among marinas ($F_{(7, 128)} = 32.49$, P-value<0.001), and a significant interaction between marina and position ($F_{(7, 128)} = 7.35$, P-value<0.001). This means the effect of position varies among marinas. Investigating further using an SNK post hoc test, there was no consistency in the difference between inner and outer in species richness (Table 2).

Figure 6

Species Richness in Eight Marinas for Early Colonization



Note. BP: Burry Port, DG: Deganwy, HH: Holyhead, MH: Milford Haven, NY: Neyland, Pwll: Pwllheli, SW: Swansea, VD: Victoria Dock in two positions (I): Inner or (O): Outer through two orientation tiles H: Horizontal and V: Vertical in the survey area.

Table 2

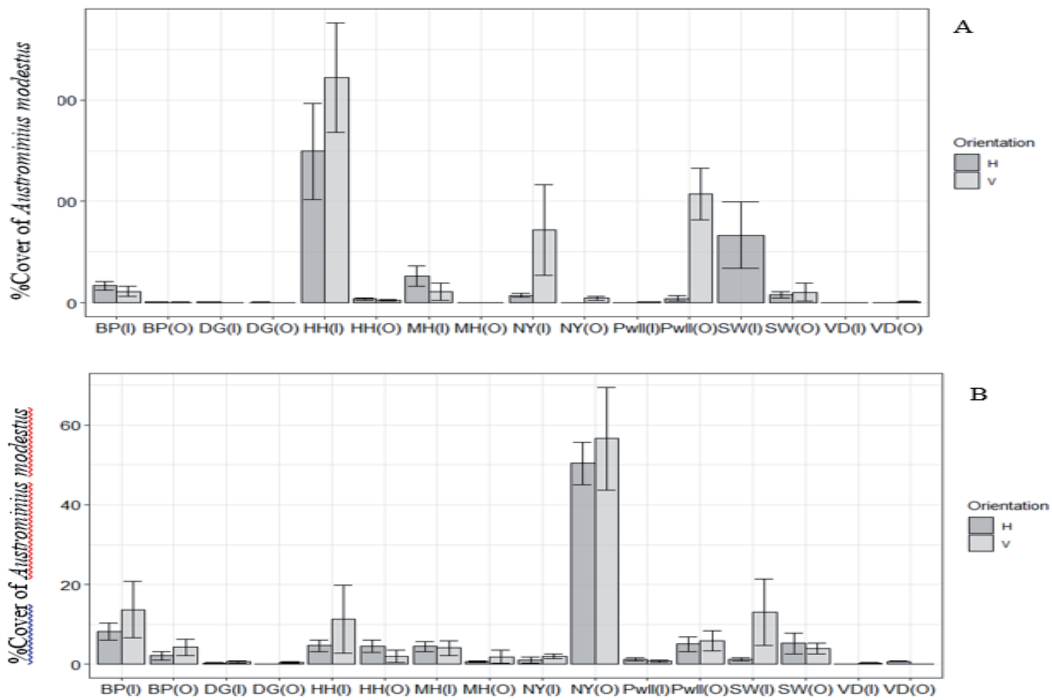
The Results of the Three-Way Mixed Model ANOVA Show Differences in Species Richness among Responsible Variables (Level of Significance P-Value < 0.05)

Source	df	Mean Squares (MS)	F-value	P-value
Marina	7	104.4679	32.49	0.0000
Position	1	0.9000	0.04	0.8508
Orientation	1	3.6000	0.85	0.3876
Marina*Position	7	23.6286	7.35	0.0000
Marina*Orientation	7	4.2429	1.32	0.2462
Position*Orientation	1	4.2250	2.62	0.1494
Marina*Position*Orientation	7	1.6107	0.50	0.8324
RES	128	3.2156		
TOTAL	159			

The univariate test for the selected five discriminating species showed there was a significant difference in the abundance of these invasive non-natives (P-value<0.05) according to marina position in orientation for both very early colonization and later colonization. The very early colonization of *A. modestus* was more vertical within HH marina, whereas there was more later colonization of *A. modestus* in vertical orientation in enclosed NY marina. This means that as colonization period increases, *A. modestus* has more preferred to vertically distribute in enclosed HH marinas (Figures 7A and 7B).

Figures 7A and 7B

The Percentage Cover of Austrominius modestus Abundance Collected from Eight Marinas in Two-Meter Depth



Note. BP: Burry Port, DG: Deganwy, HH: Holyhead, MH: Milford Haven, NY: Neyland, Pwll: Pwllheli, SW: Swansea, VD: Victoria Dock) in two positions (I): Inner or (O): Outer through two orientation tiles H: Horizontal and V: Vertical at the end of two weeks (A) and eight weeks (B).

According to Cochran's test results, the eight-week data had heterogeneous variance ($P<0.05$). After log transforming, Cochran's test showed no heterogeneity ($P>0.05$). This shows a significant difference in the abundance of *A. modestus* among marinas [$F(7,128)=23.50$, P-value<0.001], and a significant interaction between marina and position [$F(7,128)=21.10$, P-value<0.001]. Therefore, the effect of position varies among marinas, but there was no consistency in the difference between inner and outer in the abundance of *A. modestus* (Figure 7B, Table 3).

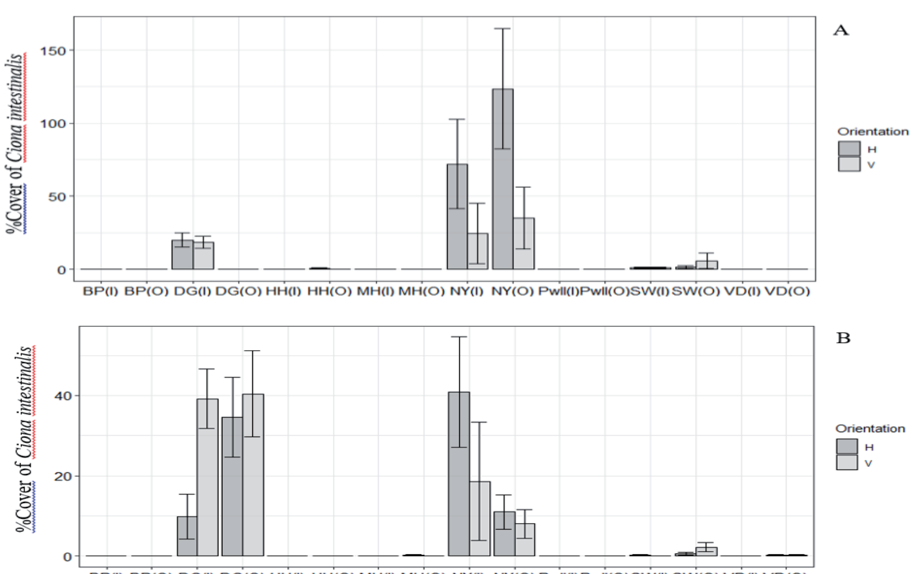
Table 3

The Results of Three-Way Mixed Model ANOVA Show Differences in Austrominius modestus Abundance among Responsible Variables (Level of Significance P-Value < 0.05)

Source	df	Mean Squares (MS)	F-value	P-value
Marina	7	10.7054	23.50	0.0000
Position	1	1.7871	0.19	0.6793
Orientation	1	1.1082	2.50	0.1577
Marina*Position	7	9.6149	21.10	0.0000
Marina*Orientation	7	0.4428	0.97	0.4547
Position*Orientation	1	1.1314	1.69	0.2348
Marina*Position*Orientation	7	0.6695	1.47	0.1838
RES	128	0.4556		
TOTAL	159			

Figures 8A and 8B

The Percentage Cover of Ciona intestinalis Abundance Collected from Eight Marinas in Two-Meter Depth



Note. BP: Burry Port, DG: Deganwy, HH: Holyhead, MH: Milford Haven, NY: Neyland, Pwll: Pwllheli, SW: Swansea, VD: Victoria Dock in two positions (I): Inner or (O): Outer through two orientation tiles H: Horizontal and V: Vertical at the end of two-weeks (A) and eight-weeks (B).

In the very early colonization, *C. intestinalis* has more abundance in horizontal orientation enclosed in the NY marina. The abundance of *C. intestinalis* decreased both within and enclosed NY marina in horizontal orientation in later colonization, while the distribution of *C. intestinalis* increased vertically enclosed in the DG marina in the later colonization. This means that as colonization period increases, these invasive non-natives have more preferred to distribute vertically enclosed in the DG marina (Figures 8A and 8B).

According to Cochran's test results, the eight-week data had heterogeneous variance ($P>0.05$). After log transforming, the result of Cochran's test did not change. This shows a significant difference in the abundance of *C. intestinalis* among marinas [$F(7,128) = 32.30$, P-value<0.001]. There was also a significant interaction between marina and position as well as marina and orientation [$F(7,128) = 2.76$, P-value<0.05], [$F(7,128) = 0.97$, P-value<0.05]. Therefore, the effect of position and orientation varies among marinas. SNK exploration of these interactions shows that there were some differences between positions, but the direction of difference was not consistent (Figure 8B, Table 4).

Table 4

The Results of Three-Way Mixed Model ANOVA Show Differences in Ciona intestinalis Abundance among Responsible Variables (Level of Significance P-Value<0.05)

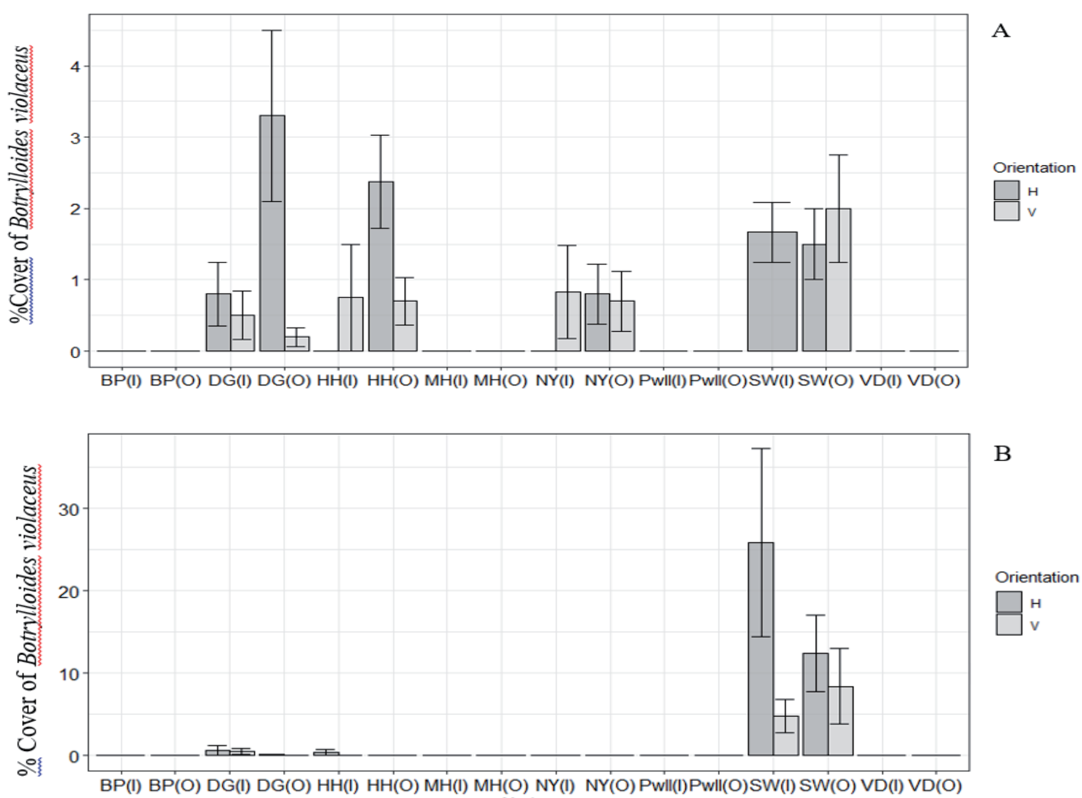
Source	df	Mean Squares (MS)	F-value	P-value
Marina	7	2978.0023	39.30	0.0000
Position	1	4.1924	0.02	0.8819
Orientation	1	1.7060	2.33	0.9306
Marina*Position	7	176.4683	2.76	0.0286
Marina*Orientation	7	209.3304	0.97	0.0105
Position*Orientation	1	6.0000	0.05	0.8312
Marina*Position*Orientation	7	122.5614	1.62	0.1361
RES	128	75.7792		
TOTAL	159			

In the very early colonization, *B. violaceus* has more abundant in enclosed DG marina in horizontal orientation, whereas the abundance of *B. violaceus* increased in horizontal orientation within SW marina in the later colonization period. Thus, as the colonization period increases, *B. violaceus* has preferred to horizontally distribute within SW marina (Figures 9A and 9B).

According to Cochran's test results, the eight-week data had heterogeneous variance ($P>0.05$).After log transforming, the result of Cochran's test did not change. This shows a significant difference in the abundance of *B. violaceus* among marinas [$F_{(7,128)} = 34.79$, P-value<0.001]. There was also a significant interaction between marina and orientation [$F_{(7,128)} = 3.97$, P-value<0.001], meaning the effect of orientation varies among marinas. SNK exploration of this interaction shows that at the only marina (SW) where *B.violaceus* was abundant, there was significantly more *B. violaceus* on horizontal surfaces than vertical (Figure 9B, Table 5).

Figure 9

The Percentage Cover of Botrylloides violaceus Abundance Collected from Eight Marinas in Two-Meter Depth



Note. BP: Burry Port, DG: Deganwy, HH: Holyhead, MH: Milford Haven, NY: Neyland, Pwll: Pwllheli, SW: Swansea, VD: Victoria Dock in two positions (I): Inner or (O): Outer through two orientation tiles H: Horizontal and: Vertical at the end of two weeks (A) and eight weeks (B).

Table 5

The Results of Three-Way Mixed Model ANOVA Show Differences in Botrylloides violaceus Abundance among Responsible Variables (Level of Significance P-Value<0.05)

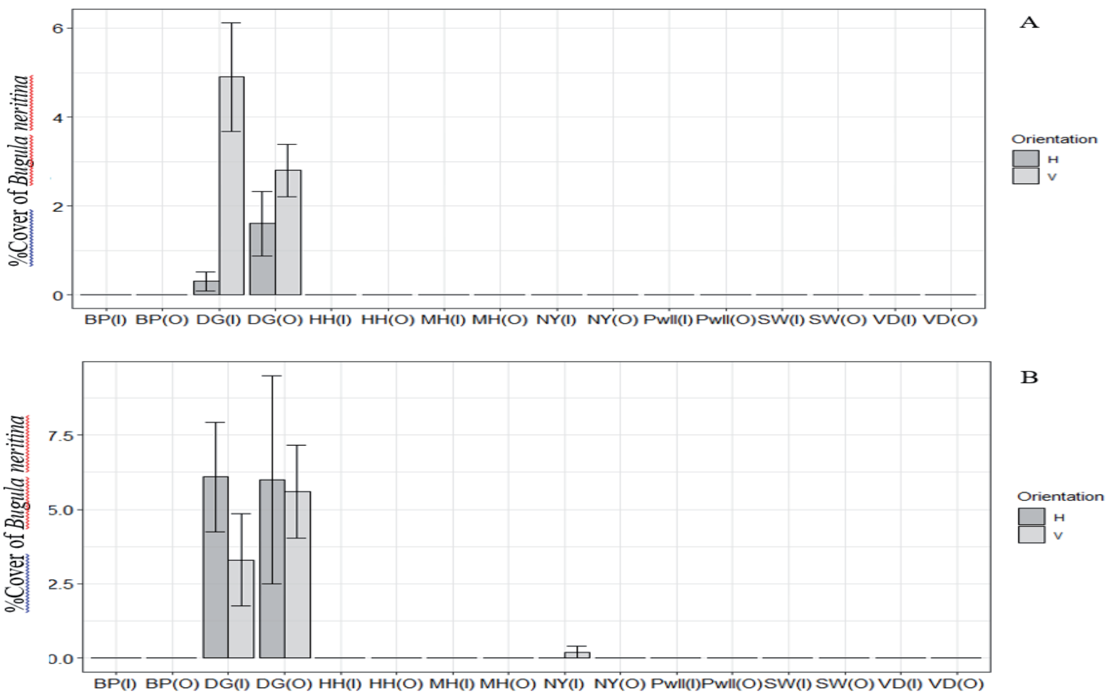
Source	df	Mean Squares(MS)	F-value	P-value
Marina	7	753.8307	34.79	0.0000
Position	1	33.2952	2.44	0.1626
Orientation	1	103.2190	1.20	0.3098
Marina*Position	7	13.6703	0.63	0.7297
Marina*Orientation	7	86.1147	3.97	0.0006
Position*Orientation	1	26.6753	1.04	0.3423
Marina*Position*Orientation	7	25.7125	1.19	0.3151
RES	128	21.6704		
TOTAL	159			

In the early colonization, *B. neritina* is the only species within DG marina in vertical orientation, whereas the abundance of *B. neritina* increased in horizontal orientation in enclosed DG marina in the later colonization period. This means that as the colonization period increased, *B. neritina* preferred to horizontally spread in enclosed DG marina (Figures 10A and 10B).

According to Cochran's test results, the eight-week data had heterogeneous variance (P>0.05).After log transforming, the result of Cochran's test did not change. This shows a significant difference in the abundance of *B.neritina* among marinas [F (7,128) = 73.51, P-value<0.001], but there was no significant interaction between marina and orientation or marina and position. This means the effects of orientation and position do not vary among marinas. However, SNK exploration of this interaction shows the only marina (DG) where *B. neritina* was abundant on vertical orientation rather than horizontal (Figure 10B, Table 6).

Figures 10A and 10B

The Percentage Cover Of Bugula neritina Abundance Collected From Eight Marinas In Two-Meter Depth



Note. BP: Burry Port, DG: Deganwy, HH: Holyhead, MH: Milford Haven, NY: Neyland, Pwll: Pwllheli, SW: Swansea, VD: Victoria Dock in two positions (I): Inner or (O): Outer through two orientation tiles H: Horizontal and V: Vertical at the end of two weeks (A) and eight weeks (B).

T. inopinata has more abundance in horizontal orientation in enclosed DG marina for both colonization periods. As colonization period increases, the distribution of *T. inopinata* has horizontally increased in the enclosed NY marina and vertically in the enclosed SW marina although their abundance has decreased both within and enclosed other marinas. Even so, these invasive non-native species have preferred to distribute vertically outside marinas (Figures 11A and 11B).

According to Cochran's test results, the eight-week data had heterogeneous variance (P>0.05). After log transforming, the result of Cochran's test did not change. This shows a significant difference in the abundance of T. inopinata among marinas [F (7,128) =16.82, P-value<0.001]. There was also a significant interaction between marina and position [F (7,128) =3.13, P-value<0.05], meaning the effect of position varies among marinas. SNK exploration of this interaction shows that there were some differences between positions, but the direction of difference was not consistent (Figure 11B, Table 7).

Table 6

The Results of Three-Way Mixed Model ANOVA Show Differences in Bugula neritina Abundance among Responsible Variables (Level of Significance P-Value<0.05)

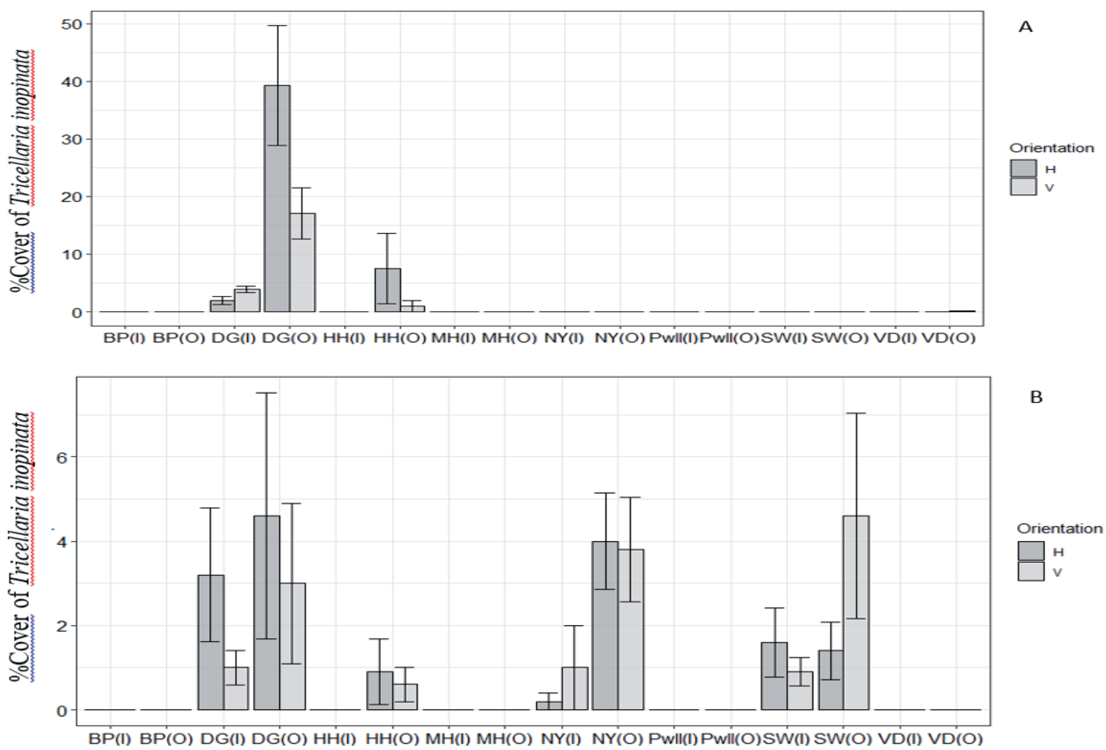
Source	df	Mean Squares (MS)	F-value	P-value
Marina	7	361.0389	73.51	0.0000
Position	1	0.6725	0.34	0.5796
Orientation	1	0.8140	0.37	0.8972
Marina*Position	7	1.9935	0.41	0.7297
Marina*Orientation	7	2.2203	0.45	0.8672
Position*Orientation	1	1.9203	0.50	0.5017
Marina*Position*Orientation	7	3.8280	0.78	0.6057
RES	128	4.9114		
TOTAL	159			

For multivariate analysis, the results of community structure analysis undertaken using PRIMER and the PERMANOVA based on Bray-Curtis similarity for the abundance of dominant species is shown through three responsible factors (Marina-random, Position-fixed, Orientation–fixed) (Table 8), and the produced MDS are presented in Figure 12.

In Table 8, there is a significant effect of marina and significant interaction between marina and position (P<0.05). Then, MDS plots were created individually for each marina to explore significant interactions. These MDS plots demonstrate that there is a clear effect of position for some marinas (VD, BP, and NY). This effect is less clear through an examination of MDS's for orientation (Figure 12).

Figures 11A and 11B

The Percentage Cover of Tricellaria inopinata Abundance Collected from Eight Marinas in Two-Meter Depth



Note. BP: Burry Port, DG: Deganwy, HH: Holyhead, MH: Milford Haven, NY: Neyland, Pwll: Pwlheli, SW: Swansea, VD: Victoria Dock in two positions (I): Inner or (O): Outer through two orientation tiles H: Horizontal and V: Vertical at the end of two weeks (A) and eight weeks (B).

Table 7

The Results of Three-Way Mixed Model ANOVA Show Differences in Tricellaria inopinata Abundance among Responsible Variables (Level of Significance P-Value<0.05)

Source	df	Mean Squares (MS)	F-value	P-value
Marina	7	232.7167	16.82	0.0000
Position	1	153.1174	3.54	0.1019
Orientation	1	0.8928	0.08	0.7912
Marina*Position	7	43.2406	3.13	0.0044
Marina*Orientation	7	11.8000	0.85	0.5457
Position*Orientation	1	3.1887	0.43	0.5317
Marina*Position*Orientation	7	7.3684	0.53	0.8084
RES	128	13.8324		
TOTAL	159			

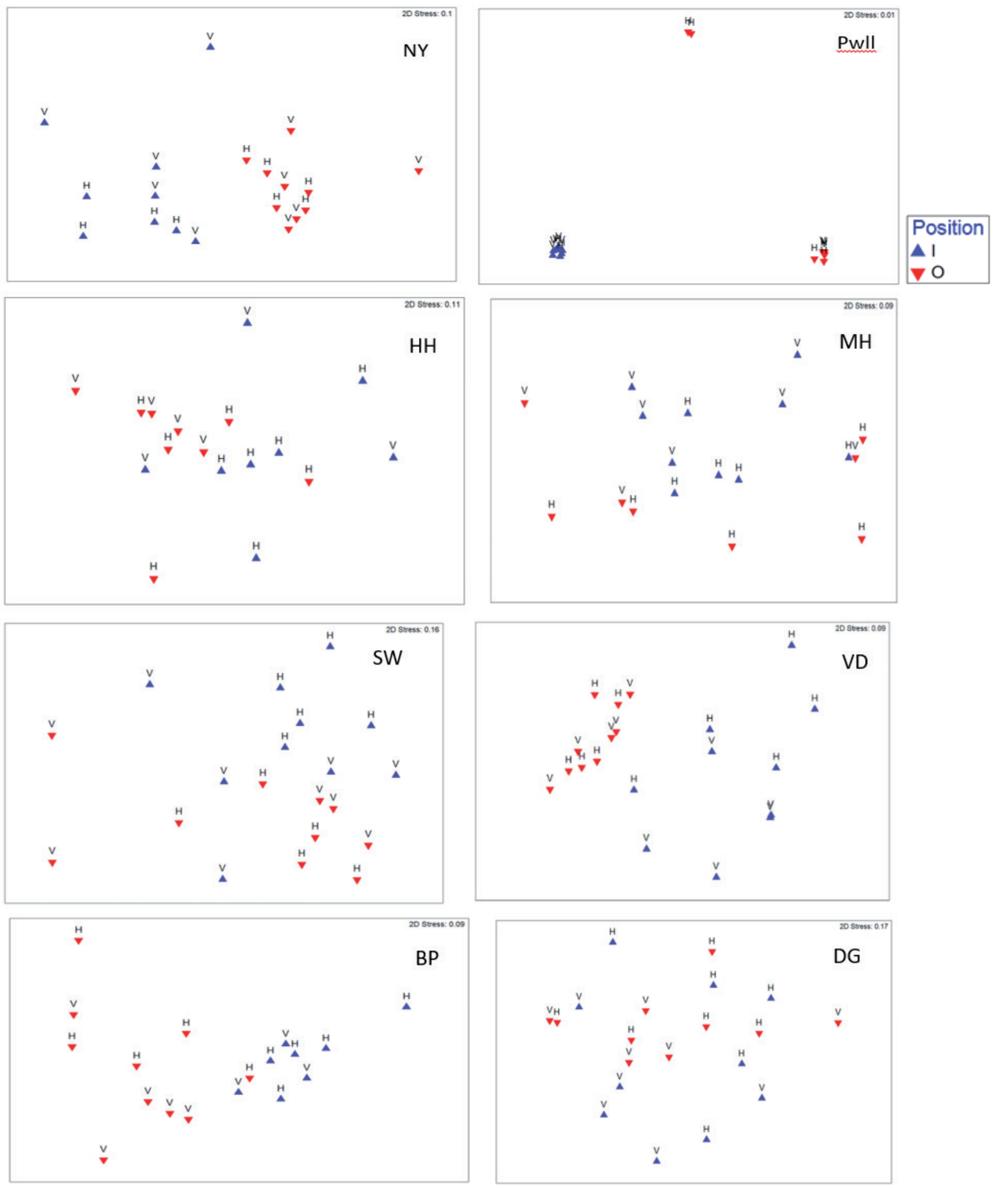
Table 8

PERMANOVA Table of Results

Source	df	Mean Squares (MS)	Pseudo-F	P(perm)
Marina	7	38557	42.262	0.001
Position	1	5989.6	0.85975	0.579
Orientation	1	872.91	0.72952	0.569
Marina*Position	7	6997.6	7.67	0.001
Marina*Orientation	7	1198	1.3131	0.099
Position*Orientation	1	686.42	0.68154	0.612
Marina*Position*Orientation	7	1007.6	1.1045	0.31
RES	120	912.33		
TOTAL	151			

Figure 12

The Significant Interaction Between Fixed Factors (Orientation and Position) and Random Factor (Marina) in Eight Marinas



Note. BP: Burry Port, DG: Deganwy, HH: Holyhead, MH: Milford Haven, NY: Neyland, Pwll: Pwlheli, SW: Swansea, VD: Victoria Dock, in two positions (I): Inner or (O): Outer through two orientation tiles H: Horizontal and V: Vertical at the end of eight weeks.

Discussion and Conclusion

How Do Invasive Non-Natives Distribute in the UK?

The Effects of Invasion

The clarification of invasive non-natives is a crucial pre-requisite step to addressing several fundamental questions in species invasion biology, such as: who are invaders? Where are they coming from? What are the effects they cause in the invaded habitats? (Zhan et al., 2015). This study has mentioned how the species number, their biological and physiological characteristics, and the quality of recipient environment have an important role in the introduction success, changing the propagule size and frequency. In addition, the distribution of invasive non-native or other species affects the other invasive non-native distribution, damaging the ecosystem because the species compete for space for food and habitat changes. However, novel substrates or attaching to other species protects invasive non-natives. Still, there can be stress factors like predators, depth levels, salinity and temperature concentration, tolerance capacity, and competitors.

The Effects of Environmental Factors

Researchers also observed that macro benthic populations exhibit spatial variations along a gradient of size. Arrighetti and Penchaszadeh (2010) proposed that sediment characteristics played a key role in the spatial dynamics of macro benthic assemblies. In this study, potential prospects for colonization of unstable ecosystems are different. The proportion of species have appeared in only a few stations such as VD, BP, and NY. Additionally, competition, predation, parasitism and symbiotic relations between macro benthos may also affect the spatial patterns of macro benthic assemblies. Recognizing major environmental variables that form the distribution of macro benthic assemblies is not an easy task, as they always vary between space and may represent various interactive factors (Lu, 2005). The relationship between the spatial patterns of the macro benthic assemblies and the environmental variables is directly linked to the type of data selected in local conditions (Arrighetti & Penchaszadeh, 2010).

How is The Invasive Non-Native Species Diversity Affected by Marina Characteristics?

The study found there was a significant difference in the abundance of five discriminating species among marinas. Invasive non-native species variations depend on some factors.

Firstly, a key factor is a closeness to the marina. In the marina, the closeness depends on the salinity range and in general marinas which are fully saline are subject to infrequent salinity excursions and harbour more invasive non-native species than brackish water sites or those subject to regular fluctuations e.g. in an estuary (Foster et al., 2016). Holyhead and Bury Port are marina sites, so are more susceptible to invasion more than Deganwy, Victoria Dock, Pwllheli Haven, and Swansea which are brackish water sites. Neyland is both a marina and brackish water site (Wood et al., 2015).

Secondly, the bigger marinas increase the risk of invasion by spreading locally, nationally and internationally, so they have high numbers of invasive non-native species, and are easily accessible (Foster et al., 2016). In this study, the number of invasive non-natives was the highest in Holyhead marina, likely a consequence of its large size and high boat traffic (Wood et al., 2015).

Thirdly, there is the factor of larval retention within more enclosed marinas which may lead to larger populations of NIS (Wood et al., 2015). Larval dispersal and recruitment are likely to occur when water temperatures increase and may result in the spread of invasive non-natives from the marina into the wider harbour area (Arenas et al., 2006). As the temperature decreases, metabolic activity decreases (Kaldy et al., 2015). Thus, as the water temperature increases, the development of species increases. This allows upper intertidal colonization and cold stratification, and increased metabolic activity that may have a significant impact on the species colonization ability (Kaldy et al., 2015). For example, the temperature range in Holyhead Marina is between 5°C and 22°C throughout the year, so it is expected that regressed colonies increase in size during the spring and summer when water temperatures reach above 8–12°C and become favourable for asexual growth. The presence of invasive non-native species in Holyhead Marina and their absence in the wider harbour area suggests that they were introduced into the region on the hulls of one or more infected recreational vessels (Arenas et al., 2006b). SM and VD are superabundant and there is a severe fouling nuisance on yacht hulls, pontoons and ropes (Wood et al., 2015).

In this study, there were significant interactions between marina and position for *A. modestus* and *T. inopinata* and significant interactions between marina and position as well as orientation effect for *C. intestinalis*. *A. modestus* is the most frequently recorded species from marinas around the UK, especially in habitats subjected to fluctuating salinity. *T. inopinata* is a frequently recorded species on primary hard substrates (Wood et al., 2015). *C. intestinalis* grows in the shallower depths (Bishop et al., 2015). According to Darling et al. (2018), through BWE, the diversity of open ocean species increase, resulting in an overall change in population composition. While in many cases this leads to dramatic declines in propagule pressure at recipient ports, several studies have suggested that the efficacy can vary widely depending on the vessel's route, travel duration, biotic composition, and environmental conditions (Darling et al., 2018). In this case, fouling rates change widely between environments, being strongly affected by factors such as local productivity, and the structure and nature of available habitats (Johnson & Shanks, 2003).

Another factor is depth. Shallow water sites may dry out during low tides. Deeper waters can provide refuges from low salinity events, as when the waters are often highly stratified with the freshwater forming a surface layer over a higher-salinity lower base layer (Wood et al., 2015). This means that narrow areas with low salinity have more niche availability because of the freshwater effect, so invasive non-natives are more vulnerable to colonization in shallow areas (Afonso et al., 2020). These species may survive at depth on ropes, chains and pilings and then recolonize rapidly on surface structures at a later date (Wood et al., 2015). In this study, Holyhead, Swansea, and Milford Haven marinas have more invasive non-native species abundance because of the increasing temperature fluctuations with depth. Especially in Swansea Marina, there was a significant interaction between marina and orientation for *B. violaceus*. The depth also affects artificial light presence. Dafforn et al. (2009) stated that marine vessels lead to biological effects of artificial light in marine ecosystems because of decreasing light with depth. Another factor is contaminant concentration. Marina vessels increase contaminants as well as the recruitment of species (Johnson & Shanks, 2003).

In terms of the depth levels, freshwater layers can persist on the surface after heavy rainfall, which is an advantage for *B. neritina* during settlement and transport on a boat and contributes to the spread of invaders on the hulls of ships (Dafforn et al., 2009). In this study, there was no significant interaction between marina and position as well as orientation effect for *B. neritina*. According to Jagerbrand et al. (2019), this outcome can also arise because of unsuitable habitats to invade a marina. In a previous study by Ulman et al. (2019a), it was stated that most invasive non-natives do not invade artificial habitats in the marinas when the surrounding habitats are not suitable for colonization due to limited circulation and/or larval scattering regimes. The high invasion success has been reported in disturbed ecosystems and communities with low species diversity because these disturbed habitats have high and frequent inputs of invasive non-natives in harbours and marinas. This integration increases the invasion success, and thus the colonization risk increases in the established communitie (Riera et al., 2018). In contrast, in polluted marine habitats, propagule pressure restricts invasive non-native settlement compared to other parameters such as abiotic factors (pollutants) or environmental disturbance, so it is interesting to note that under natural conditions, these disturbed habitats and propagule pressures are often related to each other in terms of invasive non-native diversity (Riera et al., 2018).

Overall, the number of NIS per marina varies among marinas in the UK, especially with marinas situated on the south coast of England where there is the greatest NIS number. The research field has a major spatial variability owing to geological, hydrodynamic, and anthropogenic practices. Biotic variables should be viewed in the sense of macro benthic distribution studies. Relationships between the spatial attempts of macro benthic assemblages and the environmental variables of biological parameters need to be considered in certain ways, such that the relationships have proved to shape the distribution of macro benthic abundance, complexity and multivariate composition of assemblages. In this survey, the settlement panels were used for the detection and monitoring of invasive non-natives. This method can be a major problem because identifying many of the species is difficult before they are fully developed and it is challenging to distinguish the invasive non-native species from other, often closely related, native species. Therefore, it is recommended that this method should not be used to collect samples in future surveys.

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Extended Turkish Abstract
(Genişletilmiş Türkçe Özet)

Marina Özelliklerinin İstilacı Yerli Olmayan Türlerin Kolonileşmesi Üzerindeki Etkisi

Deniz ortamında, yerli olmayan türlerin oldukça hızlı gerçekleşen dağılım oranları mevcut diğer türlere göre farklılık gösterir. Bu nedenle, genellikle bazı yerli olmayan türler, hakimiyet anlamına gelen 'istilacı yerli olmayan tür' olarak adlandırılır. Başka bir ifade ile; istilacı yerli olmayan türler önceki doğal dağılımlarının dışında bir alana tanıtılan ve yeni yerleşim ortamları üzerinde olumsuz çevresel, ekonomik veya sosyal etkisi olan türlerdir. Marinalar, çeşitli ve karmaşık doğal ekosistemlere sahip olmalarının yanı sıra, deniz sistemleri arasında güçlü bir bağlantı noktası sunar. Bu durumda, marinaların özellikleri, istilacı yerli olmayan türlerin dağılımları üzerindeki etkisini belirler. Ayrıca, marinalarda tuzluluk aralığı, gelgit dalgalanmaları, derinlik, ve ascidan varlığı gibi faktörler de istilacı yerli olmayan türlerin dağılımını etkiler. Bununla birlikte, taşıma vektörleri, su ürünleri yetiştiriciliği ve balıkçılık uygulamaları da tür istilası sürecinde istilacı yerli olmayan türlerin dağılımında önemli bir role sahiptir.

İstilacı yerli olmayan türler, rekabet, avlanma, çevresel faktörler gibi etmenlerle yerli türler üzerinde olumsuz bir etkiye sahip olabileceğinden, dünya okyanusları için önemli bir tehdit oluşturur. İstilacı yerli olmayan türlerin toplulukları, tanıtıldıkları alanlara göre kendi alanlarında strese daha az toleranslıdır. Bununla birlikte, habitatlarındaki artan biyolojik kirlilik karmaşıklığı, uygun habitatlar, yiyecek ve korunaklı niş alanları sunduğundan ek türlerin girmesine izin verebilir. Fakat, kolonizasyon için yeterli çevresel niş mevcudiyeti yoksa ve avcılar ve rakipler gibi daha büyük biyolojik direnç varsa, istilacı yerli olmayan türlerin istila ettiği ortamda kurulması, ikincil dağıtım ve kolonizasyonu mümkün olmayabilir.

İstilaların büyük çoğunluğu deniz trafiğinden, istilacı yerli olmayan türlerin gemi gövdelerinde taşınmasından ve bölgeye salınan balast sularından kaynaklanır. Balast suyunun, öncelikle serbest yüzen larvaların sınırlı hayatta kalma süresi nedeniyle, uzaktan dağılım için anahtar vektör olduğu anlaşılmasada balast tanklarındaki toprak, zemin ve/veya iç yüzeyler istilacı yerli olmayan türleri dağıtabilir. Bununla birlikte, balast suyu, aracılık ettiği istila riskini azaltmak için geliştirilmiş küresel girişimlere rağmen, istilacı yerli olmayan sucul türlerin girişlerinin güçlü bir vektörü olmaya devam ederek istilacı yerli olmayan türlerin istila sürecinde önemli role sahiptir. Sıcaklık gibi çevresel faktörlerde istilacı türlerin dağılımında önemli rol oynar. Su sıcaklığının artışı istilacı yabancı türlerin sayısını artırarak türün metabolik aktivitesininde artmasına sebep olarak , o türün tür kolonileşme yeteniği üzerinde önemli bir etkiye sahip olur.

Bu çalışmada, saha araştırmalarına ve mevcut literatüre dayanarak, Galler'deki toplam on istilacı yerli olmayan türden; beş istilacı yerli olmayan türün [*Styela clava* (Herdman, 1882), *Didemnum vexillum* (Kott, 2002), *Caprella mutica* (Schrin, 1935), *Crepidula fornicata* (Linnaeus, 1758), *Watersipora subtorquata* (d'Orbigny, 1852)] dağılımı ve ekolojisi, ayrıca diğer beş istilacı yerli olmayan türün [*Austrominius modestus* (Darwin, 1854), *Ciona intestinalis* (Linnaeus, 1767), *Botrylloides violaceus* (Oka, 1927), *Tricellaria inopinata* (D'Hondt & Ambrogi, 1985), *Bugula neritina* (Linnaeus, 1758)] kolonileşmesi üzerinde marina karakterlerinin etkileri araştırılmıştır. Marinalardaki karektarizasyon belirli alanları istilaya karşı savunmasız bırakarak istilacı yerli olmayan türlerin bu alanlarda kolonileşmesine yol açar. Buna bağlı olarak; çalışma alanı herbir marinaya iki farklı konumda olacak şekilde marina girişi ve marina dışı levha olmak üzere yerleştirilmiştir. Bu levhaların yarısı iki haftada (çok erken kolonizasyon) ve yarısı sekiz haftada (sonraki kolonizasyon) olmak üzere her alana eşit sayıda dikey ve yatay olarak konuşlandırılmıştır.

Çalışmanın birincil amacı, Birleşik Krallık'taki önemli istilacı yerli olmayan türlerin ekolojisini ve dağılımını araştırmak ve marina özelliklerinin Galler kıyılarındaki istilacı yerli olmayan türlerin çeşitliliğini nasıl etkilediğini göstermektir. Bu çalışmadaki kilit faktörler bu nedenle marina karakterleri; marina içi veya dışında levhanın konumu; ve kolonizasyon dönemidir. Bu çalışma, yerli olmayan tür çeşitliliğini ve bolluğunu nasıl etkilediklerini analiz etmek için bu faktörlerin her birini değerlendirip, istilacı yerli olmayan tür çeşitliliğinin marina büyüklüğü ile artığını, marina girişlerine kıyasla marinanın kapalı alanlarında istilacı yerli olmayan tür çeşitliliğinin daha fazla olduğunu, istilacı yerli olmayan tür çeşitliliğinin, yatay olarak yönlendirilmiş yüzeylerde dikeyden daha fazla olduğu yönündeki hipotezleri elde edilen veriler doğrultusunda çeşitli istatistik progamlar kullanılarak belirlemeyi hedeflemiştir.

Elde edilen verilere göre; en büyük marinada tekne trafiğinin fazla olması nedeniyle yerli olmayan tür çeşitliliğinin en fazla bu marinada olduğu, konum ve oryantasyonun etkisinin marinalar arasında değiştiği, ancak bazı istilacı yerli olmayan türlerin bolluğunda marina içi ve dışı arasındaki farkta tutarlılık olmadığı belirlenmiştir. Ayrıca, kolonizasyon süresi arttıkça, istilacı yerli olmayan türlerin kapalı marinaların bazılarında dikey, bazı marina girişlerinde ise yatay olarak dağılım yapmayı tercih ettiği belirlenmiştir. Nihayetinde, istilacı yerli olmayan türlerin diğer türlerden ya da yakın ilişkili olduğu yerel türlerden ayırt edilmesinin zor olması, veri eksikliği, deniz ekosistemindeki varyasyonların sürekli değişkenlik göstermesi ya da kullanılan metodların bazı türlerin araştırma zamanında tanımlanması için uygun olmayışı elde edilen çıktıların yorumlanmasını zorlaştırabilir. Fakat, ileride yapılacak çalışmalarda bu kriterlere daha dikkat edilmesi, farklı metodların kullanılması elde edilecek çıktıların daha güvenilir olmasını sağlayacaktır. Ayrıca, bu çalışmanın yorumlanması aşamasında Covid 19 pandemisinin saha çalışmalarını etkilemiş olması ve kullanılan verilerin mümkün olduğunca en güncellerinin kullanılmaya çalışılması, istatistik çalışmaları için kütüphane ve program bulmada sıkıntı yaşanması, yinede bu durumun mümkün olduğunca tolere edilmiş olmasıda göz önünde bulundurulmalıdır.

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Review Article

The Legal Framework of Water Quality Management in Turkey

Türkiye'de Su Kalitesi Yönetiminin Yasal Çerçevesi

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Abstract

After the 1970's, an unprecedented urbanization and industrial activities have resulted in severe inland and coastal water degradation in Turkey. Infrastructure investment in wastewater management has fallen behind schedule due to the lack of required extensive financing to implement large projects. On the other hand, land-use planning has been overlooked due to increased demand for housing and legal gaps that weaken the enforcement in that respect. Turkey is updating its regulations within the framework of the current EU directives after becoming a candidate country to the European Union (EU). Comprehensive reforms related to institutional and legal issues have gradually taken place with the objective of meeting the defined strategies for water pollution abatement. A new "Water Law " has been drafted for ensuring a more efficient water management by updating the existing legal framework where needed and with emerging issues (e.g. climate change). In 2021, a "Consultative Assembly on Water" was established to analyze the content and items of the new "Water Law ", in particular, and the conclusion statement was displayed. In this study, Turkey's water quality management policies are presented by taking into consideration the present urgent actions and future needs, development at national, regional and municipal levels as well as the on-going EU accession process together with regional and international agreements. An assessment of inland and coastal water quality is done according to the regulations in force and set standards.
Keywords: *water quality assessment, coastal waters, inland waters, wastewater management, EU accession*

Öz

Türkiye'de özellikle 1970'lerden sonra, hızlı kentleşme ve endüstriyel faaliyetler, kıta iç ve kıyı sularında ciddi sorunlara neden olmuştur. Atık su yönetimine yönelik olan altyapı yatırımları büyük projeler için gerekli olan yüksek maliyetlerin karşılanamaması nedeniyle planlanan hedeflerin gerisinde kalmıştır. Ayrıca, artan konut talebi nedeniyle arazi kullanım planları göz ardı edilmiş olup yasalardaki boşluklar da bu yöndeki yaptırımı zayıflatmıştır. Türkiye, Avrupa Birliğine (AB) aday ülke olmasını takip eden süreçte yönetmeliklerini mevcut AB direktifleri çerçevesinde güncellemektedir. Atık su kirliliğini azaltmak için belirlenen stratejileri hayata geçirebilmek amacıyla kademeli olarak kurumsal ve yasal yapıya yönelik kapsamlı iyileştirmeler gerçekleştirilmiştir. Daha etkin bir su yönetimi sağlamak amacıyla, gerekli olması durumunda mevcut

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yasal yapıyı güncellemek ve iklim değişikliği gibi ortaya çıkan yeni sorunları da dahil edebilmek üzere yeni bir "Su Kanunu" taslağı hazırlanmıştır. 2021 yılında esasen "Su Kanunu" nun içerik ve maddelerinin incelendiği bir "Su Şurası" oluşturularak sonuç bildirgesi yayınlanmıştır. Bu çalışmada, mevcut acil eylemler ve gelecekteki ihtiyaçlar, devam eden AB katılım süreci ile birlikte ulusal ve uluslararası anlaşmalar da dikkate alınarak, Türkiye'nin su kalitesi yönetimi politikaları sunulmaktadır. Kıta içi ve kıyı sularının kalite değerlendirilmesi belirlenen yönetmeliklere ve standartlara göre yapılmıştır.

Anahtar sözcükler: *su kalitesinin değerlendirilmesi, kıyı suları, iç sular, atık su yönetimi, AB uyum süreci*

Introduction

Impacts on water quality may be generated by many various anthropogenic factors. These are (i) uncontrolled population growth that may create large volume of domestic pollution, difficult to assimilate by the receiving environment; (ii) industrial development without adequate infrastructure; (iii) dirty energy choices; (iv) intensive land-use and hydrologic changes; (v) climate variability and changes. Furthermore, severe climatic events result in more recurrent and also extreme and long-term droughts and floods with impacts on ecosystems, human livelihoods and infrastructure (Ülker et al., 2018; Wilk & Wittgren, 2009).

Inadequate water management has caused serious water quality problems in many parts of the world. In point source pollution control, developed countries have reacted to this problem with significant and commendable progress, in particular. However, commensurate progress has not occurred in developing countries. Subsequently, most water resources within and around the urban centers have become greatly polluted because of inadequate water management (Biswas & Tortajada, 2011). Turkey has been among these examples. As a consequence, after the 1980s this alarming situation has led the governmental and local authorities to launch comprehensive and enhanced actions for remedial measures and protection of the environment. The water management policy in Turkey has developed considering its growing population, rapid urbanization, and global and regional developments with the main target of providing the present and future population with required water needs (Burak et al., 2021a; Burak et al., 2021b; Savun-Hekimoğlu et al., 2021).

The national aim is to engage in a sustainable development policy based on the principles of international conventions to which the country is a signatory. For the last three decades, the government has been struggling to increase awareness on environmental issues and encourage implementation of conservation measures in the national policy documents (i.e. development plans). After 2000, as a negotiating candidate country to the European Union (EU), Turkey has started the preparation of harmonizing its legislation with that of the EU (European Commission [EC], 2000). Subsequently, significant modification in water-related legal framework has been introduced with further improvement expected to be realized in the near future.

Insomuch that; "2020-2023 National Smart Cities Strategy and Action Plan" (Ministry of Environment and Urbanization [MoEU], 2019) has been prepared in order to guarantee that the investments are implemented with appropriate projects and activities in line with the smart city ecosystem. Regarding water management, one of the themes of the Smart Environment component, many targets and policies are included in international policy documents. Especially, high-level implementation steps are defined in the Smart Environment Component item number 15.2 (MoEU, 2019). The most important of these sub-items; improving water resources, reducing and treating wastewater, minimizing water losses, encouraging water reuse, storage, water conservation and sustainable use.

In this study, water quality of both inland and coastal waters in Turkey is evaluated; its management with regard to globally and nationally accepted criteria is assessed with a focus on evolving institutional and legal structure in Turkey. In addition, emerging concepts and policies within the framework of the EU Accession process are highlighted.

Institutional and Legal Framework of Water Quality Management

Turkey has met environmental concerns comparatively late, but from the 1970's onwards, soaring volume of pollution in inland and coastal waters with severe consequences on natural resources have caused environmental problems at dangerous levels. The worst offended water resources and coastal waters by anthropogenic pollution have been the ones located within or near by the western large metropolises and coastal cities (e.g. Istanbul, Izmir, Mersin, Antalya) due to in-migration from the eastern part of the country and natural growth. Furthermore, coastal cities on the Aegean and Mediterranean shoreline have been subjected to population increase in summer season due to touristic influx. The delay in the implementation of sewerage infrastructure has led to an increasing volume of domestic and industrial pollution, in particular. The lack of efficient and regular monitoring of receiving water bodies and enforcement of the ruling pollution control regulations have aggravated environmental concerns with regard to the public health as well as the health of aquatic ecosystems (Zeki et al., 2021). The regulatory framework for water quality management which defines policies (guidance documents), implementing measures (regulations) and enforcement (how to implement the set regulations) may vary between and within countries; even in degrees of efficiency (Cross & Latorre, 2015).

Previously, the introduction of five-year plans was the first attempt at the adoption of a long-term and centralized policy-making approach related to public investments, whose planning and programming was entrusted to the former State Planning Organization (SPO), replaced by the Ministry of Development at present. Burak (2008) stated the following:

> Since the 1920s, measures to prevent water pollution have been incorporated in numerous laws, regulations and directives enacted by Parliament and other authorized bodies, and in the provisions of international conventions. Most of this legislation, including that in the Constitution, embody provisions for protection of the environment and public health (p. 167).

The Republic of Turkey has been undergoing significant changes in its legal and institutional structures since its establishment in 1923. The historical legal basis related with water can be summarized as follows:

The Law on Waters, No. 831 (The Official Gazette No: 368, 1926), which entered into force in 1926, is the pioneer of water resources legislation in Turkey. This is the first one targeting surface waters with the one entered in force later in 1953, the establishment Law No. 6200 (The Official Gazette No: 8592, 1953) of the State Hydraulic Works (DSI). This law empowers DSI to develop surface and groundwater resources, as such, indirectly, it is a legal tool regulating water resources. These are followed by the “Groundwater Law” No. 167, promulgated in 1960 (The Official Gazette No:10688, 1960) to be enforced by DSI. Additionally, the Environment Act of 1983 (The Official Gazette No:18132, 1983) with its Water Pollution Control Regulations (WPCR) has been a valuable legal basis for regulating inland waters as well as discharge standards and coastal waters (Burak, 2008).

The legal structure of the Environment Act consists of a system of technical regulations and standards that specify the principles of implementation. WPCR (The Official Gazette No: 25687, 2004) promulgated in 1988 is one of the most crucial and inclusive components of this system. Later, Surface Water Quality Regulation (SWQR) (The Official Gazette No: 28483, 2012) is issued in 2012 and the revisions of the regulation are issued in 2015, 2016 and 2021. These regulations specify the technical principles for the protection of surface and groundwater with the aim of meeting human demands while conserving the quality of water resources. On the other hand, Water Framework Directive (WFD) (EC, 2000) and its sub-directive Marine Strategy Framework Directive (EC, 2008) aim to achieve good environmental status in all European water bodies. Turkey, as a candidate country to the EU membership should proceed in accordance with these directives.

The protection of coastal water quality is a fundamental part of the regulations; whose objectives are: (1) protection of potential water resources; (2) efficiently management of water resources; (3) prevention/elimination of water pollution. Principle of the regulations is based on the "polluter pays" principal. For this reason, polluters have to inform the authorities about the amount and content of their wastewaters and apply for a discharge permit in which the conditions for discharge and the amount of mandatory treatment are stipulated. Among other items, the regulations define the conditions of use of municipal sewerage and treatment systems, discharge standards and the conditions for payment. Provisions concerning hazardous wastes in aquatic environments are defined in the WPCR. Metropolitan municipalities are also authorized by Metropolitan Municipality Law enacted in 1981 (The Official Gazette No: 25531, 2004) to specify and apply within their boundaries the legislation required for the most efficient management of water and wastewater facilities.

Inland Water Quality

Surface Water Quality

Inland waters quality is classified of four classes based on WPCR (2004) from indicating the best quality (I) to indicating the worst quality (IV). However, revised SWQR in June 2021 is classifying water quality of three classes as Class I: very good, Class II: good and Class III: moderate. Quality criteria of inland surface waters are given in Table 1. In addition, same regulation classifies eutrophication status of lakes, ponds and reservoirs using Trophic Level Index from ultraoligotrophic to hypertrophic (The Official Gazette No: 28483, 2012).

The regulation also determines the sensitive region and sensitive water area in terms of trophic levels for both inland and coastal waters in order to stipulate the protection measures and control the nutrient pollution for the region/area. The criteria about trophic level are given in the regulation for inland and coastal waters for the Mediterranean, Aegean, Black Sea and Marmara Sea. Considering the degradation effect of eutrophication on water quality, some of the indexes are developed worldwide for monitoring and evaluating (Ülker et al., 2020). Trophic State Index (TRIX) is one of them and applied also in the Mediterranean, Aegean Sea, Marmara Sea and Black Sea between the years of 2014-2017 by the MoEU (MoEU, 2017).

Chemical and Physicochemical Quality Criteria of Inland Surface Waters

Quality Criteria	Water Quality Classes		
	I (very good)	II (good)	III (moderate)
Color (m^{-1})	RES 436 nm: ≤ 1.5 RES 525 nm: ≤ 1.2 RES 620 nm: ≤ 0.8	RES 436 nm: 3 RES 525 nm: 2.4 RES 620 nm: 1.7	RES 436 nm: > 4.3 RES 525 nm: > 3.7 RES 620 nm: 2.5
pH	6-9	6-9	6-9
Conductivity (µS/cm)	< 400	1000	> 1000
Oil and grease (mg/L)	< 0.2	0.3	> 0.3
Dissolved oxygen (mg/L)	> 8	6	< 6
Chemical oxygen demand (mg/L)	< 25	50	> 50
Biochemical oxygen demand (mg/L)	< 4	8	>8
Ammonium (mg NH_4^+-N/L)	< 0,2	1	>1
Nitrate (mg NO_3^--N/L)	< 3	10	> 10
Total kjeldahl-nitrogen (mg N/L)	< 0.5	1.5	> 1.5
Total nitrogen (mg N/L)	< 3.5	11.5	> 11.5
Ortho phosphate (mg o-PO_4-P/L)	< 0.05	0.16	> 0.16
Total phosphorus (mg P/L)	< 0.08	0.2	> 0.2
Fluoride (µg/L)	≤ 1000	1500	> 1500
Manganese (µg/L)	≤ 100	500	> 500
Selenium (µg/L)	≤ 10	15	> 15
Sulfur (µg/L)	≤ 2	5	> 5

Groundwater Quality

Groundwater quality has deteriorated due to over abstraction by domestic and agricultural use, which gave rise to salinization of the aquifers in coastal settlements. Monitoring of groundwater quality in Turkey is done in accordance with the Regulation on the Protection of Groundwater Against Pollution and Deterioration (The Official Gazette No: 28257, 2012). The purpose of this Regulation is to determine the necessary principles for maintaining the conditions of good groundwater, preventing its contamination and degradation and improving groundwater quality. This regulation determines the groundwater quality parameters and standards for threshold value.

Investment in hotel and summerhouse construction has resulted in a soaring volume of pollution that exceeded the assimilative capacity of some coastal waters. This accelerated influx to the coastal areas happened as a result of unearned and real income expectations (e.g. Izmir, Antalya, Mersin). Water has been supplied mostly from groundwater and in excessive amounts to satisfy the demand of the newly developed settlements, lowering the water table and resulting in sea-water intrusion[1] in most of the coastal aquifers. Additionally, deterioration of water quality as a result of fertilizers and pesticides used in irrigated agriculture is another major problem in Turkey, especially in the Mediterranean, Aegean, Central Anatolia, and Marmara regions. Due to heavy fertilization, nitrate and nitrite contamination is very common in these regions, where the levels are above the standards (Burak et al., 2004).

River Basins Protection Action Plans (RBPAPs) and River Basin Management Plans (RBMPs)

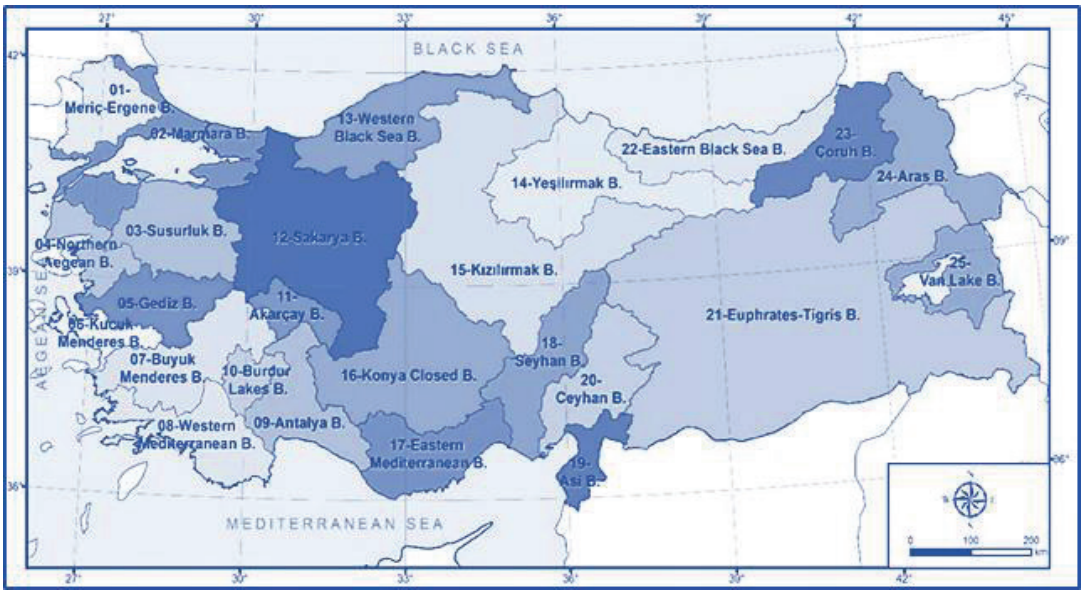
Turkey comprises of 25 hydrological watersheds (Fig. 1). Among them Konya, Akarçay, Van Lake and Burdur Basins are closed watersheds and Meriç-Ergene, Asi, Fırat-Dicle, Aras Basins are transboundary watersheds.

Within the scope of the harmonization Turkish legislation with that of the EU and WFD, regulations on the protection of watersheds and preparation of River Basin Management Plans (RBMPs) was first published on 17/10/2012 (The Official Gazette No: 28444, 2012). Then, it was amended on 28/10/2017 as a Regulation on Preparation, Implementation and Monitoring of the River Basin Management Plans (The Official Gazette No: 30224, 2017) and lastly revised in 11/01/2019. As a prerequisite of the management plans, the preparation of the RBPAPs were completed for all the 25 basins by the end of 2015 aiming at protection and planning of the river basins and wetlands with a holistic approach in terms of physical, chemical and ecological aspects. After the completion of RBPAPs, the RBMPs have being prepared. RBMPs of 8 basins including Gediz, Meriç-Ergene, Büyük Menderes, Konya, Susurluk, Burdur, Küçük Menderes and North Aegean Basins have been completed/issued by October 2021 (Ministry of Agriculture and Forestry [MoAF], 2021).

[1] Overexploitation of coastal aquifers causes the lowering of freshwater level and seawater to flow into the aquifer and to replace the freshwater- a phenomenon known as 'saline intrusion'

Figure 1

Hydrological Watersheds of Turkey (MoFWA, 2013)



Marine and Coastal Water Quality

Marine and coastal waters of Turkey are under pressure by various types of pollution sources. According to United Nations Environment Programme/Mediterranean Action Plan (UNEP/MAP), approximately 80% of marine pollution is generated by land-based activities and 20% is ship-originated (UNEP/MAP, 2012). The sources of land-based pollution can be enumerated as: (1) river discharges carrying point and diffuse pollution; (2) untreated domestic, industrial wastewater, leakage from land-fills; (3) storm-water discharges; (4) pollution generated by ports and marinas; (5) cooling water of thermal power plants; (6) marine litter; (7) aquaculture farms. Pollution conveyed through transboundary sources from the Danube River and Black Sea has also a significant impact on marine and coastal waters of the inland Marmara Sea, in particular (Çiçekalan & Öztürk, 2018). Also, marine litter, oil spills, tanker and pipeline accidents are among the considerable sources of marine pollution (Doğan & Burak, 2007; Kunt et al., 2016; Priority Actions Programme/Regional Activity Centre [PAP/RAC], 2005; Ülker & Baltaoğlu, 2018).

Table 2

Water Quality Classes in Hydrological Watersheds and Total Point Loads (MoEU, 2017)

Basin No	Basin	COD	BOD	NH_4	NO_2	NO_3	Tot. P	Overall	Contribution of Pt. Loads to Total TN	TP
11	Akarcay	I-II	III-IV	IV	IV	I	IV	IV	19	34
9	Antalya	I-II	I-II	I-II	I-II	I-II	II-III	II	9	22
24	Aras		I-II	I-II	I-II	I-II		I-II	8	20
19	Asi	I-II	I-II	III-IV	IV	I		IV	27	50
8	W. Mediterranean	I-II	I-II		I-II	I	II	I-II	10	27
13	W. Black Sea	I-II	I-II					II-IV	21	42
10	Burdur	I,IV		II, IV	IV	I		IV	4	4
7	B. Menderes	II-III		II-III	IV	I		III	12	16
20	Ceyhan	I		I, III	III-IV	I		III-IV	16	20
23	Coruh	I-II	I-II	I-II	II-III	I	II-III	II-III	5	21
17	E. Mediterranean	I	I	I-II	I-III	I		I-III	12	30
22	E. Black Sea	I	I	I	III	I	III	I-III	17	48
1	Ergene	IV	III	III	III-IV	I		III-IV	26	51
21	Euphrates&Tigris	I-II		I-IV				I-IV	19	39
5	Gediz	I-II	II-III	I-IV	III-IV	I		III-IV	26	44
15	Kizilirmak	I-II		II-IV	III-IV	I-II	II-IV	II-IV	12	15
16	Konya	III	III	II	IV	II		III-IV	4	12
4	N. Aegean	I-IV	I-IV	I-IV	III-IV	I-II		I-IV	17	31
6	K. Menderes	IV		IV	IV	I		IV	18	27
2	Marmara	I-IV	I-IV	I-IV	III-IV	I-II		II-IV	26	33
12	Sakarya	II-IV		II-IV		I	II-IV	III-IV	36	56
18	Seyhan	I		II-III	III-IV	I	II-IV	II-IV	10	11
3	Susurluk	I-II		II-IV	IV	I, III		III-IV	16	23
25	Van Lake		II	II	III-IV	I		I-IV	11	31
14	Yesilirmak	III-IV	III-IV	III-IV	III-IV	III-IV	III-IV	III-IV	10	15

Assessment of marine/coastal water quality is key to best management practices. Within this scope, Turkey developed joint research programs with white riparian countries and has become a party to the international environmental conventions like Barcelona Convention and Bucharest Convention. Regional monitoring studies were carried out according to these conventions, its protocols and programmes as MEDPOL (Mediterranean Marine Pollution Assessment and Control Programme) in Mediterranean/Aegean Sea, BSIMAP (Black Sea Integrated Monitoring and Assessment Programme) in Black Sea and MEMPHIS (Environmental Master Plan and Investment Strategy for the Marmara Sea Basin) in the Marmara Sea until 2011. Then, after the 2000s, the Water Framework Directive (WFD) and the Marine Strategy Framework Directive (MSFD), which emphases ecosystem-based management and integrated monitoring approaches, entered into force in the EU. Turkey has adopted these new approaches of monitoring strategies and within this context physical, chemical and biological characteristic of seawater, sediment and biota has been started to be monitored under "Integrated Marine Pollution Monitoring" programme by the Ministry of Environment and Urbanization since 2011 (Olgun Eker et al., 2016). So far, two monitoring periods (2014-2016 and 2017-2019) have been successfully completed and recent researches are on-going related to the classification of the Turkish marine/coastal water quality which will cover 2019-2022 period.

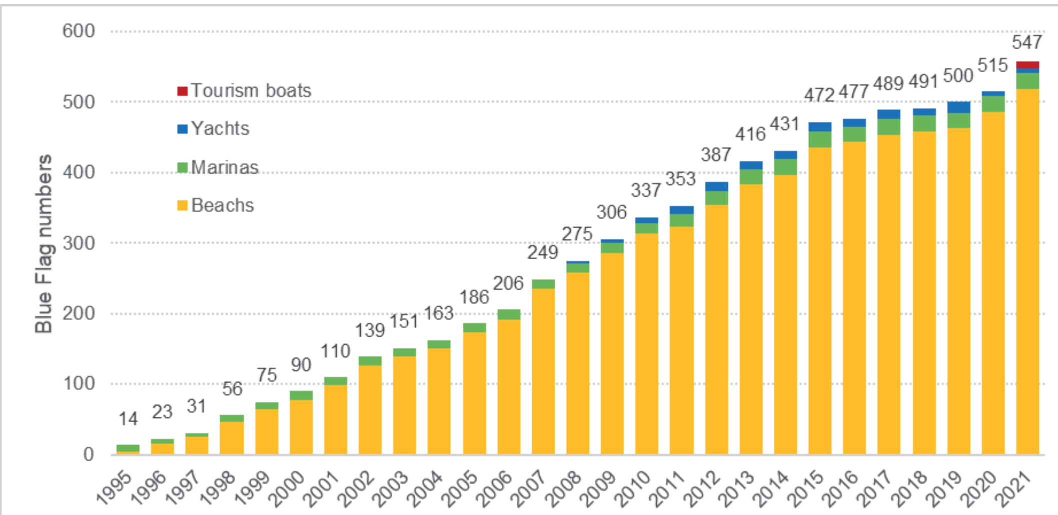
Bathing Water Quality and Blue Flag

Bathing water quality is monitored according to revised Regulation on the Management of Bathing Water Quality (RMBWQ) (The Official Gazette No: 30899, 2019). This Regulation was issued by taking into account the EU Bathing Water Directive (BWD) (2006/7/EC). Both have the same quality criteria and classifications for inland, coastal and transitional waters. Classification of bathing waters is done based on indictor bacteria (*E. coli* and intestinal enterococci) levels as very good, good, sufficient and poor. There are over 1500 coastal and inland bating waters which water quality is monitored by Ministry of Heath of Turkey. Data is available to the public through the Ministry of Heath, Bathing Water Quality Tracking System web page (Ministry of Heath, 2021).

Turkey has been participating to the Blue Flag programme since 1993, which has been a driving force to meet and maintain a series of stringent environmental, educational, safety-related and access-related criteria for qualifying the Blue Flag-awarded facilities. In total 547 beaches, marinas private yachts and tourism boats were awarded with the Blue Flag Certification in 2021. The total number has increased every year (Fig. 2) (Blue Flag, 2021).

Figure 2

Number of Facilities Awarded with Blue Flag over the Years in Turkey



Wastewater Pollution and Control

Construction of up-to-the-standards sewerage facilities began in the late 1960's initiated by the former Bank of Provinces (restructured as ILBANK). New sewerage projects have been designed on separate systems taking into account land development projections. In urban areas more than 75% of the population is connected to the sewerage network on the average. Due to high investment costs, storm water collection systems have been constructed only in limited flood prone areas of big cities (Burak & Demir, 2016). Today, wastewater management is carried out according to Urban Wastewater Treatment Regulation (UWTR) (2006) which covers the technical and administrative principles related to the collection, treatment and discharge of urban and certain industrial wastewater, monitoring, reporting and inspection of wastewater discharge (The Official Gazette No: 26047, 2006).

So far, in coastal settlements, the final disposal by deep-sea outfall of collected wastewater after primary treatment has been a common practice. This practice is mainly based on the optimum dilution and dispersion mechanism of wastewater in the marine currents. The deep sea discharges (marine disposal) of untreated or partially treated wastewater constitutes an alternative treatment and disposal strategy. In Turkey, this practice is a commonly used practice by the municipalities located on the Black Sea cost. Dispersion models are needed to estimate the dilution of wastes because of mixing and transport. The degradation of organic matter and die-off of pathogens and viruses also effect the pollution reduction. The efficient mixing through adequate outfall diffusers is significant for the dilution of wastewater effluents in the immediate vicinity of the outfall (Uslu, 1985). Table 3 gives the discharge Criteria for Deep Sea Outfalls and Figure 3 gives the location of wastewater treatment plants all over the country.

Table 3

Discharge Criteria for Deep Sea Outfalls (MoEU, 2017)

Parameter	Limit	Notes
pH	6-9	
Temperature	35˚C	
Suspended Solid (mg/L)	350	
Oil and grease (mg/L)	15	
Floating substances	Should not be present	
5-day Biological Oxygen Demand (mg/L)	250	
Chemical Oxygen Demand (mg/L)	400	
Total Nitrogen (mg/L)	40	
Total Phosphate (mg/L)	10	
Methylene Blue Surface Active Agents (mg/L)	10	Note 1
Other parameters		Note 2

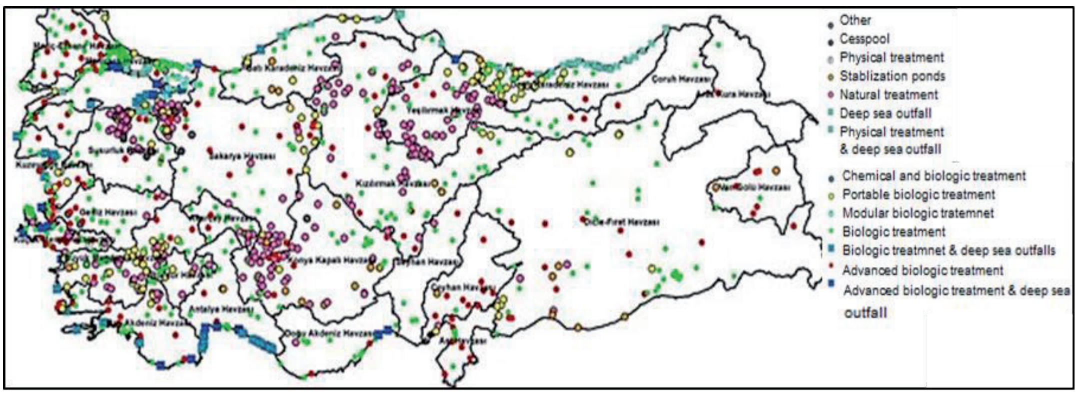
Note 1. Discharge of substances whose biodegradation does not comply with the Turkish Standards Institute is prohibited.

Note 2. It must comply with the limit values given for these parameters in the Regulation on the Amendment of the Regulation on the Control of Pollution Caused by Dangerous Substances in Water and its Environment (The Official Gazette No: 26040, 2005).

The treatment level of domestic wastewater to be discharged into the receiving media is assessed under three categories based on the population figures. The regulations prescribe a comprehensive list of effluent standards particular to domestic wastewater treatment works discharging directly to watercourses and sea, and individual industries. Areas of high ecologic importance and sensitive to environmental pollution must be given special importance as stipulated in the related clause of the Environment Act Advance treatment is gradually being introduced to the wastewater treatment plant design located in touristic coastal areas, special protected areas and water protection basins (Burak & Mat, 2020).

Figure 3

Location of Wastewater Treatment Plants (Evsel/Kentsel Atık su Arıtma Tesislerinin Mevcut Durumunun Tespiti, Revizyon İhtiyacının Belirlenmesi Projesi [TÜRAAT], 2017)



Discussion

After 2000, Turkey became an EU accession (candidate) country. The harmonization of the national legislation with that of the EU has accelerated legislative modifications with the promulgation of several regulations on river basin management and water quality, in particular. Among these, the followings can be cited as prominent achievements:

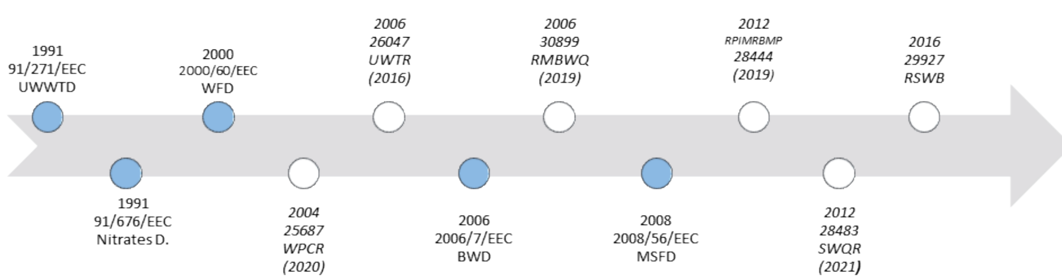
- Regulation on "Preparation, Implementation and Follow-up of Water Management Plans" issued in 2012 and the Communique on the establishment and duties of Basin Management Coordination Councils, in 2019, in relation with river basin management planning
- Regulation of Protection of Water Against Agricultural Nitrate Pollution (The Official Gazette No: 29779, 2016), Regulation on Monitoring and Implementation of Flood Management Plans (The Official Gazette No: 29710, 2016), Regulation on Protection of Groundwater Against Pollution and Deterioration (The Official Gazette No: 28257, 2012) and UWTR in relation with water quality.

The timeline of the EU directives and regulations in Turkey related to water management is given in Figure 4. Each directive indicates the requirements for the aim of achieving of good environmental status in inland, coastal and transitional waters. The criteria for the urban wastewater discharges were issued in 2006 in UWTR same with EU Urban Waste Water Treatment Directive (UWWTD) (1991), while WPCR (2004) has already more detailed criteria considering the population and BOI_5, COD and TSS load. EU BWD and MSFD are issued in 2006 and 2008, respectively and RMBWQ in 2019 is issued with the same criteria and classification of the EU BWD. The regulation was published in the Official Gazette No: 29927 dated 23.12.2016 with the name of "Regulation on Sensitive Water Bodies and the Determination of Areas Affecting these Bodies and Improving Water Quality (RSWB)" according to harmonization of the directives: WFD, UWWTD and Nitrates.

The achievement following the requirement for the EU/WFD has been (a) sound economic instruments for sustainable water management, (b) drought and flood management and preparedness together with climate change impacts inserted in planning documents, (c) improvement in institutional structure and legal framework integrated into national policy decisions (Burak & Margat, 2016).

Figure 4

Timeline of the EU Directives (Blue) and Regulations in Turkey (White)



*Years in parenthesis indicate last revision date.

However, current actions in pollution reduction and water resource management should be reinforced to remedy problems, some of which are river basin-specific and call for improved river basin management planning. Because although water-related legal and institutional structure of Turkey has been improving during the last decades as summarized in above, more improvement is needed to meet the requirements of the WFD which Turkey has to comply with in line with the harmonization of her legislation. Most importantly, due to the fact that available groundwater and surface water reserves fall short to meet the demands of the growing population with regard to quantity and quality. Therefore, there is a need for a comprehensive legal and institutional structure, which is expected to be covered by the new Water Law. This has been agreed at all levels by all national instances to replace/update the existing laws (e.g. the Law on Waters) which do not respond adequately current water needs. Thus, it is expected that the new "Water Law" will be the prominent legal document for ensuring the needed improvement. The main objectives of the new "Water Law" are addressed as (i) protecting, upgrading and improving water resources; (b) ensuring sustainable water use based on priorities on the needs; (c) ensuring integrated water management by one fully responsible body on the basis of river basin management by taking water quantity and quality into account; (d) ensuring water use efficiently by promoting physical savings and quality savings by allocating water to appropriate uses; (e) allocating water use by one fully responsible body; (f) regulating the legal aspects of water by a Law; (g) ensuring compliance with EU legislation.

Currently in Turkey, there are three regulations (RBMPs, SWQR, RSWB) in force and harmonized with the EU WFD. However, it is recommended to establish and/or expand a monitoring network, including biological monitoring, as soon as possible to fully harmonize the routine monitoring activities with WFD (MoAF, 2019).

One of the main components of water quality management depends on an overall reliable wastewater pollution control strategy. Wastewater treatment standards in Turkey are regulated by two regulations, both are in force: WPCR (2004) and the UWTR (2006). These regulations with incoherent treatment standards cause confusion in practice.

> So far, the common practice has been to take from each regulation the more stringent standard for each parameter and to request municipalities to comply with these new sets of standards, which do not comply with either of the regulations in force (The World Bank, 2016).

As a result, some issues hinder the implementation of the EU Directives. The prominent issues are overlap and conflicts in regulations, planning and institutions that can be explained as:

- Regulatory: Although a comprehensive set of regulations exists with stringent standards, the enforcement is not satisfactory in many cases (e.g.

the operation and maintenance of public-owned wastewater treatment plants, two regulations in force, other standards applied in practice, all more stringent than the EU standards resulting in failure of enforcement in most of the cases).

- Institutional: Many institutions deal with aspects of the sector (Burak et al., 1994). At present, there are more than 10 institutions responsible for water related issues as per their legal framework/establishment law (e.g. the Ministry of Environment and Urbanization, Ministry of Agriculture and Forestry, Ministry of Health, Ministry of Energy and Natural Resources at central level and Metropolitan Municipalities, at local level) among others. They are assigned different, sometimes overlapping tasks in relation with their duties and responsibilities which convey to fragmented water management. As a result, this creates also various controversies in water management. Therefore, one of the targets of the new Water Law has been put forward as to improve the existing institutional fragmented structure in the water sector in order to ensure a more efficient management. A "Consultative Council on Water" was formed with the participation of representatives from all water-related governmental institutions, public institutions, NGOs, and private sector on 29 March 2021. The issue related to institutional shortcomings have also been discussed by the Consultative Council on Water finalized with a "Conclusion Statement" published on 21 October 2021 (MoAF, 2021). An in-depth institutional critical analysis and required improvement as put forward by the 28 items of the "Conclusion Statement" will be the subject of another study.

- Planning: Numerous action plans and investment programs overlap. Based on the above mentioned key issues, it seems obvious that there is a need for a vision clarity, a coherent articulation of strategic concerns, the formation of apparent responsibilities and incentives at each level of responsibility, setting out what should be subject to central regulation and consistent, national and enforceable standards considering economic affordability, and the implementation process (Burak & Ülker, 2018). In accordance with this requirement, reconsideration of the existing legislation is necessary according to real requirements in existing/planned fields based on a comprehensive assessment of conditions in Turkey and long-term data. Feasibility, implementation of enforcement and economic discourage should be priority in this process.

Conclusion

The Republic of Turkey has acquired a long experience since the promulgation of the Law on Waters (The Official Gazette No: 368, 1926) in 1926. Also, it has well equipped facilities, trained and experienced staff and potential financial possibilities to improve water resource management and related works that will comply with the obligations of the EU WFD. However, leadership and coordination responsibilities are not fully established yet and a lack of cooperation may exist because of undefined roles. Furthermore, experience related to RBMPs is expected to be gained as one of the outputs of recently finalized and ongoing projects related to "the Conversion of RBAPs into RBMPs". The implementation of the WFD offers a potential for improvement in this respect and opens different opportunities to stimulate the development of human resources, inter-institutional cooperation and coordination, international support and partnership which will be the building blocks of ongoing efforts. As stated in the "National River Basin Management Strategy (2014-2023)", the national vision is expressed as to "become a country that is water, food and energy secure and climate resilient". Therefore, on the way to comply with this strategy, managing and restoring aquatic ecosystems, goods and services together with a water-efficient and water-saving economy is considered of the utmost importance. To achieve this target, there is still substantial need for data collection and monitoring for carrying out comprehensive and reliable researchers for further improvement and development in the water sector.

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Extended Turkish Abstract
(Genişletilmiş Türkçe Özet)

Türkiye'de Su Kalitesi Yönetiminin Yasal Çerçevesi

Türkiye'de, doğal kaynakların ve insan sağlığının korunmasını kapsayan ilk kanun 1983'te yayınlanan 2872 sayılı Çevre Kanunu'dur. Çevre Kanunu'nun yasal yapısı, uygulama esaslarını belirleyen bir teknik düzenlemeler ve standartlar sisteminden oluşur. Daha sonraki yönetmeliklerin birçoğu Çevre Kanunu'na dayanılarak hazırlanmıştır. Günümüzde Su Kirliliği Kontrol Yönetmeliği (SKKY) (Resmi Gazete No: 25687, 2004) ve Yerüstü Su Kalitesi Yönetmeliği (Resmi Gazete No: 28483, 2012), su kaynaklarının kalitesini korumayı amaçlarken, insan taleplerini karşılamak amacıyla yerüstü ve yeraltı sularının korunmasına yönelik teknik esasları belirlemektedir. Ayrıca büyükşehir belediyeleri Büyükşehir Belediyesi Kanunu (Resmi Gazete No: 25531, 2004) ile su ve atık su tesislerinin en verimli şekilde yönetilmesi için gerekli mevzuatı kendi sınırları içinde belirleme ve uygulama yetkisine sahiptir. Yeraltı suyu kalitesinin izlenmesi ise Yeraltı Sularının Kirlilik ve Bozulmaya Karşı Korunması Hakkında Yönetmelik (Resmi Gazete No: 28257, 2012) uyarınca yapılmaktadır. Bu Yönetmeliğin amacı, iyi yeraltı suyu koşullarının sürdürülmesi, kirlenmesinin ve bozulmasının önlenmesi ve yeraltı suyu kalitesinin iyileştirilmesi için gerekli ilkeleri belirlemektir. Ulusal yönetmeliklere ek olarak Su Çerçeve Direktifi (SÇD) (Avrupa Komisyonu, 2000) ve onun alt direktifi Deniz Stratejisi Çerçeve Direktifi (MSFD) (Avrupa Komisyonu, 2008) tüm Avrupa su kütlelerinde iyi bir çevresel durum elde etmeyi amaçlamaktadır. Türkiye AB aday ülkesi konumunda olduğu için yönetmeliklerini mevcut AB direktifleri çerçevesinde güncellemektedir.

Havza Yönetim Planlarının Hazırlanması, Uygulanması ve Takibi Yönetmeliği (Resmi Gazete No: 28444, 2012) uyarınca Nehir Havzalarını Koruma Eylem Planları (NHKEP'ler) ve Nehir Havzası Yönetim Planları (NHYP'ler) hazırlanmıştır. Gediz, Meriç-Ergene, Büyük Menderes, Konya, Susurluk, Burdur, Küçük Menderes ve Kuzey Ege Havzaları olmak üzere 8 havzanın NHYP'leri Ocak 2021 itibarıyla tamamlanarak yayımlanmıştır. Son yıllarda, Entegre Su Kaynakları Yönetimi ve AB SÇD ilkelerine uyum konusunda Türkiye'de önemli ilerlemeler kaydedilmiştir. Bununla birlikte, kirliliğin azaltılması ve su kaynakları yönetimindeki mevcut eylemler yeterli değildir ve uygulamaların güçlendirilmesi gerekmektedir. Bu amaçla, Nehir Havzası Yönetim Planlarının Hazırlanması, Uygulanması ve Takibi Hakkında Yönetmelik çıkarılmıştır.

Türkiye'nin deniz ve kıyı suları, çeşitli kirlilik kaynakları tarafından baskı altındadır. UNEP / MAP'a göre, deniz kirliliğinin yaklaşık % 80'i kara kökenli faaliyetlerden ve % 20'si gemi kaynaklıdır (UNEP-MAP, 2012). Deniz / kıyı suyu kalitesinin değerlendirilmesi, en iyi yönetim uygulamalarının anahtarıdır. Bu kapsamda Türkiye, kıyıdaş ülkelerle ortak araştırma programları geliştirmiş ve Barselona Sözleşmesi ve Bükreş Sözleşmesi gibi uluslararası sözleşmelere taraf olmuştur. Bölgesel izleme çalışmaları, bu sözleşmeler, protokol ve programlarına göre Akdeniz / Ege Denizi'nde MEDPOL (Akdeniz Deniz Kirliliği Değerlendirme ve Kontrol Programı), Karadeniz'de BSIMAP (Karadeniz Entegre İzleme ve Değerlendirme Programı) ve MEMPHIS (Marmara Denizi Havzası için Çevresel Master Planı ve Yatırım Stratejisi) 2011 yılına kadar yürürlüğe girmiştir. Bunların yanı sıra 2000'lerden sonra ekosistem tabanlı yönetimi ve entegre izleme yaklaşımlarını vurgulayan SÇD ve MSFD yürürlüğe girmiştir. Türkiye'de benimsenen bu yeni izleme stratejileri yaklaşımları ile Çevre ve Şehircilik Bakanlığı tarafından "Entegre Deniz Kirliliği İzleme" programı kapsamında deniz kirliliği izlenmeye başlanmıştır (Olgun vd., 2016). Şimdiye kadar, iki izleme dönemi (2014-2016 ve 2017-2019) başarıyla tamamlanmış, 2019-2022 dönemini kapsayacak olan Türkiye deniz / kıyı suyu kalitesi sınıflandırmasına ilişkin son araştırmalar devam etmektedir.

Ülkemizde yüzme suyu kalitesi, 2019 yılında revize edilmiş olan Yüzme Suyu Kalitesinin Yönetimine Dair Yönetmelik (Resmi Gazete No: 30899, 2019) ile izlenmektedir. Sağlık Bakanlığı tarafından yüzme suyu kalitesinin izlendiği 1500'den fazla kıyı ve iç sular bulunmaktadır. Ek olarak, 2021 yılında toplam 547 plaj, marina, bireysel yat ve turizm teknesi Mavi Bayrak ile ödüllendirilmiştir (Mavi Bayrak, 2021).

Atık su kirliliği ve kontrolüne yönelik ise standartlara uygun kanalizasyon tesislerinin inşası, 1960'ların sonunda mülga İller Bankası tarafından başlatılmış, günümüzde İLBANK tarafından inşa edilmektedir. Günümüzde atık su yönetimi ise kentsel ve bazı endüstriyel atık suların toplanması, arıtılması ve deşarjı, atık su deşarjının izlenmesi, raporlanması ve denetlenmesi ile ilgili teknik ve idari ilkelerin yer aldığı Kentsel Atık su Arıtma Yönetmeliğine göre yapılmaktadır (Resmi Gazete No: 26047, 2006).

Sonuç olarak, bazı sorunlar AB Direktiflerinin uygulanmasını zorlaştırmaktadır. Mevzuat, kurumsal yapı ve planlamada öne çıkan hususlar şu şekilde açıklanabilir:

• Mevzuat: Sıkı standartlar içeren kapsamlı bir dizi yönetmelik mevcut olmasına rağmen, yaptırımlar çoğu durumda tatmin edici değildir.

• Kurumsal yapı: Günümüzde suyla ilgili konulardan sorumlu 10'dan fazla kurum (ör. Çevre ve Şehircilik Bakanlığı, Tarım ve Orman Bakanlığı, Sağlık Bakanlığı, Enerji ve Tabii Kaynaklar Bakanlığı, merkezi düzeyde ve Büyükşehir Belediyeleri, yerel düzeyde) bulunmaktadır. Bu kurumlar aktaran görev ve sorumlulukları ile ilgili olarak farklı, bazen örtüşen görevler verilmektedir. Bu durum da su yönetiminde de çeşitli tartışmalara yol açmaktadır. Bu nedenle, yeni Su Kanununun hedeflerinden biri, daha etkin bir yönetimin sağlanması için su sektöründeki mevcut kurumsal parçalı yapının iyileştirilmesi olarak ortaya konmuştur. 29 Mart 2021 tarihinde suyla ilgili tüm kamu kurumları, kamu kurumları, STK'lar ve özel sektör temsilcilerinin katılımıyla "Su Suraşı" düzenlenmiştir ve 21 Ekim 2021 tarihinde bir "Sonuç Bildirgesi" (MoAF, 2021) yayınlanmıştır.

• Planlama: Çok sayıda eylem planı ve yatırım programı birbiri ile örtüşmektedir. Bu doğrultuda Türkiye'deki koşulların kapsamlı bir değerlendirmesine ve uzun vadeli verilere dayalı olarak mevcut/planlanan alanlardaki gerçek ihtiyaçlara göre mevcut mevzuatın yeniden gözden geçirilmesi gerekmektedir. Bu süreçte fizibilite, yaptırımların uygulanması ve ekonomik caydırıcılık öncelikli olmalıdır.

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Research Article

Predicting Climate-induced Groundwater Depletion: A Case Study in Şuhut Alluvial Aquifer

Yeraltı Suyunun İklime-bağlı Azalışının Tahmini: Şuhut Alüvyon Akiferi Örneği

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Abstract

In this study, we examined the potential impact of climate change on the depletion of groundwater levels and storage. To achieve so, we simulated the groundwater flow using the HİDROTÜRK hydrogeological model under the climate change projections considering the RCP4.5 and RCP8.5 scenarios. To estimate the model forcing input (recharge and evapotranspiration) for the hydrogeological model, we used precipitation and temperature outputs from two Global Circulation Models, namely HadGEM2-ES and MPI-ESM-MR. To assess the changes in groundwater level and storage, we applied our experimental design in the Şuhut alluvial aquifer in Akarçay Basin (Turkey). The study revealed that there is not necessarily a substantial difference tracked over the estimated groundwater levels between the RCP4.5 and RCP8.5 scenarios until the end of 2050s. Yet, a significant reduction in the hydraulic head (approximately 114 m) and storage change (-17.25 %) – particularly in the western part of the aquifer – is expected in 2100, according to RCP8.5. This study confirmed that the selected climate model not only leads to the different predictions in the groundwater depletion, yet also results in a different degree of confidence in the model simulations.
Keywords: *Akarçay Basin, climate change impact, global circulation models, groundwater depletion, Şuhut alluvial aquifer*

Öz

Bu çalışmada, iklim değişikliğinin yeraltı suyu seviyesi ve depolanması üzerindeki olası etkisi incelenmiştir. Bu kapsamda, RCP4.5 ve RCP8.5 iklim değişikliği projeksiyonları altında, yeraltı suyu akımı HİDROTÜRK hidrojeoloji modeli kullanılarak simüle edilmiştir. Hidrojeoloji modeline iklim girdilerinin (beslenme ve evapotranspirasyon) tahmini için, iki farklı Küresel Dolaşım Modelinin – HadGEM2-ES ve MPI-ESM-MR – iklim çıktıları (yağış ve sıcaklık) kullanılmıştır. Yeraltı suyu seviyesinde ve depolamasında iklime bağlı değişimin iklim senaryoları gözetilerek değerlendirilmesi amacıyla Akarçay Havzası'ndaki (Türkiye) Şuhut alüvyon akiferinde yeraltı suyu akım modeli kurulmuştur. Çalışma sonucunda, RCP4.5 ve RCP8.5 senaryolarının her ikisine göre, öngörülen yeraltı suyu seviyelerindeki düşüşlerin 2050'nin sonuna kadar birbirinden çok farklı olmayacağı

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ortaya konmuştur. Öte yandan, RCP8.5 senaryosuna göre, bu yüzyılın sonuna kadar akiferdeki hidrolik yük kaybının (yaklaşık 114 m) ve depolamadaki azalmanın (%-17.25) – özellikle akiferin Batı kesiminde – önemli ölçüde olabileceği öngörülmüştür. Çalışma ayrıca, iklim modellerinin seçiminin yalnızca farklı model tahminlerine yol açmadığını, aynı zamanda model simülasyonlarının da farklı güvenirlik derecesine yol açtığı sonucunu desteklemiştir.

Anahtar sözcükler: *Akarçay Havzası, iklim değişikliği etkisi, küresel iklim modelleri, yeraltısuyu azalımı, Şuhut alüvyon akiferi*

Introduction

As a critical component of the water cycle, groundwater is the largest freshwater source – except the water stored as ice – (Bovoloet et al., 2009). For this reason, groundwater resources are not only of great importance to humanity but are also essential to nursing ecosystems. However, they are currently under the threat of climate change. The threat is even more severe in the arid and semi-arid regions around the Mediterranean basin in which many aquifers have already been suffered by water scarcity (Döll & Flörke, 2005; Kundzewicz et al., 2007; Kundzewicz & Döll, 2008) due to the increase in water demand for agricultural, industrial, touristic, and domestic uses (Shamsudduha et al., 2011; Taylor et al., 2013; Wada et al., 2013, 2014; Wisser et al., 2008). Therefore, understanding the climate-induced impacts on the groundwater is vitally important to sustain all the benefits from these valuable water resources.

According to the Intergovernmental Panel on Climate Change (IPCC) in 2014 (Stocker, 2014), the changes in precipitation and temperature have a substantial effect on the hydrological cycle all over the world in the 21st century. As one of the highly vulnerable regions, the Mediterranean region (Southern Europe and Non-European Mediterranean countries including Turkey) will be particularly suffered from the multiple stresses due to climate change (IPCC, 2007, 2014; Cramer et al., 2018). The primary influences of climate change in these countries are the reduction in the total amount of precipitation with the alteration of the spatial and temporal pattern of the rainfall, and the increment in the air temperature. For this reason, these two variables are also key climatic drivers for groundwater resources in such that precipitation is the main source of aquifer recharge, while the temperature mainly controls the evapotranspiration process. Thus, it is essential to assess to what extent the aquifer systems will be affected by climate change over the Mediterranean countries.

Despite the fact that groundwater has a rather slower hydrological response to the climate effects than that of surface water (Holman, 2006; Moseki, 2017), revealing the climate-induced impacts on the aquifers is still a challenging task due to the direct and indirect effects of climatic variables, which have not yet fully understood, (Dettinger & Earman, 2007, 2011; Green et al., 2011; Woldeamlak et al., 2007). To a certain extent, while the altered climate drivers directly impact groundwater recharge, increased water demand indirectly puts severe stress on the groundwater storage. For this reason, there is an essential need to quantify the groundwater response considering the depletion of groundwater level and storage over the vulnerable climate regions to better plan and manage the groundwater resources in the immediate future.

In this context, as the mathematical models provide valuable information about hydro(geo)logical behaviours of aquifer systems under the changing climatological and/or hydrological conditions, they play a key role to mimic groundwater flow. Therefore, the models are either used to increase comprehensive understanding of the system's reality or utilized to predict the hydrological response of the system under the different climate projections by delineating the hydrological behaviour of the system of interest.

Regarding the climate projections, the Representative Concentration Pathways (RCPs) are developed to examine potential effects and responses of climate change (Moss et al., 2010; van Vuuren et al., 2011). In line with this, the climatic conditions under the projected time-period(s) are described as climate scenarios based on four different greenhouse gas concentration curves, each of which defines rather different climatic conditions, depending on the volume of greenhouse gases emitted in future years. To illustrate, while RCP2.6 and RCP8.5 represent the climate scenario with the lowest and highest greenhouse gas emissions respectively, the RCP4.5 and RCP6.0 scenarios focus on the intermediate stabilization (Petpongpan et al., 2020; Riahi et al., 2011).

The Global Circulation Models (GCMs) – also known as Global Climate Models – are considered as the most reliable tools to obtain the climate indicators (Dragoni & Sukhija, 2008; Kattenberg, 1996; Parry et al., 2007) while numerically simulating the potential changes in the climate based on the boundary conditions (McGuffie & Henderson-Sellers, 2014). On the ground of this, the selection of a plausible future climate scenario by the GCMs is essential. However, since the different models have their strengths in predicting the system reality to capture the non-identical aspects of the system, the predictions from different climate models principally differ from one another. For instance, some models in GCMs anticipate the drier and warmer climate conditions, whereas the others comparatively provide the wetter and colder (Fajardo et al., 2020), thus resulting in prediction uncertainty in model results (Her et al., 2016; Kaczmarska et al., 2018; Lehner et al., 2019; Pour et al., 2020; Salman et al., 2020; Surfleet et al., 2012). From this point of view, considering the climate predictions from a single climate model is not necessarily the plausible option as it includes a certain degree of uncertainty. Therefore, since the climate projections are predominantly dependent on which GCMs' climate scenarios are considered, it is of great importance to examine the predictions of different climate models for any hydro(geo)logical model experiment to reveal the potential uncertainties sourcing from the GCMs' outputs.

To address the impact of climate change on aquifer systems, this paper examines climate-induced groundwater depletion by predicting the potential decline in groundwater level and storage. To achieve so, we used the climate outputs (precipitation and temperature) of two GCMs considering the RCP4.5 and RCP8.5 climate scenarios to utilize these variables as the hydrogeological model forcing input. As one of the vulnerable groundwater sources to the impacts of climate change due to the decrease in the precipitation amount and increased temperature, the Şuhut groundwater body in Akarçay Basin (Turkey) was selected as a case area. By applying our experimental design into the case area, we aim to *(i)* predict the spatiotemporal variability of the groundwater level over the projected time-period (2021-2100), *(ii)* comparatively evaluate the groundwater depletion considering the two scenarios of two GCMs, and *(iii)* assess the groundwater response to the climate scenarios of each climate models, thereby revealing the model prediction uncertainty.

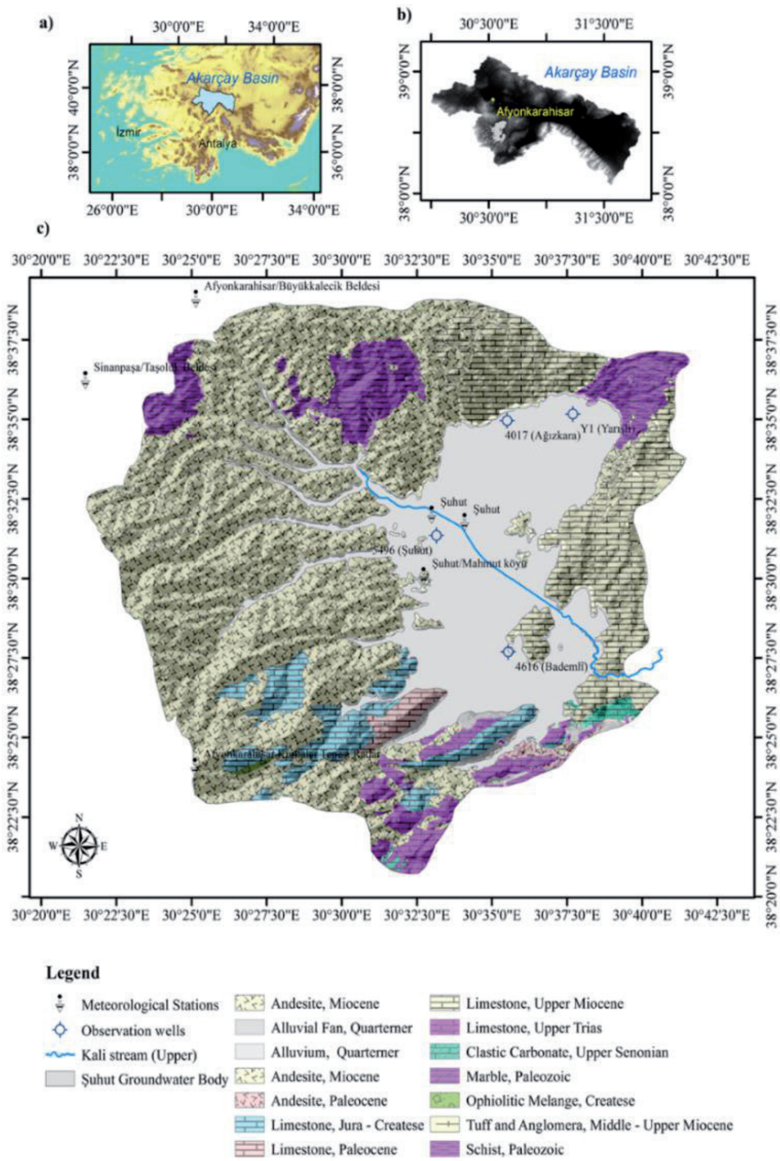
Methodology

Study Area

The Akarçay Basin is located between Central Anatolia, Aegean, and Mediterranean Regions as a closed watershed [Figure 1(a)]. The basin is one of the susceptible watersheds to climate change impact in Turkey (Önder & Önder, 2007) in such that the basin will receive %17-20 less amount of rainfall by 2100 as compared to the reference period for climate projections (1971-2000), while the expected increase in the temperature ranges from 1.5ºC to 4ºC, on average, by the end of the century (General Directorate of Water Management [GDWM], 2015(a), 2015(b), 2016). Furthermore, due to the increased water demand and hydrological drought over the basin, Akarçay Basin could face water scarcity in the immediate future (GDWM, 2016; Kale, 2021).

Figure 1

(a) The Location of the Akarçay Basin (Afyon, Turkey), (b) The Location of the Şuhut Groundwater Body in Akarçay Basin, (c) The Geological Outcrop of the Şuhut Sub-basin Accompanying by Şuhut Groundwater Body



Note. Şuhut Groundwater Body in the Akarçay Basin is indicated by the light-grey colour. The geological map is edited after the Regional Directorate of State Hydraulic Works (hereinafter referred to DSİ (Devlet Su İşleri Genel Müdürlüğü) as Turkish acronym).

The Şuhut sub-basin is one of 8 sub-basins in the Akarçay Basin. The Şuhut basin covers an approximate area of 682 km^2 bordered by the Sandıklı and Kumalar mountains from the west [Figure 1(b)]. The basin is characterized by a flat topography with an elevation ranging from 1120 m to 1150 m, which is also known as the Şuhut Plain. The annual average precipitation over the basin is nearly 487 mm, while the average annual actual evapotranspiration (A_{ET}) accounts for 379.9 mm (DSİ, 2013).

A total number of 14 groundwater bodies covering 3677 km^2 is characterized in the Akarçay Basin by GDWM, 2017. The Şuhut groundwater body (hereinafter referred to as Şuhut alluvial aquifer) is a major domestic and irrigation water supply. which makes it more vulnerable groundwater to the climate change impact in the Akarçay Basin.

The area of the Şuhut alluvial aquifer is approximately 155 km^2 [Figure 1(c)]. The regional groundwater flow direction is towards the southeast of Quaternary aged alluvium in which the system may have a hydraulic connection with the Afyon alluvial aquifer (nearly 750 km^2) according to the previous studies by DSİ (2013), GDWM [2015(b)], [2020(a)] and Sargın (2020).

The geological evolution of the Şuhut sub-basin ranges from the beginning of the Palaeozoic era to the Quaternary period. The western side of the area is mostly covered by the volcanic rocks formed in the Neogene during which the high mountains were mainly shaped under the intensive volcanic activities whereas the northern and eastern parts are characterized by the Pliocene limestone [Figure 1(c)] (Tezcan, 2002; Dişli, 2005). Stratigraphically, the Şuhut Plain is characterized by the four main hydrogeological units including alluvium, tuff, limestone, and volcanic (andesite, basalt, trachyandesite) lava (DSİ, 2013). However, the Şuhut aquifer is formed by the Quaternary alluvium and Plio-Quaternary lacustrine sediments mainly comprising of the sandy-gravelly materials, agglomerate, tuff, and Mesozoic limestone (Kuran, 1958; Gülenbay, 1971; DSİ, 2013). Therefore, the alluvial system may not only feed by the lateral interflow over the fractured volcanic tuffites in the western part of the study area, yet also the fractured/karstified (partially) limestones (includes tuff, siltstone, clay) underlying in the northern and eastern part of the area also contributes to the aquifer recharge (Gülenbay, 1971; Dişli, 2005; DSİ, 2013).

Şuhut Stream – also known as Kali Stream – is the main surface water in the Şuhut Plain [Figure 1(c)]. The stream sources from the Kumalar Mountain in the west and discharges throughout a wide alluvial valley in the plain, thereby reaching the Selevir Dam on the eastern part of the basin. While the Kali stream is fed by the

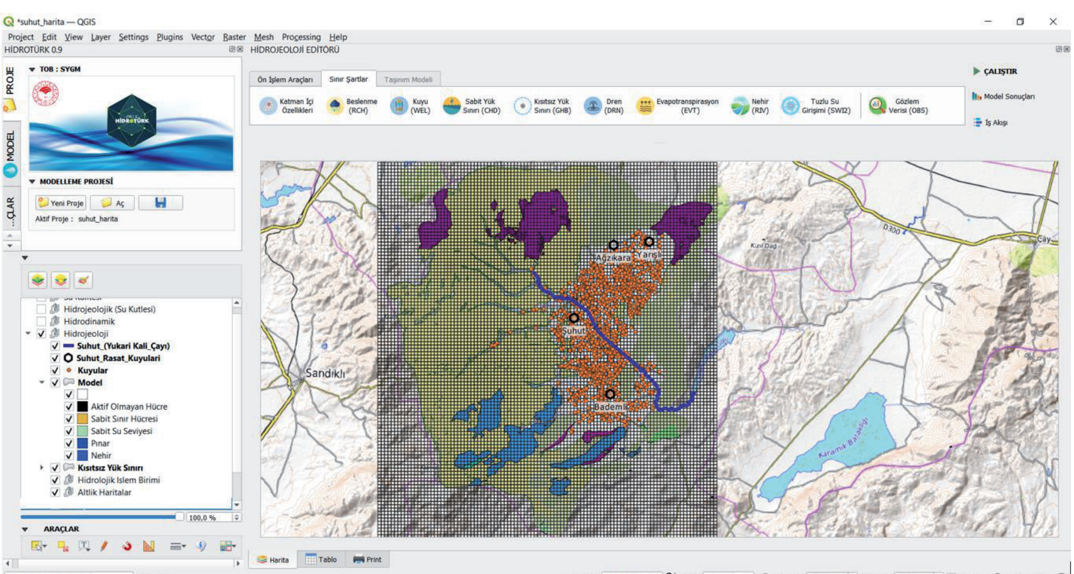
Şuhut alluvial aquifer over the past decades, it currently contributes to the aquifer due to the substantial drawdown in the groundwater level according to the reports by DSİ (2013) and GDWM (2016).

The Model

To simulate the groundwater level, we used the HİDROTÜRK hydrogeological model (Figure 2). HİDROTÜRK model is the first national model platform developed by the Republic of Turkey Ministry of Agriculture and Forestry - General Directorate of Water Management for the sustainable management of the water resources in Turkey [GDWM, 2020(b)]. This platform includes four main model components that each of which simulates the different parts of the hydrological cycle including hydrological, hydrodynamic, and hydrogeological models as well as the water quality and ecological models.

Figure 2

The GUI of the Hydrogeological Model in the HİDROTÜRK Model Platform



The hydrogeological model is one of the MODFLOW-based models that uses a set of Python scripts in the FloPy environment. The MODFLOW-2005 model is served as the core model in the HİDROTÜRK model platform, thereby solving the three-dimensional groundwater flow based on Darcy's Law and the principle of the conversion of mass (Harbaugh & McDonald, 1996, 1988; Harbaugh, 2005).

Since the FloPy Python package is provided by Bakker et al. (2016) without a Graphical User Interface (GUI), the hydrogeological model was constructed in the Geographical Information System (GIS) in the QGIS environment as a plugin to deliver a user-friendly modelling platform (Figure 2). Along with the input-output files to run the core model, the GUI of the model provides 7 main packages including well (WEL), recharge (RCH), evapotranspiration (ETP), river (RIV), constant head boundary (CHD), general head boundary (GHB), and drain (DRN).

Climate Data and Projections

Of all projected climatic scenarios, since RCP4.5 (intermediate level of emission) and RCP8.5 (high level of emission) are two preferred scenarios on a global scale (Riahi et al., 2011; Stocker, 2014), we considered these scenarios for our modelling experiment. To obtain the climate data (precipitation and temperature) we chose the HadGEM2-ES (Hadley Centre Global Environmental Model version 2 Earth System model) developed by the Met Office Hadley Centre (Collins et al., 2011; Jones et al., 2011) and MPI-ESM-MR [Max-Planck-Institute Earth System Model (MPI-ESM) mixed resolution (MR) version] by Giorgetta et al. (2013).

As the HİDROTÜRK hydrogeological model is driven by two hydrometeorological variables (recharge and evapotranspiration), to estimate the input fluxes of the hydrogeological model we used the projected precipitation and temperature data from the HadGEM2-ES and MPI-ESM-MR models considering the RCP4.5 and RCP8.5 scenarios. Both climate variables were processed by 0.1° (10 km x 10 km) resolution using the RegCM4.3.4 regional climate model with the dynamic downscaling method (Turkish State Meteorological Service [MS], 2014; Gürkan et al., 2015; GDWM, 2016; Demircan et al., 2017), and estimated for the 25 watersheds in Turkey over 2015-2100 with a 10-year interval by GDWM (2016).

The Thornthwaite method (Thornthwaite, 1948) was used to obtain the mean total potential evapotranspiration (P_{ET}) values based on the calculated temperatures considering both climate models and projections (Eq. 1).

$$P_{ET} = 16\left(10\frac{T_i}{I}\right)^a \qquad (1)$$

where P_{ET} is the annual potential evapotranspiration (mmy^{-1}), T_i is the average annual temperature (^oC). I is the annual heat index, i.e. the sum of monthly indices i ($i = (T/5)^{1.514}$) while a is the heat index calculated by $0.49239 + 1.792e^{-2}I - 7.71e^{-5}I^2 + 6.75e^{-7}I^3$.

The projected changes in the climate variables and calculated forcing inputs for the numerical model are provided in Table 1. During our experiment, the climate inputs for the hydrogeological model are assumed to be uniformly distributed over the model domain due to the gentle topographic slope (about 1% to 4%) of the Şuhut Plain.

Since the hydrological response of the groundwater to climate impact is rather slower than that of surface water, to obtain climate variables from two GCMs for RCP4.5 and RCP8.5 we assigned relatively longer sub-periods for the simulation period (1997-2100) as compared to those estimated by the 10-year intervals by GDWM, 2016. Then, we obtained the mean annual changes in precipitation (ΔP) and temperature (ΔT) from both GCMs considering the mean values of each variable over the 10 years (Table 1).

Along with the RCP4.5 and RCP8.5 scenarios, we run the hydrogeological model by keeping the values of the aquifer recharge (R) and actual evapotranspiration (A_{ET}) constant over the projection period (2021-2100), which is referred as ‘Baseline’ scenario (see Table 1). Thereafter, we used the Baseline scenario to comparatively examine the groundwater depletion with regard to the climate scenarios.

Model Development

To construct the numerical model, we developed the conceptual model of the Şuhut aquifer, mainly considering the previous hydrological, geological, and hydrogeological studies carried out DSİ. While we used the shapefile of the groundwater body as a hydrogeological model extension area characterized by GDWM (2020(a)), we reconsidered the hydrogeological units to construct the hydrogeological model layers based on the data of 50 boreholes – contain the information of lithology, well depths, borehole geophysics, static and dynamic groundwater levels, yields, and hydraulic conductivity –. We then delineated the hydrogeological characteristics and boundary conditions of the aquifer system based on those borehole data for the numerical model development.

Table 1

The Mean Annual Changes in the Climate Variables and Model Forcing Inputs

Climate Scenario	Projected period	Projected Climate Variables by GCMs				Hydrogeological Model Forcing Inputs			
		HadGEM2-ES		MPI-ESM-MR		HadGEM2-ES		MPI-ESM-MR	
		ΔP (mm)	ΔT (ºC)	ΔP (mm)	ΔT (ºC)	R (mmy^{-1})	A_{ET} (mmy^{-1})	R (mmy^{-1})	A_{ET} (mmy^{-1})
Baseline Scenario	2021-2100					547.5	365.0	547.5	365.0
RCP4.5	2021-2030	30.0	1.8	-50.0	1.0	577.5	423.1	497.5	398.2
	2031-2050	6.0	2.2	-0.5	1.2	553.5	438.0	547.0	404.0
	2051-2100	-20.8	2.9	-25.8	1.7	526.7	460.2	521.7	421.4
RCP8.5	2021-2030	10.0	1.8	-10.0	0.9	557.5	424.7	537.5	394.9
	2031-2050	-2.5	4.8	-11.0	1.4	545	524.3	536.5	411.5
	2051-2100	-50.2	4.5	-77.6	3.2	497.3	514.3	469.9	471.8

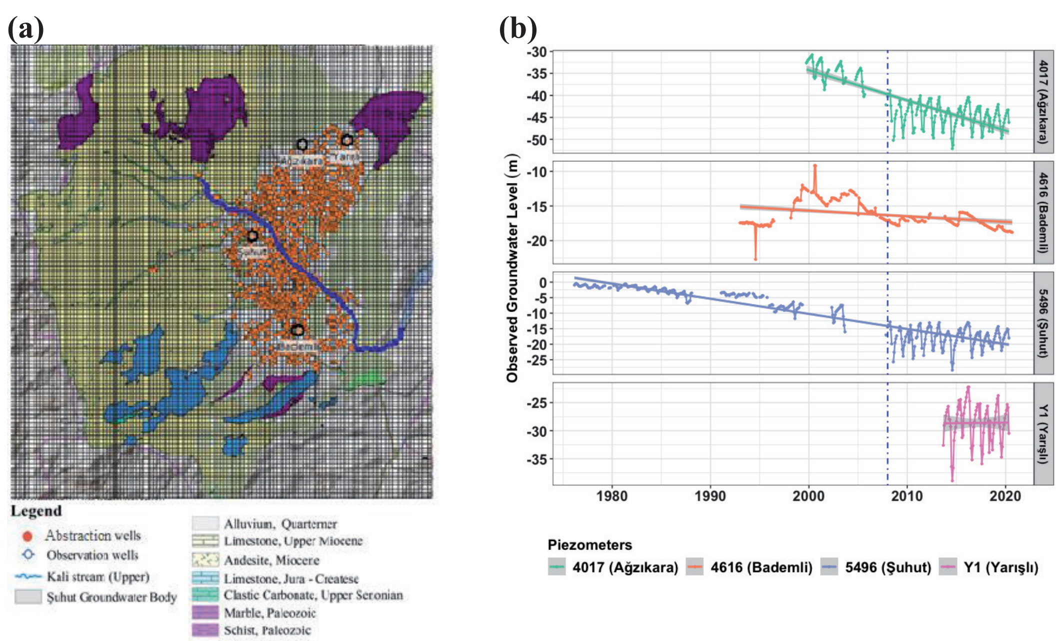
Note. The negative value indicates the decline in each climate variable. The potential aquifer recharge (R) is approximately 548 mmy^{-1}, and average actual evapotranspiration (A_{ET}) accounts for 379.9 mmy^{-1} according to the DSİ, 2013 record

A monitoring network for monthly groundwater level is enabled by 4 piezometers (observation wells) in the Şuhut aquifer by DSİ [Figure 3(a)]. While the Y1 (Yarışlı) and 5496 (Şuhut) piezometers cut the alluvium and limestone units with a depth of 200 m, the 4616 (Bademli) and 4017 (Ağzıkara) piezometers are mainly characterized by the alluvium and volcanic units (DSİ, 2013). In our experimental design, we selected the 4017 (Ağzıkara) well to compare the simulated groundwater levels with the observed values over the reference period (1997-2020). This is mainly because the 4017 (Ağzıkara) well represents one of the vulnerable regions in which the groundwater level experienced a significant drawdown – nearly 35 m over the 20 years (DSİ, 2013) –. Furthermore, the borehole lithology for this well mostly consisted of the alluvial unit.

Figure 3

a) The Spatial Distribution of the 1,281 Abstraction Wells and 4 Piezometers in the Model Area

b) The Monthly Variations of the Observed Groundwater Levels in the Piezometers



Note. The vertical blue dotted line in Figure 3(b) indicates the year-2008 during which the substantial drawdown in the hydraulic heads observed in the observation wells of 4017 (Ağzıkara) and 5496 (Şuhut). The line on each graph indicates the simple linear regression line.

Numerical Model Set-Up

For the numerical model set-up, we defined a one-layer unconfined aquifer system considering the alluvium unit as it mainly characterizes the Şuhut alluvial aquifer. Based on the borehole data, the model layer was delineated by the depth of the hydrogeological unit ranging from 10 m to 300 m (DSİ, 2013). The one-layered model area was discretized into uniform cell dimensions of 50 m x 50 m horizontal resolution considering the hydraulic characteristics of the unconfined layer (hydraulic conductivities and storage characteristics), boundary conditions, and initial hydraulic head. The model top and bottom were defined at 0 m (as topographic surface) and -185 m, respectively. Therefore, the thickness of the aquifer layer was represented by 185 m with -10 m initial head (demonstrates the groundwater level below the topographic surface, referring to the depth of the water table). The initial head for the model layer was defined considering the observed static levels in the abstraction wells [see Figure 3(b)]. As the transmissivity of the aquifer unit varies between 36.3 m^2d^{-1} and 900 m^2d^{-1} (Gülenbay, 1971; Dişli, 2005; DSİ, 2013), the horizontal hydraulic conductivity, K_h was interpolated over the model domain based on the 1,218 wells' data, while we assigned the vertical hydraulic conductivity K_v to ten-times lower than that of K_h values (Domenico & Schwartz, 1998). The specific yield (S_y) of the unconfined aquifer layer was assumed to be uniformly distributed over the model area with an average value of 0.0015 (*dimensionless*) obtained by DSİ (2013).

To assign the stress periods over the model simulation period (1997-2100), we deliberated the main hydro(geo)logical changes in the aquifer system considering the observed water levels in the piezometers [Figure 3(b)]. Therefore, the hydrogeological model was run with a three-year spin-up period (1993-1996) under the steady-state flow condition, thereby reaching a dynamic equilibrium in the modelling system. The historical (1997-2007), reference (2008-2020), and projection periods including the sub-periods of 2021-2030 (near), 2031-2050 (intermediate), and 2051-2100 (future) were set up while considering the substantial depletion in groundwater depth [see Figure 3(b)], thus running under the transient flow conditions.

To simulate the lateral interflow from the Mesozoic limestone underlying down the North and East of the Şuhut Plain, we set this model boundary as GHB. Furthermore, the no-flow condition was considered for the rest of the model domain, mainly assuming the area was mainly covered by a less permeable volcanic unit [see Figure 1(b)]. The WEL was activated by the 997 abstraction wells during the historical period (1997-2007) while a total of 1,281 wells were used during the reference period (2008-2020) and the prediction period (2021-2100).

The model calibration and sensitivity analysis were not performed in our modelling experiment as the hydrogeological model do not include a calibration toolbox. Instead, to increase the model representativeness, we used the piezometers (Figure 3) to comparatively capture the observed hydraulic heads over the reference period (2008-2020) by the simulated ones. Afterward, the post-processing of the model results was visualized using R-Studio (R Core Team, 2021).

To account for the model prediction uncertainty resulting from the selected climate model's outputs, we compared the results of the groundwater level and storage driven by the climate outputs of the two GCMs – HadGEM2-ES and MPI-ESM-MR – over 2021-2100.

Groundwater Depletion

For the sake of revealing the climate-induced changes in the groundwater level and storage, the model simulations were performed under the same boundary conditions throughout 2021-2100, thereby assuming that no further changes will be mentioned in the local water management. Here, our primary aim was to observe the groundwater depletion which only emerges from the climate change impacts. For this reason, we kept the number of groundwater abstraction wells – and the pumping rate in each pumping well – constant.

To quantify the annual groundwater depletion under the climate change projections, we first calculated annual groundwater drawdown (Δh) by

$$\Delta h = h_{sim} - h_i \quad (2)$$

Here, h_{sim} is simulated annual hydraulic head, and h_i is the initial hydraulic head which was defined by -10 m. The annual depletion in water storage (ΔS) is then calculated considering Δh using a similar approach proposed by Healy and Cook (2002) in Eq. 3:

$$\Delta S = S_y \times \Delta h \quad (3)$$

where S_y is the specific yield of the unconfined aquifer (*dimensionless*).
After obtaining the annual changes in water storage under the RCP4.5 and RCP8.5 scenarios, we estimated the relative bias in storage in percent, ΔS (%) by Eq. 3 with

respect to the baseline scenario while assuming that the absence of bias corresponds to the base model (0 %).

$$\Delta S\ (\%) = \frac{\Delta S_{baseline\ scenario} - \Delta S_{climate\ scenario}}{\Delta S_{baseline\ scenario}} \times 100 \qquad (4)$$

where $\Delta S_{climate\ scenario}$ is the annual changes in water storage of the corresponding climate scenario.

Results and Discussion

Climate-induced Depletion of Groundwater Levels

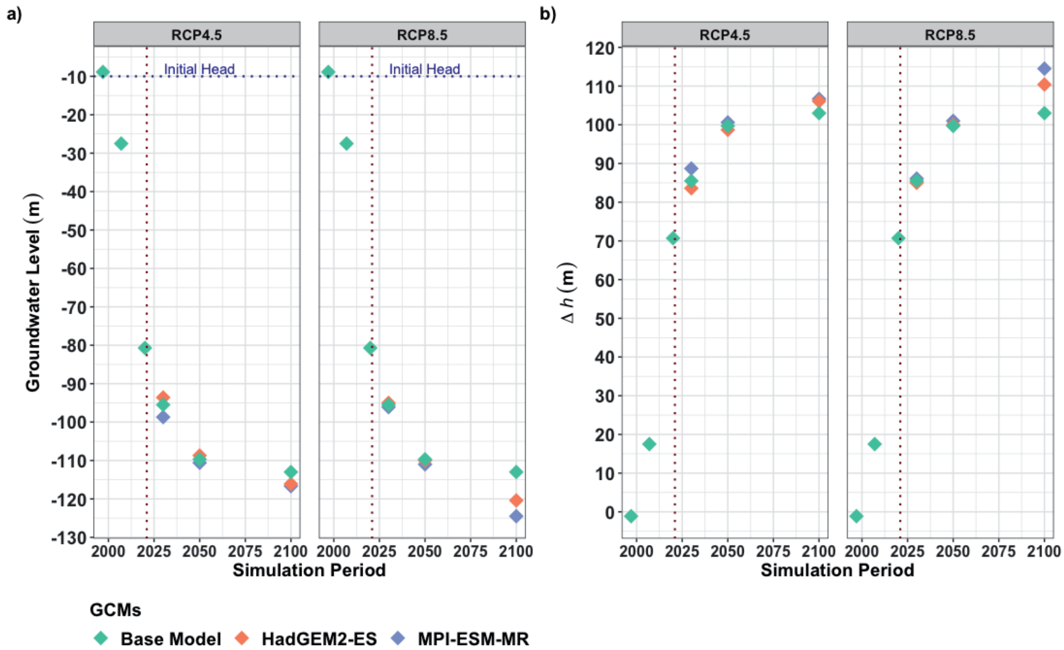
For the comparative evaluation of the climate-induced groundwater depletion over the historical (1997-2020) and projection periods – 2021-2030 (near), 2031-2050 (intermediate), and 2051-2100 (future) –, we obtained the minimum groundwater levels at the last day of each sub-period. Figure 4 demonstrates the simulated groundwater hydraulic heads in the Şuhut alluvial aquifer and the corresponding drawdowns in the groundwater drawdown (Δh) obtained from the HİDROTÜRK hydrogeological model. Overall, the simulation results driven by the climate outputs of the MPI-ESM-MR model provides rather lowered water depths and higher drawdown values than that of the HadGEM2-ES model.

Figure 4(a) confirms that the decline in the groundwater level is inherently dependent upon which climate model's outputs are served as the forcing fluxes during the model conditioning phase. Here, the minimum groundwater level was estimated to -124.5 m with a decrease by more than 10% as compared to the baseline scenario until the end of the century (for RCP8.5 by the MPI-ESM-MR model in 2100), whereas the HadGEM2-ES model led to a less drawdown with an estimated head value of -120.4 m (a -6.55% decrease for the same scenario in 2100). Similarly, the decrease in the groundwater depth is 116.7 m (-2.74%) for the MPI-ESM-MR model under the RCP4.5 scenario, which is a little higher than -116.1 m (-3.26%) by the HadGEM2-ES model. The findings are also supported by the other studies (Döll, 2009; Kurylyk & MacQuarrie, 2013; Pratoomchai et al., 2014), which also state that the model projections vary predominantly depending on the selected GCMs rather than the climate scenarios, only.

Figure 4

(a) The Projected Groundwater Levels Based on the Climate Variables Obtained from the MPI-ESM-MR and HadGEM2-ES Models

(b) The Variations in Groundwater Drawdown (Δh [m]) over the Şuhut Alluvial Aquifer Under the RCP4.5 and RCP8.5 Scenarios



Note. Here, each value indicates the maximum drawdown in the groundwater level estimated by the end of each sub-period over the model simulation. The dashed vertical dark red coloured line separates the historical/references (1997-2020) and projected (2021-2100) periods, thereby indicating past and future hydrogeological flow conditions, respectively.

As for the influences of climate change on the groundwater hydrological response, Figure 4(b) reveals the fact that a remarkable depletion in the groundwater level was already observed during the reference period (2008-2020) in which the additional 284 abstraction wells were drilled in the aquifer system (as it was conditioned in the model area). As a result, the maximum drawdown was represented by an approximate value of 53 m for the baseline climate scenario over the reference period (2008-2020). Even worse, the groundwater levels will be experienced by a dramatic drawdown by reaching the drawdown value of more than 100 m for both scenarios at the end of this century. Thus, the differences in the head losses for both climate models gradually become dramatic – in particular for RCP8.5. Yet, here, the RCP8.5 scenario for the MPI-ESM-MR model still represents the highest value for the estimated drawdown by 114 m.

Spatiotemporal variations of the groundwater level over the end of each sub-period – 2020, 2030, 2050, and 2100 – are indicated in Figure 5. Here, we only provide the numerical results driven by the climate outputs of the MPI-ESM-MR model under the two climate projections since it leads to a relatively sharp decline in the groundwater depth [Figure 4 (a)]. In general, the western part of the Şuhut Alluvial aquifer – in the vicinity of the Şuhut district (see Figure 2) – is the most vulnerable area to climate-induced effects. Over this region, the water depth ranges from -75 m to -85 m for RCP4.5 in 2100 while it varies between -82 m and -88 m for RCP8.5, meaning that the total drawdown is expected to be around 75-88 m at the end of the century. Yet, above all, the depletion in the groundwater level could also become more dramatic over the western boundary of the aquifer in which the considerable decrease in the hydraulic head occurs.

Groundwater Response to Climate Change

The annual variability of the simulated groundwater levels over the simulation period (2008-2100) and the uncertainty bound during the projection time (2021-2100) are provided in Figure 6. Overall, the annual groundwater levels demonstrated a decreasing exponential behaviour for the baseline scenario, thus flattening out between -110 m and -113 m during 2051-2100, whereas the decline in the water level under the climate-change projections slightly deviated from this exponential curve, thus characterized by two inflection points around the years of 2025 and 2050.

In agreement with the decreasing exponential behaviour of the groundwater levels, the water table starts to respond rather slowly to the climate-induced changes in both numerical results in Figure 6 (a). Here, regardless of which climate model's outputs are used as forcing fluxes for the hydrogeological model, the decline in the water level is consistently retarded by the advancing time. More specifically, while the first slowing downward trend of the depth of groundwater levels will be observed between the near (2021-2030) and intermediate (2031-2050) periods, the slowest one is tracked over the future period (2051-2100). From this point of view, this hydrological behaviour in the aquifer system could be marked by the inflection points on the drawdown curves – as indicated by Figure 6 (a) with a dashed dark red line –. This inflection on the drawdown curve could point out at which level groundwater table reaches a threshold value – called here as critical water depth indicated by red dashed line – where the water depletion may not be easily influenced by the dramatic changes over the hydro-climatological conditions. However, it should be noted that this inference is only valid under the assumption that no further changes will be available for the aquifer hydrogeological conditions and/or the local water management.

To get an idea of how sensitive the simulated annual groundwater levels are to the climate scenarios of each GCMs, we used the box-and-whisker plots in Figure 6 (b), thereby evaluating the potential uncertainty sourcing from the selected GCMs. In general, the InterQuartile Range (IQR) of the groundwater variations under the RCP8.5 scenario for both climate models was quite large, indicating that the obtained values were rather sensitive to RCP8.5, thus resulting in greater model prediction uncertainty. Interestingly, the confinement in the predicted values for the RCP4.5 is better than that of RCP8.5 for both models, ensuring a narrower IQR with an average value of -105 m for the HADGEM2-ES model. Therefore, the range of the annual groundwater level reveals that the selection of the climate models - and climate projections - not only leads to the different simulations, but it also defines the level of confidence in the model predictions.

Climate-induced Depletion in Groundwater Storage

The temporal anomalies in the annual water influx are provided in Figure 7. Overall, the annual variations in the input fluxes for both climate models demonstrated a continuous downward trend due to the expectation of the deficit in the precipitation amount in the Şuhut basin (see Table 1). The only exception, here, is the increased influx for the near future (2021-2030) for the RCP4.5 scenario of the MPI-ESM-MR model. This local increment in the influx can be explained by the relative increase in the precipitation amount (+ 30 mm) in comparison to the reference model simulation period (2008-2020).

Figure 5

The Spatiotemporal Distribution of the Depth of Groundwater Levels over the Model Domain

(a) Spatiotemporal Distribution of The Depth of Groundwater Level for the RCP4.5 Scenario

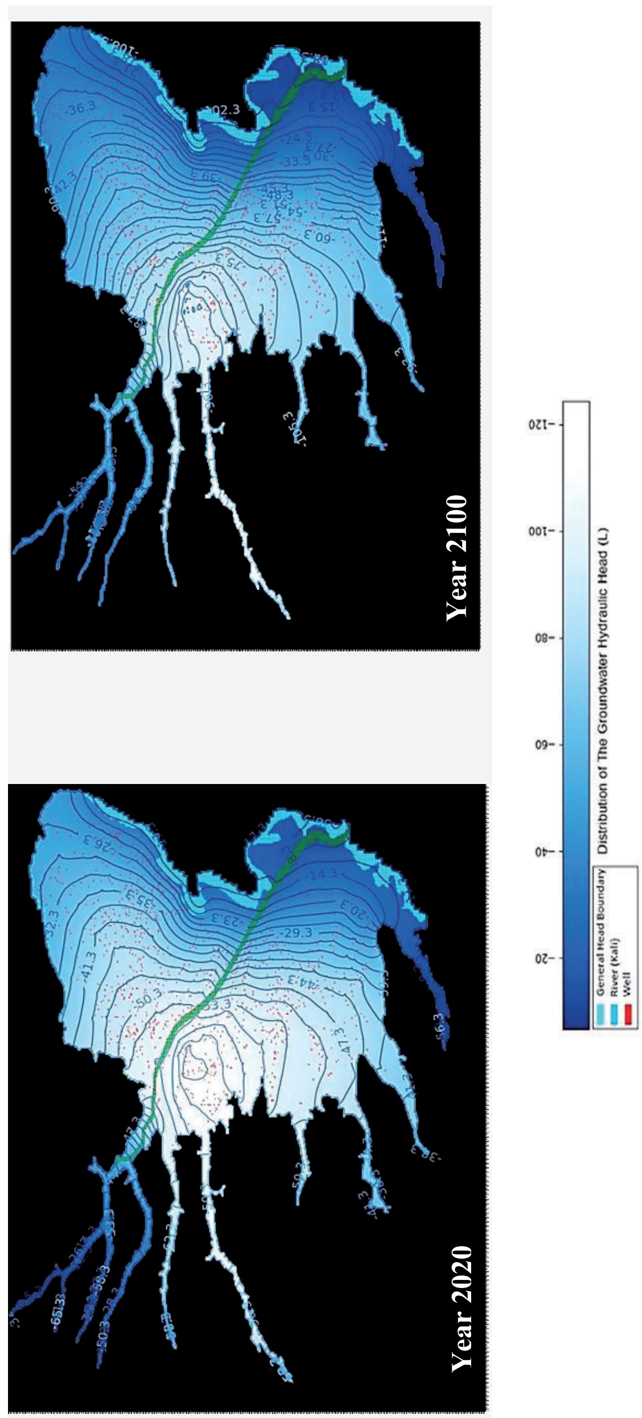
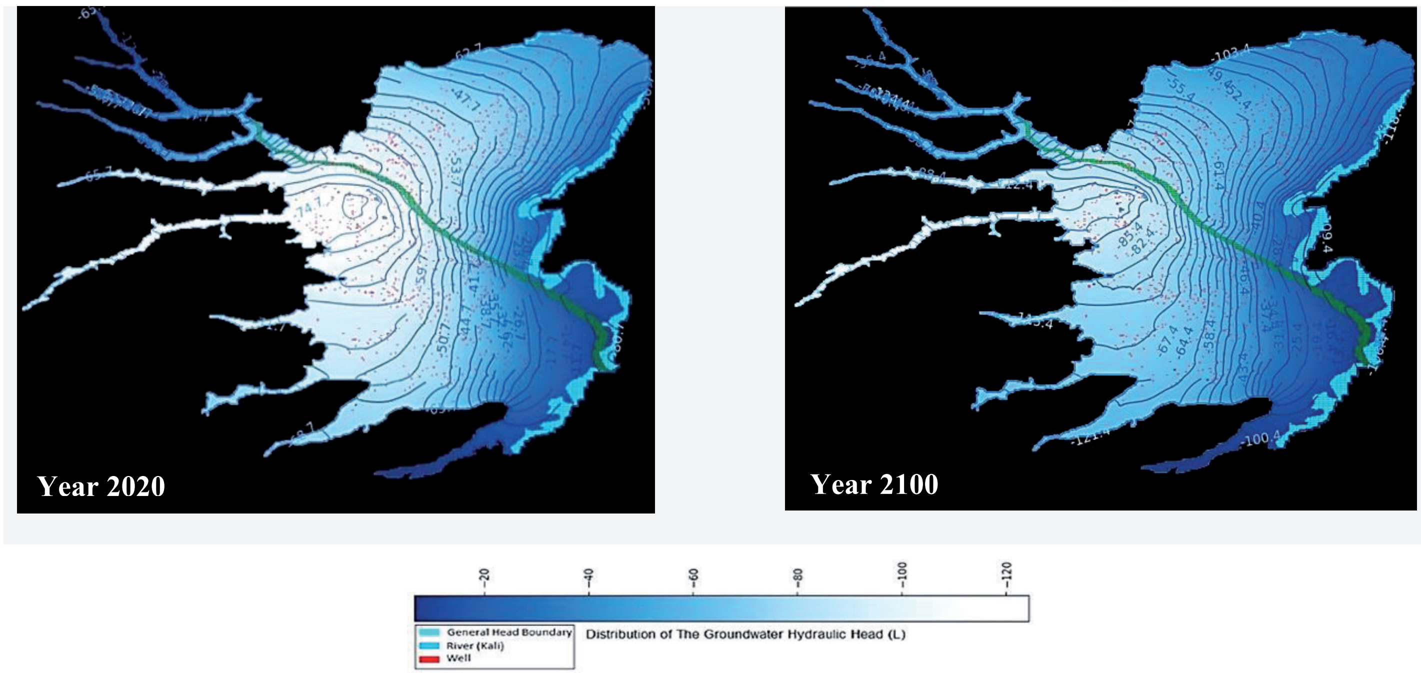


Figure 5

Continued

(b) Spatiotemporal Distribution of the Depth of Groundwater Level for the RCP8.5 Scenario



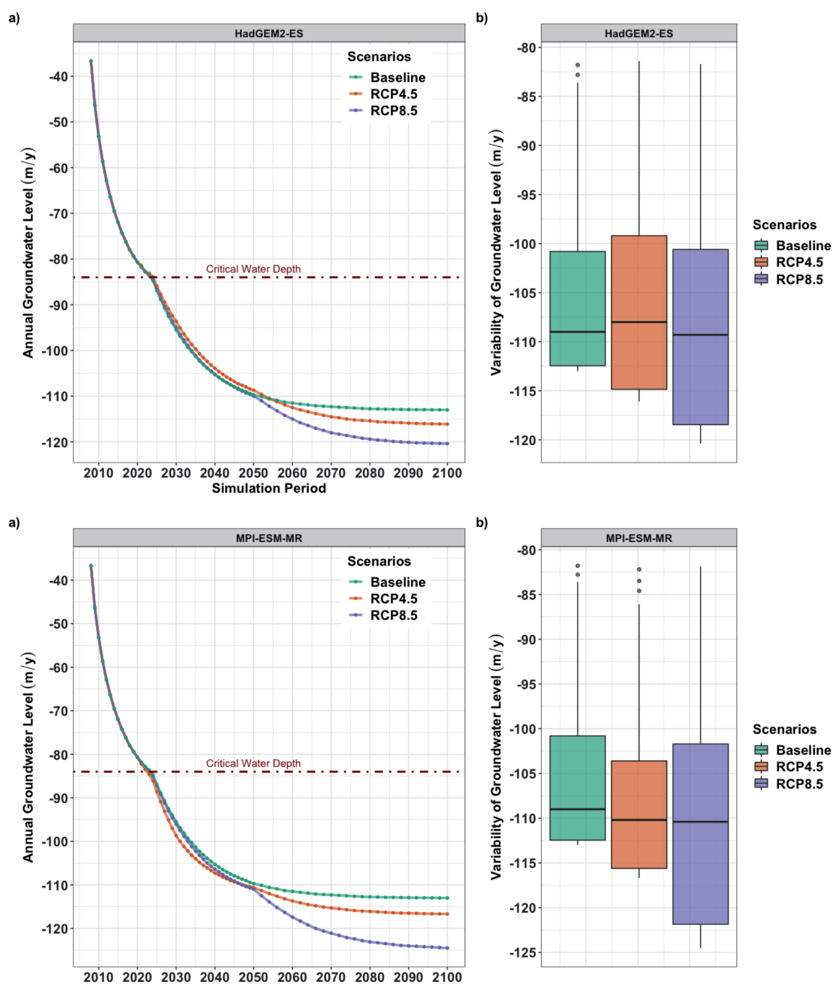
Note. Here, the simulations driven by climate outputs from the MPI-ESM-MR model are demonstrated.

Figure 6

The Model Simulations from HADGEM2-ES and MPI-ESM-MR over the period of 2008-2100

(a) The Annual Variability of the Simulated Groundwater Levels

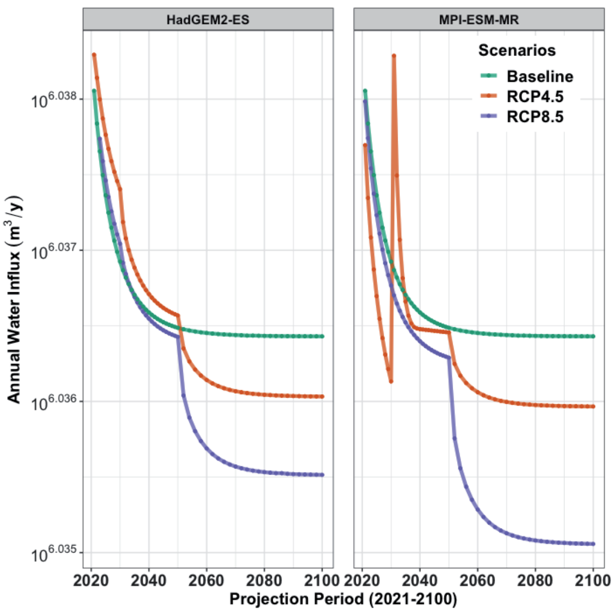
(b) The Box-and-Whisker Plots of the Simulated Groundwater Levels



Note. The horizontal dark red coloured line indicates ‘critical water depth’ (around -84 m for both model simulations) in which the drawdown of groundwater level starts to respond slowly to the variations in climate as compared to the reference period (2008-2020).

Figure 7

The Annual Anomalies of the Simulated Water Influx over the Projection Period (2021-2020).



The decreasing exponential behaviour of the water influx – it is also tracked over the simulation of groundwater level in Figure 6 – can also be seen in Figure 7. Here, the annual fluctuations in the water influx were characterized by a slowing downward trend after the year-2050 for both scenarios of the two climate models. Yet, the obtained annual values by the MPI-ESM-MR model still exhibited a strong decline, particularly for the RCP8.5 scenario. Therefore, the result verifies the fact that the hydrogeological model uses the precipitation input as a principal driver for the simulation of the hydraulic head even though the same exponential behaviour is not directly observed over the annual changes in the groundwater levels under the RCP4.5 scenario for the MPI-ESM-MR model.

Figure 8 demonstrates the temporal variability in the annual changes in water storage (*ΔS*, %) based on the climate change projections throughout the model projection period (2021-2100). Overall, *ΔS* varied primarily dependent on the climate models' outputs served as forcing inputs in the groundwater model. Here, *ΔS* was predominantly represented by the negative values in both cases, while the only exception is the positive value of *ΔS* accounted for by +2.85% (+30 mm for RCP4.5) and + 0.75 % (+10 mm for RCP8.5) for the HadGEM2-ES model.

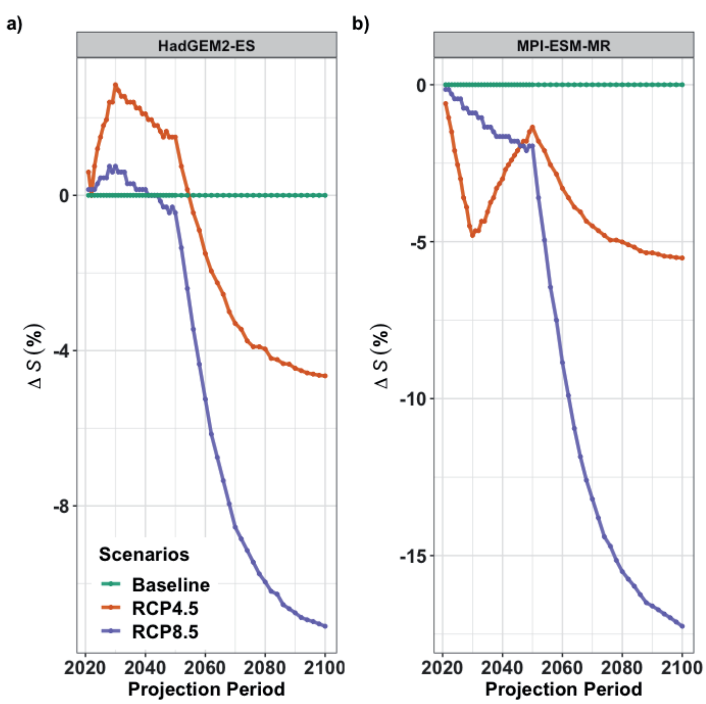
Figure 8 (b) also reveals that the storage depletion will be worsening by the end of 2100 according to the RCP8.5 scenario of the MPI-ESM-MR model in which the projected changes show a large range in the estimated values from -0.9 % (-10 mm decrease in precipitation amount over 2021-2030) to -17.25 % (-78 mm decline in precipitation amount over 2050-2100). Therefore, incompatible with the climate inputs for the hydrogeological model – decreased precipitation amount and increased temperature (see Table 1) –, the relative changes in the water storage based on the baseline scenario confirm that the climate-induced effect – especially the precipitation input – is of importance for the prediction of storage depletion.

Figure 8

The Temporal Variability of the ΔS (%) under the RCP4.5 and RCP8.5 for 2008-2100

(a) The Simulations Driven by the MPI-ESM-MR Model

(b) The Simulations Driven by the HadGEM2-ES Model



Note. Here, the baseline scenario corresponds to the absence of bias (0 %).

Conclusions

The study examines to what extent climate change influences groundwater depletion under the RCP4.5 and RCP8.5 scenarios of two GCMs. By implementing our experimental design into the Şuhut alluvial aquifer, we projected how groundwater level and storage vary over the projection time-period (2021-2100), thereby revealing the variability of the model results to the climate inputs obtained by two GCMs. The key findings from our research are as follows:

- A wide range of the projected annual groundwater level and storage changes reveals that the selection of climate models not only leads to different model predictions, yet also results in a different degree of confidence in the model simulations.

- The hydrological response of the groundwater depth over the model simulation period (2008-2100) is characterized by a decreasing exponential behaviour for the baseline scenario, whereas the slight deviations are observed under the climate-change projections for both GCMs. This hydrological response may indicate critical water depths in which the dramatic changes in hydrological and/or climatological conditions would not easily influence the aquifer hydrological conditions.

- As for the Şuhut alluvial aquifer, the climate change impact would have significant effects on the reduction of the groundwater depth and storage, especially in the western part of the aquifer system where the groundwater abstraction rate is rather higher. Furthermore, the substantial decline in the groundwater level is predicted by the near-future period (2021-2030), while the depletion in the water storage demonstrates a rather different response as compared to the groundwater depth in such that the substantial decline in the storage is projected throughout 2051-2100.

- The assessment of the model prediction uncertainty is not only essential to reliably interpret the model simulations under the climate change projections, yet it is also of importance to reveal the model prediction uncertainty caused by the climate outputs from the different GCMs.

The significance of this research is to assess the potential impact of climate change on groundwater level and storage while examining the model prediction uncertainty sourcing only from the selection of the climate models and their outputs –merely considering the precipitation and temperature variables–. However, it is worth noting that there are some limitations during the model experiment: (1) any further changes in the local water management for the experimental design were not considered after the reference period (2008-2020) to reveal the climate impact on the depletion of groundwater level and storage. For this reason, the aquifer boundary conditions were kept as identical as the reference period over the projection period (2021-2100). However, this assumption is not valid since climate change undoubtedly alters the hydrological and hydrogeological boundaries, as well. (2) the depletion of surface waters, the water transfers in/out the basin, and increased groundwater abstraction rates would be some reasonable examples under the changing hydrometeorological conditions. Henceforth, along with the prediction uncertainty coming from the selection of GCMs, it is important to consider that the complex hydro(geo)logical response of the aquifers to the variations of the hydrological and climatological conditions could also result in the model prediction uncertainty.

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Extended Turkish Abstract
(Genişletilmiş Türkçe Özet)

Yeraltı Suyunun İklime Bağlı Azalışının Tahmini: Şuhut Alüvyon Akiferi Örneği

Su döngüsünün önemli ve kritik bileşenlerinden birisi şüphesiz yeraltı suyudur. Yeraltı suyu sadece insanlık için önemli değil, aynı zamanda ekosistemleri sürdürmek için de oldukça önemlidir. Ancak bu değerli hidrolojik sistemler iklim değişikliğinin tehdidi altındadır. Bu durum, özellikle Akdeniz bölgesi etrafındaki yarı kurak iklim koşullarına sahip bölgelerde bir tehdit unsuru haline gelmiştir. Bu bölgelerdeki birçok akifer, diğer pek çok su kaynağı gibi (dereler, akarsular, goller gibi) tarımsal sulama ve endüstriyel turizm sektörünün artan talebi nedeniyle hali hazırda zarar görmüştür. Bu nedenle, iklim kaynaklı etkilerin yeraltı suları üzerindeki doğrudan ve dolaylı etkilerinin ortaya çıkarılması zorunlu bir görevdir.

Değişen iklim koşulları yeraltı suyunun beslenmesini doğrudan etkilerken, artan su talebi dolaylı olarak yeraltı suyu depolaması üzerinde ciddi stres yaratmaktadır. Bu nedenle, yeraltısuyu kaynaklarının yakın gelecekte daha iyi planlanması ve yönetilmesi için, yeraltısuyu seviyesinde ve depolanmasındaki değişim dikkate alınarak yeraltı suyunun tepkisinin sayısal tahminine ihtiyaç duyulmaktadır. Bu bağlamda matematiksel modeller, değişen iklim ve/veya hidrolojik koşullar altında akifer sistemlerin hidro(jeo)lojik davranışları hakkında önemli bilgiler sağlamakta olup, yeraltı suyu akım ve davranış tahmininde önemli rol oynamaktadır.

İklim projeksiyonları ile ilgili olarak, iklim değişikliğinin potansiyel etkilerini ve tepkilerini incelemek için Temsili Konsantrasyon Yolları (RCP'ler) geliştirilmiştir. Buna temelde, öngörülen zaman dilimlerindeki iklim koşulları, gelecekte salınan sera gazlarının hacmine bağlı olarak açıklanmış ve her biri birbirinden oldukça farklı iklim koşullarını tanımlayan dört farklı sera gazı konsantrasyon eğrisine dayanan iklim senaryoları tanımlanmıştır. Örneğin, RCP2.6 ve RCP8.5 sırasıyla en düşük ve en yüksek sera gazı emisyonlarına sahip iklim senaryosunu temsil ederken, RCP4.5 ve RCP6.0 senaryoları ara stabilizasyona odaklanmaktadır.

Küresel İklim Modelleri olarak da bilinen Küresel Dolaşım Modelleri (GCM'ler), iklim göstergelerini elde etmek için kullanılan en güvenilir araçlar olarak kabul edilmektedir. Bu modeller ile, iklimdeki potansiyel değişiklikler farklı sınır koşullarına bağlı olarak simüle edilmektedir. Bu nedenle, farklı iklim modellerinin gerçek sistemin farklı özelliklerini tahmin etme konusunda birbirlerinden farklı güçlü yanları bulunmaktadır. Ancak, farklı iklim modellerinden elde edilen iklim tahminleri birbirinden farklı olmaktadır. Örneğin, GCM’lerdeki bazı iklim modelleri daha sıcak ve kuru iklim koşullarının öngörüsünü yaparken, bir diğeri nispeten daha soğuk ve yağışlı koşulları tahmin etmektedir. Bu nedenle, iklim modelleri sonuçlarında bir tahmin belirsizliği her zaman söz konusudur. Bu açıdan bakıldığında, iklim tahminlerini tek bir iklim modelinden ele almak, belirli bir derecede belirsizlik içerdiğinden, bir hidrolojik ya da hidrojeolojik çalışmada her zaman makul bir seçenek değildir. Bu nedenle, iklim projeksiyonları ağırlıklı olarak seçilen iklim modeline ve iklim senaryolarına bağlı olduğundan hareketle, GCM'lerden kaynaklanan olası belirsizlikleri ortaya çıkarmak amacıyla herhangi bir hidro(jeo)lojik modelleme çalışmasında farklı iklim modellerinin tahminlerini karşılaştırmalı olarak değerlendirmek önem taşımaktadır.

Yeraltısuyu kaynakları üzerinde iklim değişikliği etkilerini araştırmak için yapılan bu çalışmada, yeraltısuyu seviyesindeki ve depolanmasındaki olası azalışın tahmini yapılmıştır. Çalışmanın temel amacı şu şekilde özetlenebilir: *(i)* hidrojeolojik model simülasyonunun zaman dilimi boyunca (2021-2100) yeraltı suyun seviyesinin/derinliğinin konuma ve zamana bağlı değişiminin tahmin edilmesi, *(ii)* iki farklı Küresel Dolaşım Modelinin iki farklı iklim senaryosu gözetilerek yeraltısuyu tükenmesinin karşılaştırmalı olarak değerlendirilmesi, ve *(iii)* her bir iklim senaryolarına karşılık gelen yeraltısuyu azalışının değerlendirilerek, iklim modellerine dayalı model tahmin belirsizliğinin ortaya çıkarılması.

Çalışma alanı olarak, iklim değişikliği ile birlikte yağış miktarındaki azalmaya ve artan sıcaklığa karşı duyarlı bir yeraltı suyu kaynağı olan Akarçay Havzası'ndaki (Türkiye) Şuhut alüvyon akiferi seçilmiştir. Akiferin iklim etkilerine bağlı hidro(jeo)lojik davranışını tanımlamak ve ileriye dönük sayısal tahminlerde bulunmak amacıyla yeraltı suyu akım modeli kurulumu için HİDROTÜRK hidrojeoloji modeli kullanılmıştır. Hidrojeoloji modeline iklim girdilerinin (beslenme ve buharlaşma) tahmini için, iki farklı Küresel Dolaşım Modelinin (GCM) – HadGEM2-ES ve MPI-ESM-MR – RCP4.5 ve RCP8.5 senaryolarına karşılık gelen iklim çıktıları (yağış ve sıcaklık) kullanılmıştır. Yeraltı suyu akım modeli, 1997-2100 yıllarını arasında RCP4.5 (ara emisyon seviyesi) ve RCP8.5 (yüksek emisyon seviyesi) iklim senaryoları gözetilerek çalıştırılmış ve akiferdeki hidrolik yük dağılımı (yeraltı suyu seviyesi) simüle edilmiştir. Buna ek olarak, farklı iklim modelinin hidrojeolojik model sonuçlarında yarattığı tahmin belirsizliğinin ortaya çıkarılması amacıyla, HadGEM2-ES ve MPI-ESM-MR iklim modelleri ve bu modellerin ilgili iklim senaryolarına (RCP4.5 ve RCP8.5) göre yeraltı suyu seviyesi ve depolamasındaki değişim karşılaştırmalı olarak analizi gerçekleştirilmiştir.

Çalışma sonucunda, RCP4.5 ve RCP8.5 senaryolarının her ikisine göre, 2050'nin sonuna kadar yeraltı suyu seviyesindeki düşümlerin birbirinden çok farklı olmayacağı görülmüştür. Ancak RCP8.5 senaryosuna göre, bu yüzyılın sonuna kadar, özellikle yeraltı suyu pompaj oranının fazla olduğu akiferin batı kesiminde, oldukça yüksek bir hidrolik yük düşümüne (yaklaşık 114 m) ve depolama kaybına (%-17.25) neden olabileceği tahmin edilmiştir. Buradan hareketle, iklim değişikliği etkisinin özellikle Şuhut alüvyon akiferinin batı kesiminde önemli miktarda seviye düşümlerine ve depolama değişimine neden etkili olabileceği öngörülmüştür.

Sonuçlarımız, yeraltı suyunda öngörülen tükenmenin, tercih edilen küresel iklim modeli çıktılarına doğrudan ve büyük ölçüde bağlı olduğunu ortaya koymakla birlikte, yeraltı suyu seviyesinin kritik bir derinliğe ulaşması durumunda, akifer sisteminin iklim değişikliği etkilerine daha yavaş yanıt verebileceğini göstermektedir.

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