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Contents

Aims and Scope

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Collation Article

Assessment and Comparison of Commonly Used Eutrophication Indexes

Yaygın olarak Kullanılan Ötrofikasyon Indekslerinin Değerlendirilmesi ve Karşılaştırılması

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Abstract

Eutrophication is a phenomenon that causes the degradation of water quality which results in negative impacts on living and non-living environment of any water body. Therefore, monitoring, evaluation and classification of water quality are the utmost important issue for policy makers and regional institutions/organizations in order to assess the quality and sustainable use of water bodies and to take appropriate measures. Regional conventions such as the Helsinki Convention, Oslo-Paris Convention, Barcelona Convention – Programme for the Assessment and Control of Marine Pollution in the Mediterranean, Strategic Action Programme and Integrated Coastal Zone Management as well as legislative instruments such as the Water Framework Directive, Marine Strategy Framework Directive, Urban Wastewater Treatment Directive and Nitrates Directive in Europe; the Clean Water Act, Water Quality Act, National Environmental Policy and Coastal Zone Management Act issued by the United States Environmental Protection Agency addresses the importance of monitoring for eutrophication. The eutrophication indexes have been developed in line with regional requirements using specific data sets of state variables and parameters, representative of a location. In this study, we reviewed commonly used eutrophication indexes within the areas applied, parameters, methods and classification scales and presented the trophic status equivalences between the indexes. This study aimed to provide a comprehensive examination of eutrophication indexes to researchers in trophic status assessment of any water body.

Keywords: *Trophic status classification, eutrophication indexes, water quality management*

Öz

Ötrofikasyon su kütlesindeki canlı ve cansız çevre üzerinde olumsuz etkilere yol açarak su kalitesinde bozulmalara sebep olan bir olgudur. Bu nedenle, su kalitesinin izlenmesi ve sınıflandırılması, su kütlelerinin kalitesini ve sürdürülebilir kullanımını değerlendirebilmek ve gerektiğinde rehabilitasyonu için zamanında ve yerinde önlemler alabilmek adına politika yapıcılar ve bölgesel kurumlar/kuruluşlar için büyük önem taşımaktadır. Bölgesel sözleşmelerden Helsinki Sözleşmesi, Oslo-Paris Sözleşmesi, Barselona Sözleşmesi - Akdeniz'de Deniz Kirliliğinin Değerlendirilmesi ve Kontrolü Programı, Stratejik Eylem Programı ve Entegre Kıyı Alanları Yönetimi ve yasal araçlardan Su Çerçeve Direktifi, Deniz Stratejisi Çerçeve Direktifi, Kentsel Atıksu Arıtımı Direktifi ve Nitrat Direktifi; Amerika Birleşik Devletleri Çevre Koruma Ajansı tarafından yayınlanan Temiz Su Yasası, Su Kalitesi Yasası, Ulusal Çevre Politikası ve Kıyı Bölgesi Yönetim Yasası, ötrofikasyonu önlemek adına izlemenin önemine değinir. Bölgesel gereklilikler doğrultusunda ötrofikasyon indeksleri, bir bölgeyi temsil eden belirli durum değişkenleri ve parametrelerden oluşan veri setleri kullanılarak geliştirilmiştir. Bu çalışmada, dünya çapında yaygın olarak kullanılan ötrofikasyon indeksleri uygulama alanları, parametreleri, yöntemleri ve sınıflandırma ölçeklerine göre incelenmiş ve indekslerin trofik durum eşdeğerleri ortaya konmuştur. Bu çalışma, bir su kütlesinin trofik durum değerlendirmesinde araştırmacılara ötrofikasyon indeksleri ile ilgili kapsamlı bir inceleme sunmak amacıyla yapılmıştır.

Anahtar kelimeler: *Trofik durum sınıflandırma, ötrofikasyon indeksleri, su kalitesi yönetimi*

Introduction

In ecosystems, living and non-living environment interacts each other (Kupchella & Hyland, 1989). For example, algal growth relies on nutrient uptake whose main mechanism enables the removal of dissolved nutrient from water. Algae constitutes the boosting compounds for primary production in lakes and estuaries, having a prevailing role in subsequent trophic status (Bowie et al., 1985). Eutrophication occurs under the effect of algal dynamics and poses significant problems to aquatic ecosystem as losses in biodiversity, ecosystem degradation, harmful algal blooms (HAB) and oxygen deficiency in bottom waters (EC 2010/477/EU, 2010; EU 2008/56/EC, 2008).

Eutrophication is one of the most important threats for ecological health and water quality (Dodds, 2002; HELCOM, 2007; Jiang et al., 2011; OSPAR, 2008; Pan et al., 2015). Despite the nutrients are natural ingredients of the ecosystem, their excessive concentrations may cause negative effects in the aquatic strata (Heiskary & Bouchard Jr, 2015). Dramatic increase of nitrogen and phosphorus accelerates algal growth and thus, biomass increases. Harmful algal species altered by the eutrophication dominate other algae populations. Changing the balance between organisms results in the degradation of water quality (Heiskary & Bouchard Jr, 2015). Overgrowth of algae and later, inhibition of sun rays in water surface causes the shading decay, suffocation and even toxicity in shellfishes and fishes. In that sense, the effects of nutrient enrichment are increase in chlorophyll concentration, decrease in water transparency related to suspended algae, abundance of opportunistic macro algae, abundance of perennial seaweeds and seagrasses, decrease in dissolved oxygen, species shift in floristic composition (EC 2010/477/EU, 2010).

Three types of HAB occur in aquatic ecosystem related with the global increase of nutrient pollution which are toxic algae, potentially toxic algae, and red tides. Toxic algae mainly involve *Karenia, Alexandrium, Dinophysis and Pseudonitzschia* whereas potentially toxic algae mainly refer to *Pseudonitzschia.* The occurrence of red tides is generated by large biomass blooms which may compose of *Phaeocystis, Lepidodinium, Noctiluca,* and reflect brownish, green or white tides (Ferreira et al., 2011; USEPA, 2008). Some of the studies shows HAB occurrence in aquatic ecosystem in Turkey. For example, Tas and Noyan (2015) listed 23 potentially harmful and/or bloom-forming microalgae of the Golden Horn Estuary which includes species of *Pseudonitzschia, Dinophysis, Noctiluca*. Turkoglu (2013) observed that dramatically increase on density of *Noctiluca scintillans* in bloom period March-June and October-December in Çanakkale Strait.

There are significant causes of nutrient enrichment in water resources namely population increase resulting in domestic wastewater discharges and overuse of nitrogen and phosphorus in agriculture. As a result of these pressures, aquatic ecosystem continues to alter the sources in the food chain (Galloway & Cowling, 2002; Glibert et al., 2006; Howarth et al., 2002; Smil, 2004).

In the light of the above-mentioned reasons, the appropriate amount of nutrients to protect the aquatic life, and the relationship between the biological conditions of aquatic organisms and the nutrient concentrations requires a variety of methods to describe. Considering chronological studies on eutrophication assessment, the studies of Hoyer et al. (2015) and Kitsiou and Karydis (2011) have gained the importance. The focus of relevant scientific resources is mainly on aquatic ecosystem's health and therefore, nutrient concentrations in water. The prominent studies about eutrophication indexes were the ones related mainly to phosphorus concentrations (Canfield Jr & Bachmann, 1981; Chapra, 1977; Liebig, 1840; Vollenweider, 1976). Thereafter, estimation of chlorophyll concentrations from total phosphorus was studied (Canfield Jr & Bachmann, 1981; Jones & Bachmann, 1976). The chlorophyll concentration might still be regarded as an important indicator independently. On the other hand, studies on water clarity were carried out and trophic status was defined by the level of dependent phytoplankton colonization (Carlson, 1977; Hoyer et al., 2015; USEPA, 2008). In that sense, various models were developed in order to assess the relationship between biodiversity and nutrient loads (Cloern, 2001). Finally, various indexes were developed to determine the trophic status based on specific indicators (e.g. dissolved oxygen (DO), dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), total nitrogen (TN), total phosphorus (TP), secchi disk depth (SD), chlorophyll-a (Chl-a), seagrass, macrobenthos, HAB, benthic invertebrates, phytoplankton, primary production, phanerogam coverage (Ph)) (Bricker et al., 2003; Bricker et al., 1999; Carlson, 1977; Cloern, 2001; Giordani et al. 2009; HELCOM, 2007; Ignatiades, 2005; Souchu et al., 2000; USEPA, 2008; Vollenweider et al., 1992; EC WFD, 2009). These parameters and biological indicators are commonly used in research and management of lakes and coastal waters (Andersen et al., 2011; Conley et al., 2000; Cunha et al., 2013; Devlin et al., 2011; Nielsen et al. 2002; Parparov et al., 2010; Pettine et al., 2007; Rask et al., 1999; Rinaldi & Giovanardi, 2011; Stips et al., 2016).

The indexes provide statistical data on water quality for research and management activities according to the sustainability criteria and objectives of local or regional conventions and legislative frameworks (e.g. to achieve good ecological status (GES) by Water Framework Directive (WFD), to achieve successful goals by United State Environmental Protection Agency (USEPA), to protect the Northeast Atlantic by Oslo-Paris Convention (OSPAR), to protect Baltic Sea by Helsinki Convention (HELCOM), to protect the Mediterranean Sea by the Programme for the Assessment and Control of Marine Pollution in the Mediterranean (MEDPOL)) (HELCOM, 2007; OSPAR, 2008; USEPA, 2008; EC, 2000).

In this study, worldwide used eutrophication indexes evaluated and compared with regard to their parameters, variables, regions and specific states.

Method

Commonly Used Eutrophication Indexes

Several eutrophication monitoring and assessment methods have been developed and implemented in lakes and marine water bodies. The summary of these methods based on their integrated indicators and implementation areas (Table 1).

Table 1

Methods for eutrophication assessment modified from Ferreira et al. (2011) and Borja et al. (2012)

Method name	Area of application	Biological indicators and parameters	Physico-chemical parameters	Assessment
EPA NCA Water Quality Index	US	Chl-a	Water clarity, DO, DIN, DIP	Classified as good, fair, poor
TRIX	EU	Chl-a	DO, DIN, TP	Scales from 0 (elevated) to 10 (bad).
ASSETS	US, EU, Asia, Australlia	Chl-a, macroalgae, seagrass, HAB	DO	Scales from high to bad in four categories
TWQI/LWQI	EU	Chl-a, macroalgae, seagrass, phanerogam coverage (Ph)	DO, DIN, DIP	Scales from 0 (worst status) to 100 (best status).
OSPAR	North East Atlantic	Chl-a, macroalge, seagrass, phytoplankton indicator species	DO, TP, TN, DIN, DIP	Identify non-problem areas, potential problem areas and problem areas
WFD	Basque Country, UK	Phytoplankton, Chl-a, macroalgae, benthic invertebrates, seagrass	DO, TP, TN, DIN, DIP, water clarity	Scales from high to bad in five categories
HEAT	Baltic	Chl-a, primary production, seagrass, benthic invertebrates, HAB, macroalgae	DIN, DIP, TN, TP, DO, water clarity	Areas with values <1.00 are defined as 'unaffected by eutrophication', while areas with values ≥1.00 are defined impaired and 'affected by eutrophication'.
BEAST	Black Sea, Ukraine Romania, Bulgaria	Chl-a, primary production, seagrass, benthic invertebrates, HAB, macroalgae	DIN, DIP, TN, TP, DO, water clarity	Scales from 1 (high) to 5 (bad)
IFREMER	France	Chl-a, seagrass, macrobenthos, HAB	DO, water clarity, SRP, TP, TN, DIN, DIP, sediment organic matter, sediment	Scales from blue (high) to red(bad) in five categories
CTSI	North America	Chl-a	TP, SD	Oligotrophic CTSI ≤ 40 Mesotrophic 40 < CTSI ≤ 50 Eutrophic 50 < CTSI ≤ 70 Hyper-eutrophic CTSI>70
TLI (for lakes)		Chl-a	SD, TP, TN	Scales from 1 (ultramicrotrophic) to 7 (hypertrophic)

Environmental Protection Agency National Coastal Assessment (EPA NCA) Water Quality Index (WQI).

This index has been used to characterize degraded water quality conditions in the US waters. The criteria have been identified for East/Gulf Coast Sites, West Coast, Hawaii, Puerto Rico and Florida Bay. According to the EPA 2004 report, water quality cannot be described by a simple index for all estuarine systems. For instance, an index cannot work out for a specific estuary while it may be used for several regions appropriately. Therefore, it is important to determine which indicator should be weighted greater or less according to the characteristics of the region.

EPA, NCA and WQI includes five parameters which are Chl-a, water clarity, DO, DIN and DIP. It describes poor, fair and good water quality status for each parameters and defines a WQI value according to the number of each parameters specific to region (see Table 2 and Table 3) (USEPA, 2004)

Table 2

Regional Biological Parameter Criteria (USEPA, 2004)

	Chl-a (µg L^{-1})		
	Good	Fair	Poor
East/Gulf, West Coast Sites	< 5	5–20	> 20
Hawaii, Puerto Rico	< 0.5	0.5–1	> 1
Florida Bay	< 1	1–5	> 5

Table 3

Regional Physico-chemical Indicator Criteria (USEPA, 2004)

<table>
<tr><th></th><th colspan="3">DIN (mg L^{-1})</th><th colspan="3">DIP (mg L^{-1})</th><th colspan="3">WCI ratio*</th><th colspan="3">DO (mg L^{-1})</th></tr>
<tr><th></th><th>Good</th><th>Fair</th><th>Poor</th><th>Good</th><th>Fair</th><th>Poor</th><th>Good</th><th>Fair</th><th>Poor</th><th>Good</th><th>Fair</th><th>Poor</th></tr>
<tr><td>East/Gulf Coast Sites</td><td><0.1</td><td>0.1–0.5</td><td>>0.5</td><td><0.01</td><td>0.01–0.05</td><td>>0.05</td><td rowspan="3">>2</td><td rowspan="3">11–2</td><td rowspan="3"><<1</td><td rowspan="3">>5</td><td rowspan="3">22–5</td><td rowspan="3"><<2</td></tr>
<tr><td>West Coast sites</td><td><0.5</td><td>0.5–1.0</td><td>>1</td><td><0.01</td><td>0.01–0.1</td><td>>0.1</td></tr>
<tr><td>Hawaii, Puerto Rico, Florida Bay</td><td><0.05</td><td>0.05–0.1</td><td>>0.1</td><td><0.005</td><td>0.005–0.01</td><td>>0.01</td></tr>
</table>

Note. **WCI*= (observed clarity at 1 meter)/ (regional reference clarity at 1 meter)

Criteria for the determination of the WQI specific to region are given as "**good status**: maximum one parameter is fair, and none of parameters are poor"; "**fair status**: one of the parameters is poor or two/more parameters are fair"; "**poor status**: two/more of the five parameters are poor" (USEPA, 2004).

Trophic Index (TRIX).

As a multimetric index, TRIX describes trophic status regarding four indicators (Chl-a, DO, DIN, DIP). It was presented by Vollenweider within the scope of the OECD Programme on Eutrophication (Vollenweider & Kerekes, 1980; Vollenweider et al., 1998). This index was formed to evaluate the coastal water quality in Italy by collecting data between 1982-1993 along the coast of Emilia-Romagna in NW Adriatic. The area was strongly affected by the Po river inputs (Vollenweider et al., 1992).

TRIX was adopted by MEDPOL for trophic classification of the coastal waters in Mediterranean Sea and by this way, the parameters of TRIX were agreed to be monitored (UNEP/MAP, 2007a, 2007b; WFD 2000/60/EC Technical Report, 2009). TRIX was also used in various areas of the Mediterranean Sea (e.g. Adriatic Sea and Tyrrhenian Sea) (EC WFD Technical Report, 2009).

Four parameters of TRIX are Chl-a, oxygen as absolute (%) deviation from saturation, DIN (as NO_3-N + NO_2-N + NH_4-N) and TP (Giovanardi & Vollenweider, 2004; R. Vollenweider et al., 1998). Numerically, the index scales from 0 to 10 covering a wide range of trophic conditions from oligotrophy to eutrophy. Accordingly, lower TRIX values show a good eutrophication status, while higher values represent worse conditions.

The basic structure of TRIX is given below Eq. 1 and Eq. 2:

$$TRIX = (k/n)\sum_{1}^{i=n}\left(\frac{logU - logL}{logM - logL}\right)i \tag{1}$$

where
k: scale coefficient,
n: number of the variables,
i: number of the variables,
M: measured value of the variable,
U: upper limit,
L: lower limit.

$$TRIX = (Log10\ [ChA \times aD\%O \times minN \times TP] - k)/m. \quad (2)$$

where
ChA: chlorophyll-*a* concentration ($\mu g\ L^{-1}$),
aD%O: oxygen as absolute % deviation from saturation,
minN: mineral nitrogen: DIN: N (as NO_3-N + NO_2-N + NH_4-N) ($\mu g\ L^{-1}$),
TP: total phosphorus ($\mu g\ L^{-1}$)
m: scale coefficient

In addition to TRIX, turbidity index (TRBIX) defines the Secchi disc (SD) transparency in combination with chl-a concentration. The trophic index (TRIX) is combined with turbidity index (TRBIX) and a general water quality index (GWQI) is formed covering microbiological conditions (Vollenweider et al., 1998).

TRIX is studied also in Mediterranean, Aegean Sea, Marmara Sea and Black Sea between the years of 2014-2017 by Ministry of Environment and Urbanization of Turkey (MEU) (MEU, 2018). For instance, their results show that in the Mediterranean, minnimum TRIX value is less than 1 and maximum is more than 5 and in the Marmara Sea, minnimum TRIX value is less than 3 and maximum is more than 6.

Assessment of Estuarine Trophic Status (ASSETS).

A substantial part of ASSETS methodology was developed for National Estuarine Eutrophication Assessment (NEEA) by Bricker et al. (1999). NEEA has three tools as Overall Eutrophic Condition (OEC), Overall Human Influence (OHI) and Definition of Future Outlook (DFO). This approach combines primary and secondary symptoms. NEEA approach was extended by Bricker et al. (2003) as ASSETS method by modelling the relative contribution of anthropogenic nutrient sources (OHI) and based on the combination of relational databases (e.g., Geographical Information Systems) with a more quantitative procedure (e.g., statistical criteria) for the determination of parameters to evaluate the status of OEC (Bricker et al., 2003).

While NEEA is applied only in the coastal waters of United States, ASSETS aims to cover the requirements of the WFD with regard to a few quality elements for transitional waters (S. Bricker et al., 2003). ASSETS which is a synthesis of three different NEEA tools: OEC, OHI and DFO, defines five categories as bad, poor, moderate, good and high, respectively. The categories of OHI value are classificated as low: 0 to <0.2, moderate low: >0.2 to 0.4, moderate: >0.4 to 0.6, moderate high: >0.6 to 0.8 and high: >0.8, respectively.

The OEC value has two groups that are primary (early) and secondary (advanced) symptoms of eutrophication. Chl-a, macroalgae and epiphytes are considered primary symptoms. Low DO, losses of submerged aquatic vegetation (SAV) and occurrence of nuisance and/or toxic algal blooms are considered secondary symptoms.

The formulas indicating symptom level and the level of expression value for primary symptoms obtained are given below (Bricker et al., 2003).

$$\sum_{i=1}^{n}\left(\frac{Az}{Ae}\right)(Expression\ Value) = Sypmtom\ level\ of\ expression\ value\ for\ estuary \tag{3}$$

$$P1 = \frac{1}{p}\sum_{1}^{P}\left[\sum_{1}^{n}\left(\frac{Az}{Ae}E1\right)\right] \tag{4}$$

where
Az is the surface area of each zone,
Ae is the total estuarine surface area,
E_1 is the expression value at each zone,
n is the number of estuarine zones,
P_1 is the level of expression of the primary symptoms for the estuary,
p is the number of primary symptoms (Bricker et al., 2003).

Evaluation of ASSETS method is based firstly on the selection of the highest of the three estuary symptom level of expression values, secondly on the chosen the level of expression value of secondary symptoms for the estuary. Secondary symptoms are evaluated to be a clear indicator of the problem. The evaluation is done based on expression values between 0-1 and the conditions as high, moderate high, moderate, moderate low and low, respectively (Bricker et al., 2003).

According to the ASSETS approach, some primary symptoms (e.g. epiphytes) and secondary symptoms (e.g. toxic blooms) may only be evaluated by estimation while others such as Chl-a and DO contents are assessed based on the quantitative values (Bricker et al., 2003).

Transitional Water Quality Index (TWQI).

TWQI was developed from the water quality index of the U.S. National Sanitation Foundation by Giordani et al. (2009). It is a simple tool which integrates the information from abiotic and biotic measurements where SAV controls primary production due to shallow depth and provides a comprehensive assessment of trophic status quantitatively. This multimetric index includes six main variables which are the relative coverage of benthic phanerogams and opportunistic macroalgae species, concentrations of DO, phytoplankton, Chl-a, DIN and DIP (Giordani et al., 2009). The index scales from 0 (worst status) to 100 (best status).

TWQI has been used in transitional waters of Southern Europe. TWQI is obtained by the sum of weighted Quality Values as a non-linear quality function of measured variables as described in Eq. 5 (Bonometto et al., 2016).

$$TWQI = \sum (wfQVs) \tag{5}$$

Where wf is weighing factors and QVs is the quality values of the six main variables.

Oslo Paris Convention Method (OSPAR).

The OSPAR Common Procedure inserts two procedural stages: an initial screening of the selected marine areas and the implementation of a comprehensive procedure assessment. Screening stage identifies the areas where there is no eutrophication threat. These areas are classified as “non-problem area” which does not need second stage of comprehensive procedure assessment. The other classifications fall under “potential problem area” and “problem area” which need to apply comprehensive procedure assessment (OSPAR, 2008).

The OSPAR comprehensive procedure has four categories to evaluate eutrophication conditions in North East Atlantic. Category 1, namely nutrient enrichment, includes nutrient inputs, DIN and DIP concentrations in winter period. Category 2, namely direct effects of nutrient enrichment, includes Chl-a concentration, elevated levels of toxic phytoplankton indicator species and macrophytes and shift from long-lived to short-lived nuisance macrophytes species. Category 3, namely indirect effects of nutrient enrichment, includes oxygen deficiency, kills and long-term area-specific changes in zoobenthos biomass and fish and elevated levels of organic carbon/organic matter (area-specific) in relation to oxygen deficiency. Category 4, namely other possible effects of nutrient enrichment, includes algal toxins and transboundary transport.

Water Framework Directive (WFD).

The WFD has five ecological classes which are “high, good, moderate, poor and bad”, aimed to achieve good ecological status in all European water bodies by 2015 (EC, 2000; WFD Guidance, 2003). However, a large number of exemptions were given to member states for extending the first deadline (2015) for meeting the objectives to further deadlines as 2021 or 2027 due to technical inability, disproportionate expenses or natural barriers for timely improvement (Tsakiris, 2015). Each member state is required to adapt the WFD assessment processes. In this context, eutrophication assessment is done to define ecological status where nutrient enrichment changes biological and physico-chemical parameters.

The WFD has two assessments. The first one is ecological status assessment for current situation and reveals eutrophication status indicating the movement of quality elements towards moderate/poor/bad. The second is risk assessment (predictive analysis) to estimate future condition and prevent deterioration using information on predicted changes in pressure that likely end in aquatic environment under the risk of eutrophication in near future (EC WFD Technical Report, 2009).

Marine Strategy Framework Directive (MSFD) as a sub-directive of WFD aims to achieve “Good Environmental Status (GES)” in the marine environment by 2020 at the latest, while the WFD aims “Good Ecological Status”. MSFD focuses on minimizing anthropogenic sources of eutrophication in marine environment, while the WFD covers the whole pressures for eutrophication. MSFD has complementarity with the WFD in coastal waters as defined in Article 3.1 of MSFD (EC WFD Technical Report, 2009)

HELCOM Eutrophication Assessment Tool (HEAT) and Black Sea Eutrophication Assessment Tool (BEAST).

HEAT is an eutrophication indicator and developed for the Baltic Sea where eutrophication has been a major problem since the 1900s (HELCOM, 2007). Based on the same principles already proposed by Vollenweider (1998) HEAT determines eutrophication level by five parameters which are nitrogen, phosphorus, chl-a, water clarity and oxygen.

HEAT identifies areas as “affected by eutrophication” and “unaffected by eutrophication” according to the Ecological Quality Ratio (EQR). EQR changes between 0 (worst) and 1 (best). The threshold value of EQR is 0,67 and if the result is less than 0,67, it is unacceptable as its deviations from reference conditions are moderate, major or strong indicating that the area affected by eutrophication as its status is moderate, poor or bad (HELCOM, 2009).

BEAST is developed for the Black Sea within the scope of the EU funded Baltic2Black Project and tested for the eutrophication assessment of the Romanian coastal waters (BSC, Helcom & EC, 2014). It has the same principles with HEAT and its parameters are specific to country (Lazar et al., 2016). According to the developer of the HEAT tool, BEAST could ideally be an improved version of HEAT. It can also be used to assess the influence of seawater temperature at eutrophic condition (BSC, Helcom & EC, 2014)

Institut Français pour l'Exploration de la Mer (IFREMER).

IFREMER is in charge of the overall coordination of the WFD in France. IFREMER method uses mean annual or mean seasonal data compared to a fixed scale to define the status for chl-a with five coloured level to match the WFD evaluation (see Table 4). The IFREMER method is based on the description of physical, chemical and biological potential indicators of eutrophication in the various sections of the lagoon ecosystem: benthic, phytoplankton, macrophytes, macrofauna, sediment and water. This method uses the 90th percentile of annual or seasonal Chl-a data (Souchu et al., 2000; Zaldívar et al., 2008).

Table 4

Trophic Status Classification Based on IFREMER (Zaldívar et al., 2008)

Parameter			BLUE (High)		GREEN (Good)		YELLOW (Moderate)		ORANGE (Poor)		RED (Bad)
$\Delta\%O_2$ sat		0		20		30		40		50	
TUR	NTU	0		10		20		30		40	
PO_4^{3-}	µM	0		0,3		1		1,5		4	
diN	µM	0		15		20		40		60	
NO_2-N	µM	0		0,5		1		5		10	
$NO_{3-}N$	µM	0		7		10		20		30	
NH_4-N	µM	0		7		10		10		30	
Chl-a	µM	0		5		7		15		30	
Chl-a/phaeo	µM	0		7		10		10		40	
TN	µM	0		50		75				12	
TP	µM	0		1		2		5		8	

Statistical Trophic Index (STI).

STI method was implemented in the Aegean Sea by Ignatiades (2005). This index defines the classification as “open oligotrophic < offshore mesotrophic < inshore eutrophic waters” according to Chl-a content and primary production. Accordingly, the values for index parameters are 0.5<(0.5– 1.0)<1.0 mg m^{-3} and 1.5<(1.5–3.0)<3.0 mg C m^{-3} h^{-1} for Chl-a and primary production, respectively (Ignatiades, 2005).

Carlson Trophic State Index (CTSI).

CTSI was calculated in three Minnesota Lakes in 1972. This index used SD transparency, TP and Chl-a as indicator parameters. According to Carlson, the best indicator of trophic status may differ from one lake to another and seasonally. For this reason, the best indicator should be chosen pragmatically (Carlson, 1977).

Carlson derived the TSI formula using the equations of the relationship between TP and summer Chl-a concentration defined by Dillon and Rigler (1974). CTSI is calculated via the formulas below (Carlson, 1977).

$$TSI(SD) = 10\left(6 - \frac{lnSD}{ln2}\right) \tag{6}$$

$$TSI(Chl) = 10\left(6 - \frac{2{,}04 - 0{,}68\,lnChl}{ln2}\right) \tag{7}$$

$$TSI(TP) = 10\left(6 - \frac{ln\frac{48}{TP}}{ln2}\right) \tag{8}$$

$$CTSI = \frac{1}{3}\left(TSI(SD) + TSI(Chl) + TSI(TP)\right) \tag{9}$$

where
TSI is trophic state index,
SD is secchi disk,
Chl-a is surface chlorophyll-a concentration (mg/m^3),
TP is total surface phosphorus concentration (mg/m^3)

The index indicates the potential concentration of a watershed or region, at least on the basis of phosphorus. Table 5 gives the classification of CTSI.

Table 5

Trophic StatusClassification According to CTSI (Carlson & Simpson, 1996)

Trophic State	CTSI value
Oligotrophic	≤40
Mesotrophic	40<CTSI≤50
Eutrophic	50<CTSI≤70
Hypertrophic	CTSI>70

Trophic Level Index (TLI).

TLI has been used for lakes in New Zealand. The index includes three indicators which are SD depth, TP and Chl-a concentrations. TLI has subsequently been supported with TN concentrations. Thus, each individual index of TLI (Tli) is the form of a logarithmic function connecting the trophic level to four “trophic” parameters (Burns et al., 2000).

$$Tli = ai + biLOG(Pari) \quad (10)$$

where
I shows the indices (each of the four parameters),
a and b are coefficients,
Par_i is SD depth, concentration of TN, TP and Chl-a

Equations (11), (12), (13) and (14) given below for the indices of TLI were defined with the coefficients by Burns et al. (1999) and Burns et al. (2000).

$$TLc = 2.22 + 2.54 \log(Chl\text{-}a) \quad (11)$$

$$TLs = 5.10 + 2.27 \log(1/SD -- 1/40) \quad (12)$$

$$TLp = 0.218 + 2.92 \log(TP) \quad (13)$$

$$TLn = -3.61 + 3.01 \log(TN) \quad (14)$$

Calculation of TLI is as below Eq. (15).

$$TLI = ¼ (TL_{Chl} + TL_{SD} + TL_{TP} + TL_{TN}) \quad (15)$$

TLI has seven classification categories ranking from 1 (ultramicrotrophic) to 7 (hypertrophic).

Assessment of Eutrophication Indexes

Throughout the development stages of eutrophication indexes, information about trophic status has been provided to local authorities and policy makers. International legislation like the WFD and the USEPA require further developing tools, which can be implemented for different water bodies. Therefore, integrated approaches for combining, analysing and evaluating information from various interrelated variables are important for water quality management, environmental sustainability and development of cost-effective indicators regarding eutrophication. Consequently, different targets are required for water bodies at regional scale.

The indexes show that trophic status of water bodies related with nutrient levels, water clarity (Secchi disc depth), DO and Chl-a levels, and the phytoplankton community regarding physical, chemical and biological aspects. Additionally, it is observed that the main phytoplanktonic parameter Chl-a is a common parameter for all eutrophication indexes as it is directly related to the trophic status.

On the other hand, there is not any internationally defined and accepted eutrophication index. Indexes are developed using specific data sets of state variables and parameters representing a location. Therefore, there is not comparability between indexes as they are specific to the region where they have been developed. They might be adopted by the additional data representing the study area.

It is observed that the requirements of USEPA and WFD for eutrophication monitoring are taken as basic approach. The EPA method is associated with US waters, while the WFD comprises water resources of the European Union and candidate countries. The WFD addresses all types of pressures and hence, assessment is done accordingly. However, the HEAT and OSPAR indexes take into account nutrient enrichment and deriving impacts.

In order to calculate an EQR, deviation from the recent monitoring data and the data related to type-specific reference conditions are compared as recommended by the WFD and HEAT. The results are assessed based on the status of the quality element having the worst condition (one-out all-out principle).

The OSPAR uses area-specific and/or historical reference levels for each criteria. This method reflects an additive mechanism across causative, direct and indirect effects as well as other tangible effects. According to the WFD Guidance No 23, the OSPAR Common Procedure can be tested by non-OSPAR Contracting Parties but it is a difficult to handle nutrients as it only uses winter values of nutrients and is used in open seas (WFD 2000/60/EC Technical Report, 2009).

Table 6

Comparison of Assessment Results According to WFD Guidance No 23 (EC WFD Technical Report, 2009)

Ecological status	WFD	OSPAR	HELCOM	MSFD
High	Nearly undisturbed conditions	Non-problem area	Area not affected by eutrophication	-
Good	Slight change in composition, biomass	Non-problem area	Area not affected by eutrophication	Human induced eutrophication is minimized
Moderate	Moderate change in composition, biomass	problem area	Area affected by eutrophication	Human induced eutrophication is not minimized
Poor	Major change in biological communities	problem area	Area affected by eutrophication	Human induced eutrophication is not minimized
Bad	Severe change in biological communities	problem area	Area affected by eutrophication	Human induced eutrophication is not minimized

Table 6 gives the assessment results of conventions and legislation instruments. Within this framework, the OSPAR and HELCOM assessments are similar with respect to the eutrophication criteria. While the MSFD focuses on human induced eutrophication, the WFD deals with the changing conditions of water bodies. TLI has seven; WFD, IFREMER and ASSETS have five; OSPAR, TRIX and CTSI have four; EPA and STI have three; and HEAT has two assessment categories, respectively. Although the indexes have different methods and variables, they aim to categorise trophic status according to their regional objectives. In that sense, the assessment types are matching with each other. Table 7 shows a general assessment of the widely used indexes and clearly indicates the trophic status equivalence between them.

Table 7

Comparison of 11 Eutrophication Indexes According to Their Assessment Types of Trophic Status

Index	Trophic Status						
WFD		High	Good	Moderate	Poor	Bad	
HEAT		Not affected by eutrophication 1		0,67	Affected by eutrophication	0	
IFREMER		Blue	Green	Yellow	Orange	Red	
OSPAR		Non-problem area	Non problem area or potential problem area	Potential problem area or problem area	Problem area	Problem area	
TWQI		100				0	
ASSETS		High	Good	Moderate	Poor	Bad	
TRIX		<4 Elevated	4-5 Good	5-6 Mediocre		>6 Bad	
EPA NCA		Good 1		Fair		Poor 5	
STI		Open Oligotrophic		Offshore Mesotrophic	Inshore Eutrophic		
TLI	Ultramicrotrophic 0	Microtrophic	Oligotrophic	Mesotrophic	Eutrophic	Supertrophic	Hypertrophic 7
CTSI			Oligotrophic ≤40	Mesotrophic 40-50	Eutrophic 50-70	Hypertrophic >70	

According to Table 7, trophic state equivalences of TRIX results of MEU (2018) in the other indexes could be read as Table 8.

Table 8

Trophic Status Equivalences of TRIX Results of MEU (*2018) in the Other Indexes*

	Mediterranean		Marmara	
	Minnimum result	Maximum result	Minnimum result	Maximum result
TRIX results	Less than 1	More than 5	Less than 3	More than 6
	Trophic status equivalences in the other indexes			
WFD	High	Poor	High	Bad
HEAT	Not affected by eutrophication	Affacted by eutrophication	Not affected by eutrophication	Affacted by eutrophication
IFREMER	Blue	Yellow	Blue	Red
OSPAR	Non-problem area	Potential problem area or problem area	Non-problem area	Problem area
TWQI	More than 84	Less than 14	More than 50	Worst
ASSETS	High	Moderate	High	Poor
EPA NCA	Good	Fair	Good	Poor
STI	Open oligitrophic	Offshore mesotrophic	Open olıgitrophic	Inshore eutrophic
TLI	Ultramicrotrophic	Mesotrophic	Microtrophic	Supertrophic
CTSI	Oligotrophic	Eutrophic	Oligotrophic	Eutrophic

By reviewing the indexes, researchers revealed that TRIX and HEAT have the same basic structures and provide rather harmonic results concerning the eutrophication status and trend (Stips et al., 2016). When TRIX and TWQI are compared one can see that data inputs are comparable with the exception of benthic flora and SD depth whose utilization is not obligatory in TWQI. IFREMER and TWQI show consistency except for waters having lower quality. IFREMER is restricted due to the limited variables whereas TWQI relies on integrated variables (Giordani et al., 2009 The new tool BEAST matches with the MSFD requirements in terms of water quality, and is much more correlated with the eutrophication parameters than TRIX as well as takes into account the influence of seawater temperature on eutrophication.

Conclusion

Eutrophication indexes are used for monitoring and quality assessment of water bodies. This study indicates that the classifications of the indexes are basically similar to each other, while their indicator parameters are different resulting from the inherent characteristics of the applied regions. In this context, there is not an internationally defined and accepted eutrophication index.

This paper is important to set forth a comprehensive search and categorizes commonly used eutrophication indexes in terms of their trophic status equivalence while indicating that there is no comparability between indexes as they are developed specific to region.

References

Andersen, J. H., Axe, P., Backer, H., Carstensen, J., Claussen, U., Fleming-Lehtinen, & V.,Korpinen, S. (2011). Getting the measure of eutrophication in the Baltic Sea: towards improved assessment principles and methods. *Biogeochemistry, 106(2)*, 137-156.

Bonometto, A., Giordani, G., Ponis, G., Facca, C., Brusà, R. B., Sfriso, A., & Viaroli, P. (2016. March). Eutrophication assessment in transitional waters A performance analysis of the Transitional Water Quality Index (TWQI). *The VII Eurolag - European Coastal Lagoons Symposium.* University of Murcia, Spain.

Borja, A., Basset, A., Bricker, S., Dauvin, J., Elliot, M., Harrison, T.,…West, R. (2012). Classifying ecological quality and integrity of estuaries. In E. Wolanski & D. McLusky (Eds.), *Treatise on Estuarine and Coastal Science* (pp. 125-162). Waltham: Academic Press.

Bowie, G. L., Mills, W. B., Porcella, D. B., Campbell, C. L., Pagenkopf, J. R., Rupp, G. L., & Chamberlin, C. E. (1985). *Rates, constants, and kinetics formulations in surface water quality modeling* (2nd ed., Vol. EPA/600/pp. 3-85/040). Athens, Georgia.

Bricker, S., Ferreira, J., & Simas, T. (2003). An integrated methodology for assessment of estuarine trophic status. *Ecological modelling, 169(1)*, 39-60.

U.S. Department of Commerce, National Oceanographic and Atmospheric Administration, National Ocean Service. (1999). *National estuarine eutrophication assessment: effects of nutrient enrichment in the nation's estuaries.* Silver Spring: National Ocean Service, National Oceanic and Atmospheric Administration.

BSC, HELCOM, & EC. (2014). *Environmental monitoring of the Black Sea with focus on nutrient pollution (Acronym: Baltic2Black).* (Final Report for the Grant Agreement No 07.0204/2010/580913/SUB/D2). Retrieved from https://ec.europa.eu/environment/marine/international -cooperation/regional-sea-conventions/bucharest/pdf/Baltic2Black%20report.pdf

Burns, N., Bryers, G., & Browman, E. (2000). *Protocol for monitoring trophic levels of New Zealand lakes and reservoirs*. New Zealand: Ministry for the Environment.

Burns, N. M., Rutherford, J. C., & Clayton, J. S. (1999). A monitoring and classification system for New Zealand lakes and reservoirs. *Lake and Reservoir Management, 15(4)*, 255-271.

Canfield Jr, D. E., & Bachmann, R. W. (1981). Prediction of total phosphorus concentrations, chlorophyll a, and Secchi depths in natural and artificial lakes. *Canadian Journal of Fisheries and Aquatic Sciences, 38(4)*, 414-423.

Carlson, R. E. (1977). A trophic state index for lakes1. *Limnology and oceanography, 22(2)*, 361-369.

Carlson, R. E., & Simpson, J. (1996). A coordinator's guide to volunteer lake monitoring methods. *North American Lake Management Society, 96*, 305.

Chapra, S. C. (1977). Total phosphorus model for the Great Lakes. *Journal of the Environmental Engineering Division, 103(2)*, 147-161.

Cloern, J. E. (2001). Our evolving conceptual model of the coastal eutrophication problem. *Marine ecology progress series, 210*, 223-253.

Conley, D. J., Kaas, H., Møhlenberg, F., Rasmussen, B., & Windolf, J. (2000). Characteristics of Danish estuaries. *Estuaries, 23(6)*, 820-837.

Cunha, D. G. F., do Carmo Calijuri, M., & Lamparelli, M. C. (2013). A trophic state index for tropical/subtropical reservoirs (TSItsr). *Ecological Engineering, 60*, 126-134.

Devlin, M., Bricker, S., & Painting, S. (2011). Comparison of five methods for assessing impacts of nutrient enrichment using estuarine case studies. *Biogeochemistry, 106(2)*, 177-205.

Dillon, P. J., & Rigler, F. (1974). The phosphorus-chlorophyll relationship in lakes 1, 2. *Limnology and Oceanography, 19(5)*, 767-773.

Dodds, W. K. (2002). *Freshwater ecology: concepts and environmental applications.* California, USA: Academic Press, Elsevier.

European Commission. (2010). 2010/477/EU Commission decision of 1 September 2010 on criteria and methodological standards on good environmental status of marine waters (C(2010) 5956). *Official Journal of the European Union*, L, 232, 14-24.

European Commission WFD Guidance. (2003). *Common Implementation Strategy for the Water Framework Directive (2000/60/EC),* Guidance Document No5. Transitional and Coastal Waters – Typology, Reference Conditions and Classification Systems.

European Commission WFD Technical Report. (2009). *Common Implementation Strategy for the Water Framework Directive (2000/60/EC)*, (Guidance Document No. 23). Guidance Document on Eutrophication Assessment in the Context of European Water Policies.

European Communities. (2000). Council directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for community action in the field of water policy. Official Journal of European Communities, 50(327), 1-72.

European Union. (2008). Directive 2008/56/EC of The European Parliament and of the Council of 17 June 2008 Establishing a Framework for Community Action in the Field of Marine Environmental Policy (Marine Strategy Framework Directive MSFD). *Official Journal of the European Union*, 19-40.

Ferreira, J. G., Andersen, J. H., Borja, A., Bricker, S. B., Camp, J., Da Silva, M. C., & Ignatiades, L. (2011). Overview of eutrophication indicators to assess environmental status within the European Marine Strategy Framework Directive. *Estuarine, Coastal and Shelf Science, 93(2)*, 117-131.

Galloway, J. N., & Cowling, E. B. (2002). Reactive nitrogen and the world: 200 years of change. *AMBIO: A Journal of the Human Environment, 31(2)*, 64-71.

Giordani, G., Zaldívar, J. M., & Viaroli, P. (2009). Simple tools for assessing water quality and trophic status in transitional water ecosystems. *Ecological Indicators, 9(5)*, 982-991.

Giovanardi, F., & Vollenweider, R. A. (2004). Trophic conditions of marine coastal waters: experience in applying the Trophic Index TRIX to two areas of the Adriatic and Tyrrhenian seas. *Journal of Limnology, 63(2)*, 199-218.

Glibert, P. M., Harrison, J., Heil, C., & Seitzinger, S. (2006). Escalating worldwide use of urea–a global change contributing to coastal eutrophication. *Biogeochemistry, 77(3)*, 441-463.

Heiskary, S. A., & Bouchard Jr, R. W. (2015). Development of eutrophication criteria for Minnesota streams and rivers using multiple lines of evidence. *Freshwater Science, 34(2)*, 574-592.

HELCOM. (2007). *HELCOM Baltic Sea Action Plan*. Krakow, Poland.

HELCOM. (2009). *Eutrophication in the Baltic Sea – An integrated thematic assessment of the effects of nutrient enrichment and eutrophication in the Baltic Sea region* (Balt. Sea Environ. Proc. No. 115B). Helsinki Commision, Finland.

Howarth, R. W., Sharpley, A., & Walker, D. (2002). Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. *Estuaries, 25(4)*, 656-676.

Hoyer, M. V., Terrell, J. B., & Canfield Jr, D. E. (2015). Chain of eutrophication models for assessing the potential impact of nutrient enrichment on Choctawhatchee Bay, FL, USA. *Florida Scientist*, 20-36.

Ignatiades, L. (2005). Scaling the trophic status of the Aegean Sea, eastern Mediterranean. *Journal of Sea Research, 54(1)*, 51-57.

Jiang, P., Cui, Y., Wang, H., & Wang, H. (2011). Macrobenthos communities and bio-assessment of lakes in Hanyang district. *Resources and Environment in the Yangtze Basin, 20(5)*, 525-533.

Jones, J. R., & Bachmann, R. W. (1976). Prediction of phosphorus and chlorophyll levels in lakes. *Journal (Water Pollution Control Federation)*, 2176-2182.

Kitsiou, D., & Karydis, M. (2011). Coastal marine eutrophication assessment: a review on data analysis. *Environment international, 37(4)*, 778-801.

Kupchella, C. E., & Hyland, M. C. (2nd ed.). (1989). *Environmental Science: Living within the system of nature.* Boston: Allyn and Bacon.

Lazar, L., Boicenco, L., Beken, C., Dzhurova, B., Moncheva, S., Shtereva, G., & Vasiliu, D. (2016). Western black sea eutrophication status according to the black sea eutrophication asessment tool, BEAST-MISIS cruise results. *Cercet. Mar., 46*, 48-60.

Liebig, J. v. (1840). *Organic chemistry in its application to vegetable physiology and agriculture. Readings in ecology.* New York, NY: Prentice Hall.

MEU, (2018). Marine quality bulletin. Retrieved from https://ced.csb.gov.tr/deniz-kalitesi-bulteni-i-84199

Nielsen, S. L., Sand-Jensen, K., Borum, J., & Geertz-Hansen, O. (2002). Phytoplankton, nutrients, and transparency in Danish coastal waters. *Estuaries, 25(5)*, 930-937.

OSPAR. (2008). *Marine biodiversity monitoring and assessment: Activities to improve synergies between EU directives and international conventions, Activities in all OSPAR Contracting Parties*: OSPAR Commission.

OSPAR. (2017). Eutrophication Status of the OSPAR Maritime Area, Third Integrated Report on the Eutrophication Status of the OSPAR Maritime Area. *Eutrophication Series*. Retrieved from www.ospar.org

Pan, B.-Z., Wang, H.-Z., Pusch, M. T., & Wang, H.-J. (2015). Macroinvertebrate responses to regime shifts caused by eutrophication in subtropical shallow lakes. *Freshwater Science, 34(3)*, 942-952.

Parparov, A., Gal, G., Hamilton, D. P., Kasprzak, P., & Ostapenia, A. (2010). Water quality assessment, trophic classification and water resources management. *Journal of Water Resources Protection, 2*, 907-915

Pettine, M., Casentini, B., Fazi, S., Giovanardi, F., & Pagnotta, R. (2007). A revisitation of TRIX for trophic status assessment in the light of the European Water Framework Directive: Application to Italian coastal waters. *Marine Pollution Bulletin, 54(9)*, 1413-1426.

Rask, N., Pedersen, S. E., & Jensen, M. H. (1999). Response to lowered nutrient discharges in the coastal waters around the island of Funen, Denmark. In *Biological, Physical and Geochemical Features of Enclosed and Semi-enclosed Marine Systems* (pp. 69-81): Springer.

Rinaldi, A., & Giovanardi, F. (2011). Contribution of Richard A. Vollenweider toward understanding eutrophication of the coastal Adriatic Sea. *Aquatic Ecosystem Health & Management, 14(2)*, 200-203.

Smil, V. (2004). *Enriching the earth: Fritz Haber, Carl Bosch, and the transformation of world food production.* London, England: The MIT Press.

Souchu, P., Ximenes, M., Lauret, M., Vaquer, A., & Dutrieux, E. (2000). Mise à jour d'indicateurs du niveau d'eutrophisation des milieux lagunaires méditerranéens, août 2000. *Ifremer-Créocean-Université Montpellier II,* 412.

Stips, A., Macias, D., Garcia-Gorriz, E., & Miladinova, S. (2016). Alternative assessments of large scale Eutrophication using ecosystem simulations: Hind-casting and scenario modelling. *Luxembourg (Luxembourg) EUR, 27904*.

Tas, S., & Yılmaz, I.E. (2015). Potentially harmful microalgae and algal blooms in a eutrophic estuary in the Sea of Marmara (Turkey). *Mediterranean Marine Science, 16(2)*, 432-443.

Tsakiris, G. (2015). The status of the European waters in 2015: a review. *Environmental Processes, 2(3),* 543-557.

Turkoglu, M. (2013). Red tides of the dinoflagellate Noctiluca scintillans associated with eutrophication in the Sea of Marmara (the Çanakkale Strait, Turkey). *Oceanologia*, *55(3)*, 709–732.

UNEP/MAP. (2007a). *Approaches to the Assessment of Eutrophication in Mediterranean Coastal Waters* (UNEP(DEPI)/MED WG.321/Inf.6). Athens: UNEP/MAP.

UNEP/MAP. (2007b). *Eutrophication Monitoring Strategy for the MED POL* (UNEP(DEPI)/MED WG.321/Inf.5). Athens: UNEP/MAP.

USEPA. (2004). *National Coastal Condition Report II* (EPA-620/R-03/002). Washington, DC: Office of Research and Development. Retrived from: https://www.epa.gov/sites/production/files/2014-10/documents/nccriicomplete.pdf

USEPA. (2008). *EPA's 2008 Report on the Environment* (EPA/600/R-07/045F). Washington, DC: National Center for Environmental Assessment.. Available from the National Technical Information Service, Springfield, VA,Retrived from: http://www.epa.gov/roe.:

Vollenweider, R., Giovanardi, F., Montanari, G., & Rinaldi, A. (1998). Characterization of the trophic conditions of marine coastal waters with special reference to the NW Adriatic Sea: proposal for a trophic scale, turbidity and generalized water quality index. *Environmetrics, 9(3)*, 329-357.

Vollenweider, R., & Kerekes, J. (1980). Loading concept as basis for controlling eutrophication philosophy and preliminary results of the OECD Programme on Eutrophication. *Progress in Water Technology, 12*, 5-38.

Vollenweider, R., Marchetti, R., & Viviani, R. (1992). Marine coastal eutrophication. The response of marine transitional systems to human impacts: problems and perspectives for restoration. *Science of the Total Environment Supplement.* Amsterdam, Netherlands: Elsevier Science.

Vollenweider, R. A. (1976). Advances in defining critical loading levels for phosphorus in lake eutrophication. *Memorie dell'Istituto Italiano di Idrobiologia, Dott. Marco de Marchi Verbania Pallanza*, *33*, 53–83.

Zaldívar, J.-M., Viaroli, P., Newton, A., De Wit, R., Ibañez, C., Reizopoulou, S., Holmer, M. (2008). Eutrophication in transitional waters: an overview. *Transitional Waters Monographs, 2(1)*, 1-78.

Extended Turkish Abstract
(Genişletilmiş Türkçe Özet)

Yaygın Olarak Kullanılan Ötrofikasyon İndekslerinin Karşılaştırılması ve Değerlendirilmesi

Ötrofikasyon sucul ortamdaki besi maddesi artışına bağlı olarak su kütlesindeki canlı ve cansız çevre üzerinde olumsuz etkilere, su kalitesinde bozulmalara yol açan bir olgudur. Sucul ortamlarda trofik durumun oligotrofikten ötrofik duruma doğru gitmesi ortamdaki besi maddesi artışının bir göstergesidir. Ötrofikasyon indeksleri, trofik durumu sınıflandırmaya yarayan bir araç olarak kullanılmaktadır. Ayrıca sucul ortamın sürdürülebilirliği açısından trofik durumun izlenmesine, sınıflandırılmasına ve iyileştirilmesine bölgesel ve uluslararası sözleşmelerde sıklıkla değinilmektedir. Bu nedenle geçmişten günümüze, trofik durumu belirlemede birçok çalışma yapılmış; birçok yaklaşım ortaya konmuştur.

Ötrofikasyon indeksleri bölgelere özgü olarak geliştirilmiştir. Her bir bölgenin koşulları birbirinden farklı olduğu için bir bölgeye özgü geliştirilen indeks, başka bir bölgede gerçekçi sonuçlar vermeyebilir. Bu kapsamda, kullanılacak parametreler birbirinden farklılık gösterebilmektedir. Bununla birlikte, bazı indekslerin su kalitesinin iyi olduğu bölgelerde birbiriyle uyumlu sonuçlar verdiği görülmektedir. Ancak trofik durum kötüleştikçe, bu sonuçların birbirleriyle tutarlılığının azaldığı görülmektedir. Bu nedenle, bölgeye özgü indeks seçimi önem arz etmektedir.

Su Çerçeve Direktifi (WFD) ve ABD Çevre Koruma Ajansı (USEPA) gibi uluslararası kuruluşların mevzuatlarında, farklı su kütleleri için uygulanabilecek değerlendirme araçlarının geliştirilmesi önerilmektedir. Bu nedenle, ötrofikasyonu yönetmek için birbiriyle ilişkili değişkenlerden gelen bilgileri analiz etmek ve değerlendirmek amacıyla entegre yaklaşımlar büyük öneme sahiptir. Bununla birlikte uluslararası geçerliliği olan ve her bölge için kullanılabilecek standart bir ötrofikasyon indeksi yoktur. Ötrofikasyon indeksleri, bölgesel gereklilikler doğrultusunda, bir bölgeyi temsil eden belirli durum değişkenleri ve parametrelerden oluşan veri setleri kullanılarak geliştirilmiştir. Bu nedenle, indeks sonuçlarının kıyaslanması mümkün değildir.

Bu çalışmada, yaygın olarak kullanılan 11 farklı ötrofikasyon indeksi, her bir indeksin uygulandığı bölgeler, kullanılan parametreler ve trofik durum sınıflandırmaları açısından değerlendirilmiştir. Bu çalışma ile araştırmacılara, bir su kütlesinin trofik durumunu değerlendirirken ötrofikasyon indeksleri ile ilgili kapsamlı ve karşılaştırılmalı bir yaklaşım sunmak hedeflenmiştir. Bu değerlendirme kapsamında klorofil-a temel biyolojik parametre olarak tüm indekslerde görülmektedir. Fiziko-kimyasal parametreler içerisinde ise çözünmüş oksijen göstergesinin tüm indekslerde kullanıldığı görülmektedir. Bu parametrelerin doğrudan ötrofikasyon ile ilgilidir. Bununla birlikte; çözünmüş oksijen, çözünmüş inorganik azot, çözünmüş inorganik fosfor, toplam azot, toplam fosfor, seki derinliği, bitkisel yayılım gibi parametreler indekslerin alan koşullarına göre hesaplama kriterleri içerisinde yer almaktadır.

Aşağıda verilen tablo indekslerin trofik durum sınıflandırma türlerini göstermektedir. Bu kapsamda sırasıyla TLI yedi; WFD, IFREMER ve ASSETS beş; OSPAR, TRIX ve CTSI dört; EPA ve STI üç; HEAT iki kategoride trofik durum sonuçlarını sunmaktadır. İndeksler farklı metot ve parametreler kullansalar da, indekslerin trofik durum değerlendirme şekilleri temelde birbiriyle örtüşmektedir. Bu çalışmada, indekslerin trofik durum eşdeğerleri ortaya konmuştur.

Ötrofikasyon indeksleri incelendiğinde USEPA ve WFD yöntemindeki değerlendirme tiplerinin temel bir yaklaşım olduğu görülmektedir. USEPA yöntemi, ABD sularındaki uygulamaları kapsamakta; WFD ise Avrupa Birliği ve aday ülke su kaynaklarına ulaşan her türlü baskıyı dikkate almaktadır. WFD su kütlelerinin değişen durumları üzerinde durmaktadır. WFD, HELCOM, OSPAR ve MSFD gibi uluslararası mevzuatlarına bakıldığında, WFD'nin ötrofik durum sınıflandırmasının daha detaylı olduğu görülürken diğer mevzuatlar iki kategoride sınıflandırma ile sınırlıdır. WFD'nin alt direktifi olan MSFD'de antropojenik kökenli ötrofikasyona dikkat çekildiği görülmektedir.

Tablo

Trofik Durum Değerlendirme Özelliklerine Göre Ötrofikasyon Indekslerinin Karşılaştırılması

INDEKS	Trofik Durum						
WFD		Çok İyi	İyi	Orta	Zayıf	Kötü	
HEAT		Ötrofikasyondan etkilenmemiş		Ötrofikasyondan etkilenmiş			
		1		0,67		0	
IFREMER		Mavi	Yeşil	Sarı	Turuncu	Kırmızı	
OSPAR		Sorunsuz bölge	Sorunsuz potansiyel veya sorunlu bölge	Potansiyel sorunlu veya sorunlu bölge	Sorunlu bölge	Sorunlu bölge	
TWQI		100				0	
ASSETS		Çok iyi	İyi	Orta	Zayıf	Kötü	
TRIX		<4 Çok iyi	4-5 İyi	5-6 Orta		>6 Kötü	
EPA NCA		İyi		Orta		Kötü	
		1				5	
STI		Açıkdeniz Oligotrofik		Kıyıdanuzak Mezotrofik	Kıyı Ötrofik		
TLI	Ultramikrotrofik	Mikrotrofik	Oligotrofik	Mezotrofik	Ötrofik	Supertrofik	Hipertrofik
	0						7
CTSI			Oligotrofik ≤40	Mezotrofik 40-50	Ötrofik 50-70	Hipertrofik >70	

HEAT ve OSPAR indeksleri özellikle besin zenginliği ve çoğalma durumlarının etkilerini dikkate almaktadır. OSPAR ve HELCOM ötrofikasyon değerlendirmelerinin benzer olduğu görülmektedir. WFD ve HEAT tarafından önerilen şekilde, HEAT indeksinde bir EQR'yi hesaplamak için, son izleme verilerinden sapma ve tipe özgü referans koşul durumları karşılaştırılmaktadır. Sonuçlar, ekolojik durum açısından değerlendirilerek en kötü sonuca sahip parametre üzerinden belirlenir. WFD'nin "One-out all-out" prensibine göre, izleme sonuçlarından bir parametre dahi kötü ise o bölgenin nihai su kalitesi sınıfı o parametrenin sonucuna göre değerlendirilir. OSPAR, kriterlerin her biri için bölgeye özgü ve/veya geçmiş referans verilerini kullanır. Bu yöntemde besin maddelerinin yalnızca kış dönemindeki izleme sonuçlarının kullanılması ve yöntemin açık denizlerde kullanılması, OSPAR'a akit olmayan taraflarca indeksin test edilmesini zorlaştıracağı düşünülmektedir.

Detaylı literatür analizinden anlaşıldığı üzere araştırmacılar, TRIX ve HEAT'in aynı temel yapılara sahip olduğunu ve ötrofikasyon durumu/eğilimi ile ilgili oldukça uyumlu bir sonuç çıkardığını ortaya koymuşlardır. TRIX ve TWQI indekslerine bakıldığında ise, veri girişlerinin TWQI'de belirtilen bentik flora ve aynı zamanda TRIX'te kullanımı zorunlu olmayan seki diski derinliği parametresi haricindeki benzerlikleriyle karşılaştırılabilir oldukları görülebilir. IFREMER ve TWQI sonuçları, daha düşük kaliteye sahip bölgeler dışında tutarlılık göstermektedir. IFREMER sınırlı değişkenlere sahip iken, TWQI bütünleşik değişkenlere dayanır. Diğer taraftan, BEAST yaklaşımı su kalitesi değerlendirme prensibi açısından MSFD gereklilikleriyle eşleşmektedir. BEAST yaklaşımının TRIX'e göre daha fazla ötrofikasyon parametresi ile eşleşmekte olduğu ve aynı zamanda deniz suyu sıcaklığının ötrofikasyon üzerindeki etkisini de dikkate aldığı görülmektedir.

Case Study

Composition and Distribution of Benthic Diatoms in Different Habitats of Burdur River Basin

Burdur Nehir Havzasındaki Farklı Habitatlarda Bentik Diyatome Kompozisyonu ve Dağılımı

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Abstract

Diatoms constitute an essential component for biomonitoring studies to determine the ecological quality of waterbodies. In this study, benthic diatoms of Burdur River Basin were investigated as a consequence of a project on river basin management plan. This is the first detailed taxonomical study of diatoms taken from 13 streams, 4 lakes and 6 reservoirs of Burdur River Basin and a total of 223 taxa have been observed. Among genera, *Navicula* Bory (27) and *Nitzschia* Hassal (27) were represented with the highest numbers of taxa and followed by *Gomphonema* Agardh with a total of 22 species. *Navicula antonii* Lange-Bertalot and *Nitzschia palea* var. *debilis* (Kützing) Grunow had the highest relative abundance with 17.1% and 15.5% respectively. As a closed basin, salinity varied greatly from fresh to saline water between the sampling stations and diatom composition contained species with a different tolerance level. *Navicula antonii* and *Nitzschia frustulum* (Kützing) Grunow observed in all habitats indicating their euryhaline characters; however *Navicula digitoconvergens* was only detected in saline Acıgöl lake with a high relative abundance (38%) in autumn. Other two dominant species were *Halamphora coffeiformis* (Kützing) Levkov and *Navicula cincta* (Ehrenberg) Ralfs in Acıgöl lake showing their tolerance to high salt content as a brackish species. According to first results, 11 species were new records for Turkish diatom flora. The high biodiversity of diatoms revealed the presence of different habitat characteristics within the basin. These results are an important contribution of Turkish diatom flora and could be useful for monitoring specific areas like Burdur River Basin.

Keywords: *Burdur River Basin, diatoms, phytobenthos, salinity tolerance*

Öz

Diyatomeler, su kütlelerinin ekolojik kalitesini belirlemek için yapılan biyolojik izleme çalışmalarında kullanılan önemli bir bileşendir. Bu çalışmada, Burdur Nehir Havzası'nın tatlı su bentik diyatomeleri nehir havzası yönetim planının hazırlanması projesi kapsamında 13 nehir, 4 göl ve 6 rezervuar olmak üzere toplam 23 noktadan örneklenmiştir. Çalışma kapsamında havzada bentik diyatomelerin ilk defa detaylı taksonomik incelenmesi yapılmış olup, toplamda 223 takson gözlenmiştir. Cinsler arasında Navicula Bory (27) ve Nitzschia Hassal (27) en fazla taksonla temsil edilmiş olup, bunu 22 tür ile Gomphonema Agardh takip etmiştir. Navicula antonii Lange-Bertalot ve Nitzschia palea var. debilis (Kützing) Grunow, sırasıyla %17,1 and %15,5 nisbi bolluk değerleri ile en yüksek nispi bolluğa sahip türler olmuştur. Kapalı bir havza olan Burdur Havzası'nda örnekleme alanlarının tuzluluk değerleri tatlı sudan tuzlu suya kadar farklılık göstermiş olup, diyatome kompozisyonu da farklı tuzluluk tolerans seviyelerine sahip türleri içermektedir. Örihalin karaktere sahip olan Navicula antonii ve Nitzschia frustulum bütün habitatlarda gözlenirken, Navicula digitoconvergens sadece tuzlu su karakterine sahip olan Acıgöl'de yüksek nisbi bolluğa (%38) ulaşmıştır. Acıgöl'de tespit edilen diğer iki dominant tür olan Halamphora coffeiformis ve Navicula cincta türleri de acısu karakterindeki sularda artış yaptığı bilinen türlerdir ve bu havzada yüksek tuzluluğa tolerans göstermişlerdir. Elde edilen ilk sonuçlara göre, 11 tür Türkiye diyatome florası için yeni kayıtlardır. Diyatomelerin yüksek biyoçeşitliliği Burdur Nehir Havzası'ndaki farklı habitat karakteristiklerinin varlığını ortaya koymuştur. Bu sonuçlar, Türkiye diyatome florasına önemli bir katkı sağlayacak ve Burdur Nehir Havzası gibi özel alanların izlenmesinde yararlı olacaktır.

Anahtar kelimeler: *Burdur Nehir Havzası, diyatomeler, fitobentoz, tuzluluk toleransı*

Introduction

Amongst the algae groups, diatoms have been a subject of study, especially in recent years due to the biomonitoring of freshwater bodies for water quality assessments. Their prompt response to the environmental changes and their presence throughout the year together with rapid reproduction stages made diatoms an essential tool for monitoring freshwater systems. Studies were conducted in various waterbodies of Turkey to detect the environmental quality of the lakes (Dalkıran et al. 2016; Şanal & Demir, 2018) and rivers (Atıcı, 1997; Karacaoğlu & Dalkıran, 2017; Demir et al. 2017; Solak et al. 2018; Çelekli et al. 2019).

Burdur River Basin is located in the southwest of central Anatolia, and it is one of the smallest basins. In the neighbouring river basins, benthic diatoms had become an interest for the researchers to determine the flora of several streams or lakes. Çiçek & Yamuç (2017) investigated epilithic algae in relation to environmental factors in Kovada Lake in Antalya River Basin. In Büyük Menderes, there were many studies concerning benthic diatoms as well; e.g., Barlas et al. (2002), studied the epilithic algae of Akçapınar Stream and Kadın Azmağı Stream.

Detection of ecological quality of rivers and lakes has become more important for the last decades in Europe, according to Water Framework Directive (WFD) regulations (Ács et al. 2004; Rimet, 2012). Similarly, studies were conducted to use biological components, including benthic diatoms, of aquatic ecosystems to assess the ecological quality based on WFD. As a result, basin-scale studies on diatom biodiversity and ecology have increased recently. Demir et al. (2017) reported diatom composition of Lake Eber and the streams in Akarçay River Basin, while Solak et al. (2018) analysed the distribution of diatoms in streams and reservoirs of Küçük Menderes River. Benthic diatom community of streams, lakes and reservoirs of Gediz River Basin was also studied by Solak et al. (2019). Unlike the other 24 river basins in Turkey, there is no data on benthic diatom composition in the lakes and streams of Burdur River Basin.

The aim of this research is to determine the benthic diatom flora of streams and lakes/reservoirs of Burdur River Basin and provide a taxonomical data for the environmental quality monitoring of the basin according to regulations of WFD (2000/60/EC).

Method

Sampling

Burdur River Basin is located in the southwest of Turkey, covering some natural lakes and wetlands such as Burdur Lake, Acıgöl Lake and Salda Lake (Figure 1). Sampling was carried out in 23 waterbodies including the streams, lakes and reservoirs. A total of 30 sampling points were selected in the basin area; however, sampling could not be performed due to the drought in seven locations (Table 1). Benthic diatom samples were taken twice a year (April and October 2018). According to Communique on Biological Monitoring (T.C. Resmi Gazete, 2019), one sample collected from lakes and reservoirs smaller than 50 ha; 2 samples in areas between 50-500 ha area and 3 samples from the lakes and reservoirs greater than 500 ha were taken. However, all samples were mixed and one subsample was prepared for each lake and reservoir. Epilithic diatom samples were collected from the submerged stones, and epiphytic samples were collected from macrophytes. Physicochemical measurements were performed monthly in 2018.

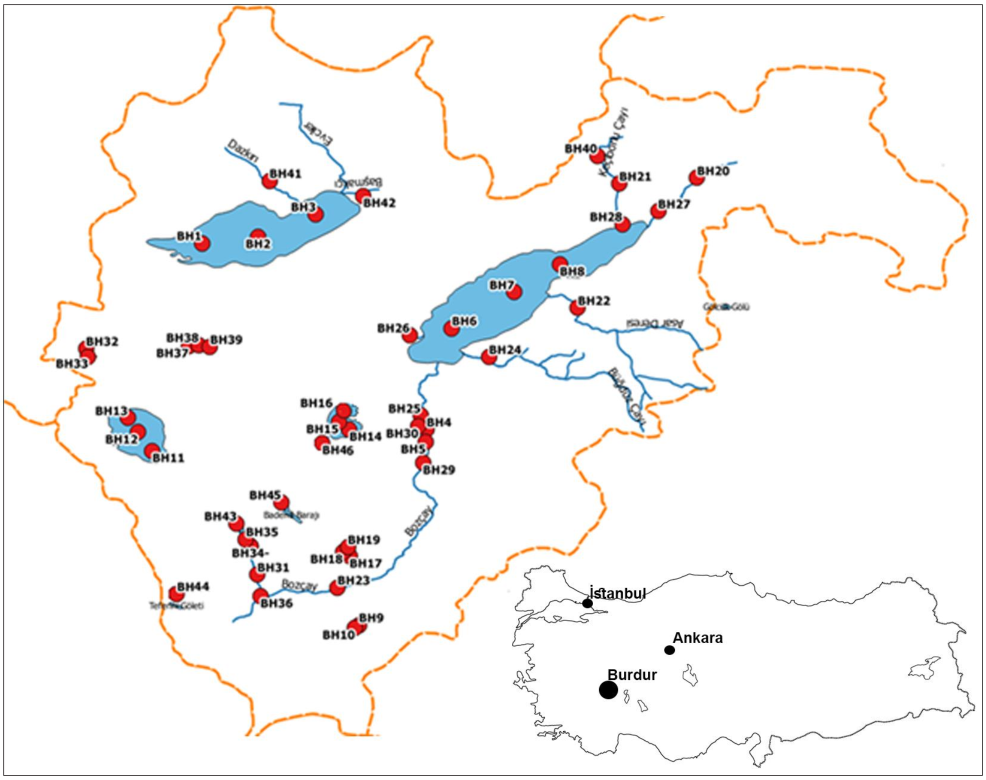

Figure 1. Study area and sampling stations.

Diatom Analysis

Samples were cleaned from organic material by boiling with H_2O_2 and washed by distilled water several times. In order to remove the carbonates, samples were treated with 10% HCl. Frustules were air-dried and mounted in Naphrax®. Zeiss Axio Observer Z1 (Carl Zeiss microscopy Gmbh, Jena, Germany) microscope was used for light microscopy (LM) observations at Limnology Laboratory, Department of Freshwater Resource and Management, Istanbul University.

Diatom valves were identified according to the following literature; Krammer & Lange-Bertalot (1986, 1988, 1991a,b), Hofmann et al. (2011) and Kulikovskiy et al. (2016). Taxonomic classification and nomenclatural on genera and taxa names follow the latest updates from Guiry & Guiry (2019) and Kociolek et al. (2019). Slides and processed materials were deposited at the collection of the Department of Freshwater Resource and Management, Istanbul University and the Ministry of Agriculture and Forestry archives.

The relative abundance of the species was expressed as percentages of the total number of frustules counted. The relative abundance (RA) of particular taxa and the taxa richness of the assemblages were estimated on the basis of at least 300 diatom valves counted per sample. The RA of the species identified in lakes, reservoirs and streams were determined separately and the species constituting higher than 5% were evaluated in Table 4.

Table 1

Burdur River Basin Sampling Coordinates

Code	Name	Category	Salinity	Province	Coordinates	
					X	Y
BH1	Acıgöl	Lake	Saline (39‰)	Afyon	37.81844	29.77163
BH2					37.82721	29.84914
BH3					37.85838	29.92875
BH4	Karaçal	Reservoir	Freshwater	Burdur	37.56175	30.08104
BH5					37.54444	30.08007
BH6	Burdur	Lake	Brackish (18.2‰)	Burdur	37.70125	30.11597
BH7					37.75088	30.20211
BH8					37.78928	30.26449
BH9	Belenli	Reservoir	Freshwater	Burdur	37.29235	29.98792
BH10					37.28849	29.98258
BH11	Salda	Lake	Brackish (1.1 ‰)	Burdur	37.53238	29.70349
BH12					37.55788	29.68351
BH13					37.57749	29.66961
BH14	Yarışlı	Lake (Dry)	-	Burdur	37.56182	29.97482
BH15					37.57202	29.96020
BH16					37.58831	29.96729
BH17	Karataş	Lake	Freshwater	Burdur	37.38701	29.97508
BH18					37.39292	29.96579
BH19					37.39989	29.97259
BH20	Gönen	Stream (Dry in October)	Freshwater	Isparta	37.90820	30.45358
BH21	Çukurharman	Stream (Dry)	-	Isparta	37.90046	30.34642
BH22	Asar	Stream	Brackish (0.53 ‰)	Burdur	37.72920	30.28935
BH23	Bozçay	Stream	Freshwater	Burdur	37.34401	29.95839

Code	Name	Category	Salinity	Province	Coordinates	
					X	Y
BH24	Büğdüz	Stream (Dry in October)	Freshwater	Burdur	37.66174	30.16703
BH25	Bozçay	Stream	Freshwater	Burdur	37.58073	30.07316
BH26	Ulupınar	Stream (Dry)	-	Burdur	37.69160	30.05794
BH27	Gönen	Stream	Brackish (1.9 ‰)	Isparta	37.86231	30.39985
BH28	Çukurharman	Stream	Brackish (0.7‰)	Isparta	37.84485	30.35068
BH29	Bozçay	Stream	Freshwater	Burdur	37.51586	30.07599
BH30	Bozçay	Stream	Freshwater	Burdur	37.56599	30.07000
BH31	Karamanlı	Stream (Dry)	-	Burdur	37.36173	29.84796
BH32	Beylerli	Reservoir	Freshwater	Denizli	37.67316	29.61199
BH33					37.66217	29.61534
BH34	Karamanlı	Reservoir	Freshwater	Burdur	37.40121	29.83880
BH35					37.41066	29.83226
BH36	Sarı	Stream (Dry)	-	Burdur	37.33204	29.85228
BH37	Akgöl	Lake (Dry)	-	Burdur	37.67674	29.75361
BH38					37.67821	29.76745
BH39					37.67559	29.78304
BH40	Keçiborlu	Stream (Dry in April)	Brackish (1.45 ‰)	Isparta	37.93891	30.31606
BH41	Dazkırı	Stream (Dry)	-	Afyon	37.90446	29.86472
BH42	Başmakçı	Stream (Dry in April)	Brackish (6 ‰)	Afyon	37.88374	29.99386
BH43	Özdere	Stream	Freshwater	Burdur	37.43224	29.81980
BH44	Tefenni	Reservoir	Freshwater	Burdur	37.31703	29.74366
BH45	Bademli	Reservoir	Freshwater	Burdur	37.43503	29.90621
BH46	Yarışlı	Stream	Freshwater	Burdur	37.43503	29.90621

Results

Physicochemical Parameters

The physicochemical measurements revealed that reservoirs, lakes and streams in the basin were generally alkaline. The mean conductivity values were generally low in reservoirs and streams (0.5 and 1.7 mS cm^{-1}, respectively) but higher in lakes (22.3 mS cm^{-1}). Notably, in Burdur and Acıgöl lakes, conductivity was higher throughout the year, and the mean conductivity values were 29.4 mS cm^{-1} and 57.2 mS cm^{-1} respectively. Significant changes were observed in dissolved oxygen values in reservoirs, lakes and streams throughout the year. Although the lowest dissolved oxygen (DO) values were observed in Burdur Lake as 2.7 mg L^{-1} and Acıgöl Lake as 1 mg L^{-1}, the average values were between 6.9 and 7.8 mg L^{-1} in the reservoirs and lakes (Table 2).

As a closed basin, salinity varied remarkably between freshwater to saline among studied areas. While all reservoirs were classified as freshwater (<0.5‰), streams were categorized between fresh to brackish water (<0.5 - 6‰). On the other hand, salinity variation was higher in lakes, from freshwater (Karataş Lake, <0.5‰) to brackish (Salda Lake, 1‰ and Burdur Lake, 18‰) and even saline environment (Acıgöl Lake, 39 ‰).

Table 2

Mean, Min and Max Values of Selected Water Quality Parameters Measured Studied Areas

	Temperature (°C)			pH			Conductivity (mS cm^{-1})			Dissolved Oxygen (mg L^{-1})			Salinity (‰)		
	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
Reservoirs	13.7	3.8	24.0	8.5	7.4	9.3	0.5	0.2	0.6	6.9	3.0	11.4	<0,5	<0,5	<0,5
Lakes	15.2	5.1	29.0	8.8	6.4	9.6	22.3	0.5	69.6	7.8	1.0	11.4	19.4	<0,5	47.6
Streams	15.8	2.2	27.9	8.3	7.0	9.7	1.7	0.3	12.9	6.9	0.5	12.7	2.7	<0,5	7.5

Diatom Composition

Composition and distribution of diatoms have been studied in two seasons, and a total of 223 taxa belonging to 57 genera were identified; within these taxa, 11 were identified in genera level. The seasonal composition of species were presented according to sampling habitats; streams, lakes and reservoirs (Table 3).

Table 3

Diatom Composition and Distribution in Streams, Lakes and Reservoirs of Burdur River Basin

Taxa	Stream		Lake		Reservoir	
	Sp.	Aut.	Sp.	Aut.	Sp.	Aut.
Achnanthes petersenii Hustedt	+					
Achnanthidium eutrophilum (Lange-Bertalot) Lange-Bertalot	+					+
Achnanthidium minutissimum (Kützing) Czarnecki	+	+			+	
Achnanthidium minutissimum var. jackii (Rabenhorst) Lange-Bertalot	+	+	+	+	+	+
Adlafia minuscula var. *muralis* (Grunow) Lange-Bertalot		+				
Amphora aequalis Krammer		+				
Amphora alpestris Levkov		+				
Amphora copulata (Kützing) Schoeman & R.E.M. Archibald		+				+
Amphora inariensis Krammer		+				+
Amphora indistincta Levkov						+
Amphora lange-bertalotii Levkov & Metzeltin						+
Amphora ovalis (Kützing) Kützing		+			+	+
Amphora pediculus (Kützing) Grunow	+	+			+	+
Amphora stechlinensis Levkov & Metzeltin	+					+
Anomoeoneis sphaerophora Pfitzer		+				
Aulacoseira ambigua (Grunow) Simonsen					+	+
Aulacoseira italica (Ehrenberg) Simonsen	+					
Berkeleya sp.		+	+	+		+
Caloneis amphisbaena (Bory) Cleve		+				
Caloneis bacillum (Grunow) Cleve					+	
Caloneis silicula (Ehrenberg) Cleve			+			
Cocconeis lineata Ehrenberg	+	+		+	+	+

Taxa	Stream		Lake		Reservoir	
	Sp.	Aut.	Sp.	Aut.	Sp.	Aut.
Cocconeis pediculus Ehrenberg	+	+				
Cocconeis placentula var. *placentula* Ehrenberg	+	+		+	+	+
Craticula accomoda (Hustedt) D.G. Mann						+
Craticula ambigua (Ehrenberg) D.G. Mann	+	+		+	+	+
Craticula buderi (Hustedt) Lange-Bertalot*	+	+				
Craticula cuspidata (Kützing) D.G. Mann	+					
Craticula subminuscula (Manguin) Wetzel & Ector	+	+				
Craticula sp.		+				
Cyclostephanos dubius (Hustedt) Round		+				
Cyclostephanos invisitatus (Hohn& Hellermann) Theriot, Stoermer & Håkasson						+
Cyclotella meneghiniana Kützing	+	+			+	
Cymbella affinis Kützing	+					+
Cymbella cymbiformis Agardh	+	+	+	+	+	+
Cymbella dorsenotata Østrup	+					+
Cymbella excisa Kützing		+				
Cymbella helvetica Kützing						+
Cymbella lanceolata (Agardh) Agardh			+			
Cymbella lange-bertalotii Krammer*		+			+	+
Cymbella neocistula Krammer						+
Cymbella vulgata Krammer		+				
Cymbopleura amphicephala (Nägeli) Krammer		+	+		+	+
Cymbopleura inaequalis (Ehrenberg) Krammer						+
Cymbopleura rhomboidea Krammer						+
Cymbellafalsa diluviana (Krasske) Lange-Bertalot & Metzeltin	+					
Diatoma moniliformis (Kützing) D.M. Williams	+			+	+	
Diatoma tenuis C. Agardh		+				
Diatoma vulgaris Bory	+	+				
Diploneis elliptica (Kützing) Cleve					+	
Diploneis krammeri Lange-Bertalot & E. Reichardt	+				+	
Diploneis parma Cleve		+				
Dorofeyukea kotschyi Kulikovskiy et al.		+			+	

Taxa	Stream		Lake		Reservoir	
	Sp.	Aut.	Sp.	Aut.	Sp.	Aut.
Encyonema caespitosum Kützing	+	+	+		+	+
Encyonema hebridicum Grunow ex Cleve						+
Encyonema lacustre (C. Agardh) Pantocsek		+				+
Encyonema lange-bertalotii Krammer		+				
Encyonema latum Krammer		+				
Encyonema leibleinii (C. Agardh) Silva et al.			+		+	
Encyonema minutum (Hilse) D.G. Mann		+			+	+
Encyonema silesiacum (Bleisch) D.G. Mann		+	+	+	+	+
Encyonema ventricosum (C. Agardh) Grunow	+				+	+
Encyonema vulgare Krammer*					+	+
Encyonopsis cesatii (Rabenhorst) Krammer		+			+	
Encyonopsis microcephala (Grunow) Krammer		+		+	+	
Encyonopsis minuta Krammer & E. Reichardt				+	+	
Encyonopsis subminuta Krammer & E. Reichardt	+	+		+	+	+
Encyonopsis sp.		+				
Epithemia adnata (Kützing) Brébisson		+				
Epithemia gibba (Ehrenberg) Kützing		+				+
Epithemia smithii Carruthers				+		+
Epithemia sorex Kützing		+		+		
Fallacia pygmaea (Kützing) Stickle & D.G. Mann	+	+	+	+	+	
Fragilaria gracilis Østrup						+
Fragilaria henryi Lange-Bertalot					+	
Fragilaria pararumpens Lange-Bertalot, G. Hofmann & Werum				+		
Fragilaria radians (Kützing) D.M. Williams & Round	+	+				
Fragilaria tenera var. *nanana* (Lange-Bertalot) Lange-Bertalot & S. Ulrich		+				
Fragilaria vaucheriae (Kützing) J.B. Petersen	+		+	+	+	+
Frustulia sp.					+	
Geissleria decussis (Østrup) Lange-Bertalot & Metzeltin						+
Gomphonema auritum A. Braun ex Kützing*					+	
Gomphonema calcareum Cleve				+		
Gomphonema clavatum Ehrenberg	+	+				

Taxa	Stream		Lake		Reservoir	
	Sp.	Aut.	Sp.	Aut.	Sp.	Aut.
Gomphonema drutelingense E. Reichardt*	+					
Gomphonema exilissimum (Grunow) Lange-Bertalot & E. Reichardt*		+				+
Gomphonema innocens E. Reichardt	+					
Gomphonema italicum Kützing		+				+
Gomphonema lippertii E. Reichardt & Lange-Bertalot		+				
Gomphonema minusculum Krasske		+				
Gomphonema minutum (C. Agardh) C. Agardh		+				
Gomphonema olivaceum (Hornemann) Brébisson	+	+	+	+	+	+
Gomphonema pala E. Reichardt		+				
Gomphonema parvulum (Kützing) Kützing	+	+			+	
Gomphonema pseudoaugur Lange-Bertalot	+	+				
Gomphonema pumilum (Grunow) E. Reichardt & Lange-Bertalot	+	+		+		+
Gomphonema pumilum var. *rigidum* E. Reichardt & Lange-Bertalot		+				
Gomphonema rhombicum Fricke					+	
Gomphonema saprophilum (Lange-Bertalot & E. Reichardt) Abraca et al.	+					
Gomphonema subclavatum (Grunow) Grunow		+				
Gomphonema tergestinum (Grunow) Fricke						+
Gomphonema truncatum Ehrenberg			+		+	
Gomphonema sp.		+				
Gyrosigma acuminatum (Kützing) Rabenhorst					+	
Gyrosigma attenuatum (Kützing) Rabenhorst		+				
Halamphora coffeiformis (C. Agardh) Levkov		+	+	+		+
Halamphora veneta (Kützing) Levkov		+		+		+
Hantzschia abundans Lange-Bertalot	+				+	
Hantzschia amphioxys (Ehrenberg) Grunow		+				
Hippodonta capitata (Ehrenberg) Lange-Bertalot, Metzeltin & Witkowski		+			+	
Hippodonta hungarica (Grunow) Lange-Bertalot, Metzeltin & Witkowski				+		
Lemnicola exigua (Grunow) Kulikovskiy, Witkowski & Plinski	+				+	
Lemnicola hungarica (Grunow) Round & Basson		+				

Taxa	Stream		Lake		Reservoir	
	Sp.	Aut.	Sp.	Aut.	Sp.	Aut.
Lindavia balatonis (Pantocsek) Nakov et al.					+	+
Luticola ventricosa (Kützing) D.G. Mann	+				+	
Mastogloia elliptica (C. Agardh) Cleve		+				
Mastogloia smithii Thwaites ex W. Smith	+	+	+	+		
Mastogloia cf. *pseudosmithii* Lee et al.			+			
Melosira varians C. Agardh		+			+	
Navicymbula pusilla (Grunow) Krammer	+	+	+	+	+	+
Navicula antonii Lange-Bertalot	+	+	+	+	+	+
Navicula capitatoradiata H. Germain	+	+	+	+	+	+
Navicula cari Ehrenberg		+				+
Navicula cincta (Ehrenberg) Ralfs		+	+		+	
Navicula cryptocephala Kützing	+	+				+
Navicula cryptotenella Lange-Bertalot	+	+	+	+		+
Navicula digitoconvergens Lange-Bertalot				+		
Navicula erifuga Lange-Bertalot	+					
Navicula gottlandica Grunow					+	
Navicula gregaria Donkin						+
Navicula hanseatica Lange-Bertalot & Stachura					+	
Navicula lanceolata Ehrenberg	+				+	
Navicula menisculus Schumann	+				+	+
Navicula metareichardtiana Lange-Bertalot & Kusber	+	+				
Navicula notha J.H. Wallace	+				+	+
Navicula phyllepta Kützing		+				
Navicula rhynchotella Lange-Bertalot	+					
Navicula rostellata Kützing					+	
Navicula simulata Manguin		+	+			
Navicula striolata (Grunow) Lange-Bertalot			+			
Navicula tripunctata (O.F. Müller) Bory	+	+				+
Navicula trivialis Lange-Bertalot						+
Navicula upsaliensis (Grunow) M. Peragallo			+			
Navicula vandamii Schoeman & R.E.M. Archibald		+				
Navicula veneta Kützing	+	+				

Taxa	Stream		Lake		Reservoir	
	Sp.	Aut.	Sp.	Aut.	Sp.	Aut.
Navicula viridula var. *germainii* (Wallace) Lange-Bertalot		+		+		+
Navicula sp.				+		
Neidium affine (Ehrenberg) Pfitzer						+
Nitzschia alpina Hustedt	+	+			+	+
Nitzschia amphibia Grunow	+	+	+		+	
Nitzschia bulnheimiana (Rabenhorst) H.L. Smith*		+				
Nitzschia capitellata Hustedt	+	+				+
Nitzschia denticula Grunow	+	+			+	+
Nitzschia desertorum Hustedt				+		
Nitzschia dissipata (Kützing) Rabenhorst	+	+	+		+	+
Nitzschia filiformis (W. Smith) Van Heurck		+				
Nitzschia fonticola (Grunow) Grunow	+			+		+
Nitzschia frustulum (Kützing) Grunow	+	+	+	+		+
Nitzschia gracilis Hantzsch				+		
Nitzschia hantzschiana Rabenhorst	+	+			+	
Nitzschia heufleriana Grunow	+				+	
Nitzschia inconspicua Grunow	+	+	+	+		+
Nitzschia linearis W. Smith	+	+	+		+	
Nitzschia palea (Kützing) W. Smith	+	+		+	+	+
Nitzschia palea var. *debilis* (Kützing) Grunow		+				+
Nitzschia palea var. *minuta* (Bleisch) Grunow	+					
Nitzschia pusilla Grunow	+			+		
Nitzschia recta Hantzsch ex Rabenhorst					+	
Nitzschia rosenstockii Lange-Bertalotii	+					
Nitzschia sociabilis Hustedt		+				+
Nitzschia solita Hustedt	+			+		
Nitzschia supralitorea Lange-Bertalot		+				+
Nitzschia tenuis W. Smith	+					
Nitzschia tubicola Grunow		+				
Nitzschia umbonata (Ehrenberg) Lange-Bertalot		+				
Pantocsekiella iranica (Nejdsattari et al.) Kiss et al.		+		+		+
Pantocsekiella ocellata (Pantocsek) Kiss & Ács	+	+			+	+

Taxa	Stream		Lake		Reservoir	
	Sp.	Aut.	Sp.	Aut.	Sp.	Aut.
Paraplaconeis minor (Grunow) Lange-Bertalot*					+	
Paraplaconeis placentula (Ehrenberg) Kulikovskiy & Lange-Bertalot	+			+		
Pinnularia brebissonii (Kützing) Rabenhorst	+	+			+	
Pinnularia divergens W. Smith					+	+
Pinnularia suchlandtii Hustedt			+		+	
Pinnularia sp.		+				
Placoneis anglophila (Lange-Bertalot) Lange-Bertalot*					+	
Placoneis clementis (Grunow) E.J. Cox						+
Placoneis clementioides (Hustedt) E.J. Cox *					+	
Placoneis ignorata (Schimanski) Lange-Bertalot						+
Placoneis sp.						+
Planothidium frequentissimum (Lange-Bertalot) Lange-Bertalot	+	+	+	+		
Planothidium lanceolatum (Brébisson ex Kützing) Lange-Bertalot		+				
Planothidium rostratum (Østrup) Lange-Bertalot	+					
Pseudofallacia monoculata (Hustedt) Liu, Kociolek & Wang	+					
Pseudostaurosira brevistriata (Grunow) D.M. Williams & Round	+	+				
Reimeria sinuata (W. Gregory) Kociolek & Stoermer						+
Rhopalodia gibberula (Ehrenberg) Otto Müller		+		+		+
Sellaphora absoluta (Hustedt) Wetzel et al.						+
Sellaphora pupula (Kützing) Mereschkovsky	+	+	+	+	+	+
Sellaphora sp.				+		
Stauroneis acidoclinata Lange-Bertalot & Werum*				+	+	
Stauroneis gracilis Ehrenberg			+			
Staurosira dubia Grunow					+	
Staurosira venter (Ehrenberg) Cleve & J.D. Möller				+		
Staurosirella pinnata (Ehrenberg) D.M. Williams & Round		+				
Stephanodiscus astraea (Kützing) Grunow			+		+	
Surirella amphioxys W. Smith						+
Surirella angusta Kützing		+				+
Surirella brebissonii Krammer & Lange-Bertalot	+	+				

Taxa	Stream		Lake		Reservoir	
	Sp.	Aut.	Sp.	Aut.	Sp.	Aut.
Surirella librile (Ehrenberg) Ehrenberg		+			+	
Surirella minuta Brébisson ex Kützing	+	+				
Surirella ovalis Brébisson	+	+				
Surirella robusta Ehrenberg	+					+
Surirella subsalsa W. Smith	+					
Tabularia fasciculata (C. Agardh) D.M. Williams & Round	+					
Tryblionella angustata W. Smith	+	+				
Tryblionella apiculata W. Gregory	+	+	+	+		
Tryblionella brunoi (Lange-Bertalot) Cantonati & Lange-Bertalot						+
Tryblionella calida (Grunow) D.G. Mann		+				
Tryblionella hungarica (Grunow) Frenguelli	+	+				
Ulnaria acus (Kützing) Aboal	+	+			+	+
Ulnaria biceps (Kützing) Compère	+	+			+	
Ulnaria delicatissima (W. Smith) Aboal & P.C. Silva		+			+	+
Ulnaria ulna (Nitzsch) Compère	+	+			+	

Note. Sp= Spring*, Aut*= Autumn*,* * =New records.

Amongst the diatom genera, *Navicula* Bory and *Nitzschia* Hassall were represented with the highest numbers of taxa (27) in the river basin, this was followed by *Gomphonema* Agardh (22), *Encyonema* Kützing (10) and *Cymbella* Agardh (9). 20 genera were represented with only one species. 136 species were identified in spring, while the number of species increased to 174 in autumn. The species number varied between the environments; 131 taxa were observed in reservoir samples, while 61 and 160 taxa found in lake and stream samples, respectively. The taxa numbers also showed a variance in different habitats between spring and autumn. Total 78 taxa found in spring whereas 86 taxa were detected in autumn in the reservoirs. There were 34 taxa in spring with an increase to 45 taxa in autumn in lakes, and the highest species diversity was found in streams, 90 and 124 taxa in spring and autumn, respectively.

The most abundant species was *Navicula antonii* (17.1%) in the whole basin. The other common taxa (≥5%) observed were *Nitzschia palea* var. *debilis* (15.5%), *Tabularia fasciculata* (6.6%) *Cyclotella meneghiniana* (5.6%), *Nitzschia frustulum* (5.4%), and *Amphora pediculus* (5.3%). The remaining numbers of species showed less than 5% occurrences in the samples (Figure 2;3;4).

The most abundant species in the studied areas differed significantly according to the sampling period (Table 4). The number of species which exceeded 5% was higher in spring than in autumn. Diatom material observed in the reservoir samples showed that *Pantocsekiella iranica* (in autumn), *P. ocellata* and *Ulnaria delicatissima* (in spring) were the most abundant species (14%, 17% and 17%, respectively). In the lake samples, abundant species changed and *Achnanthidium minutissimum* var. *jackii* (in autumn) and *Encyonema caespitosum* (in spring) were the most abundant taxa (20% and 35%). On the other hand, especially in autumn, only two species, *Nitzschia palea* var. *debilis* (38%) and *Navicula antonii* (38%), were dominant in the stream samples and other species remained below total RA 5%.

Table 4

Species Identified Above 5% of the Total RA in Reservoirs, Lakes And Streams

	Spring (%)		Autumn (%)	
Reservoirs	*Pantocsekiella ocellata*	17	*Pantocsekiella iranica*	14
	Ulnaria delicatissima	17	*Encyonema lacustre*	9
	Cymbella cymbiformis	13	*Nitzschia palea*	6
	Aulacoseira ambigua	10	*Amphora pediculus*	5
	Encyonema ventricosum	5		
Lakes	*Encyonema caespitosum*	35	*Achnanthidium minutissimum* var. *jackii*	20
	Fragilaria vaucheriae	19	*Encyonopsis subminuta*	17
	Navicymbula pusilla	19	*Gomphonema calcareum*	11
	Navicula capitatoradiata	9	*Nitzschia fonticola*	7
			Mastogloia smithii	6
			Berkeleya sp.	8
Streams	*Tabularia fasciculata*	15	*Nitzschia palea* var. *debilis*	38
	Cyclotella meneghiniana	13	*Navicula antonii*	38
	Amphora pediculus	11		
	Nitzschia frustulum	11		
	Nitzschia dissipata	7		
	Nitzschia palea	7		

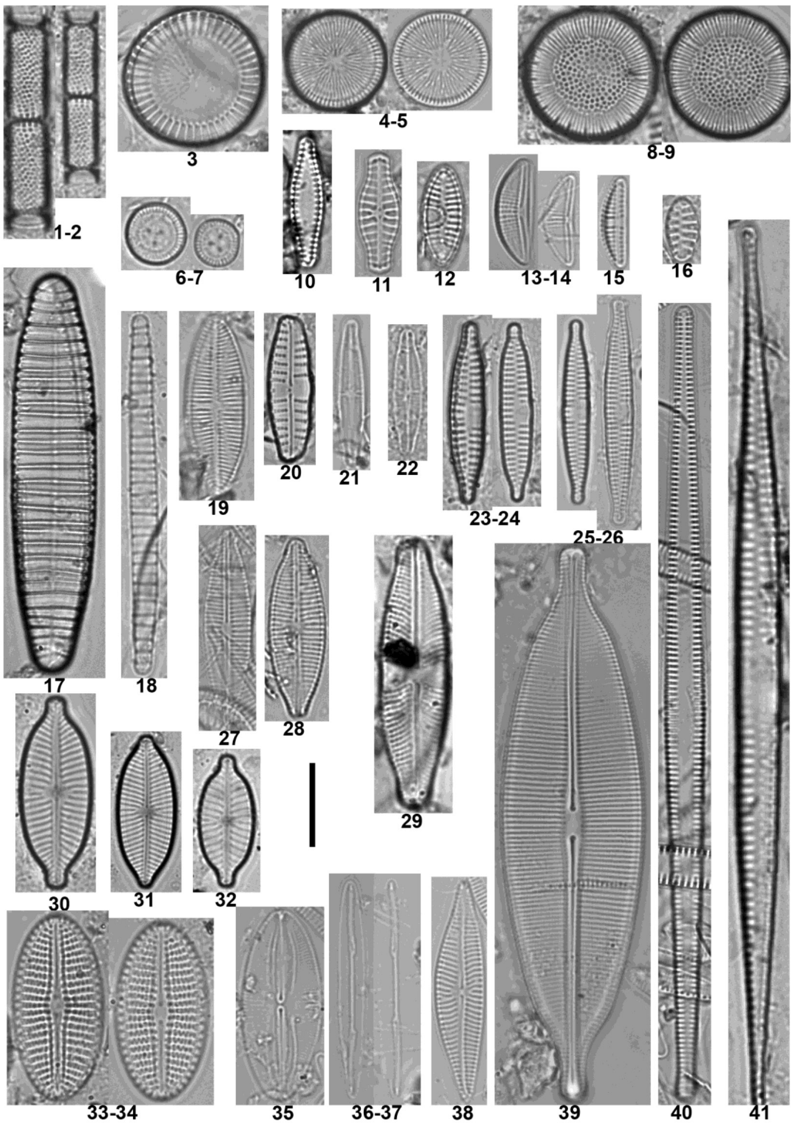

Figure 2. Common Diatoms of Burdur River Basin **1-2.** *Aulacoseira ambigua*; **3.** *Cyclotella meneghiniana*; **4-5.** *Pantocsekiella iranica*; **6-7**. *P. oceallata*; **8-9.** *Lindavia balatonis*; **10**. *Pseudostaurosira brevistriata*; **11.** *Hippodonta capitata*; **12.** *Planothidium frequentissimum*; **13-14.** *Halamphora veneta*; **15.** *Amphora pediculus*; **16.** *Staurosirella pinnata*; **17.** *Diatoma vulgaris*; **18.** *D. moniliformis*; **19.** *Lemnicola hungarica*; **20.** *Reimeria sinuata*; **21.** *Achnanthidium minutissimum*; **22.** *A. minutissimum* var. *jackii*; **23-24.** *Fragilaria vaucheriae*; **25-26.** *F. radians*; **27.** *Craticula buderi*; **28.** *C. accomoda*; **29.** *Stauroneis acidoclinata*; **30.** *Placoneis clementoides*; **31.** *Paraplaconeis minor*; **32.** *Placoneis anglophila*; **33-34.** *Diploneis parma*; **35.** *Fallacia pygamea*; **36-37.** *Berkeleya* sp.; **38.** *Navicymbula pusilla*; **39.** *Craticula ambigua*; **40.** *Tabularia fasciculata*; **41.** *Ulnaria acus*. Scale bar: 10 µm.

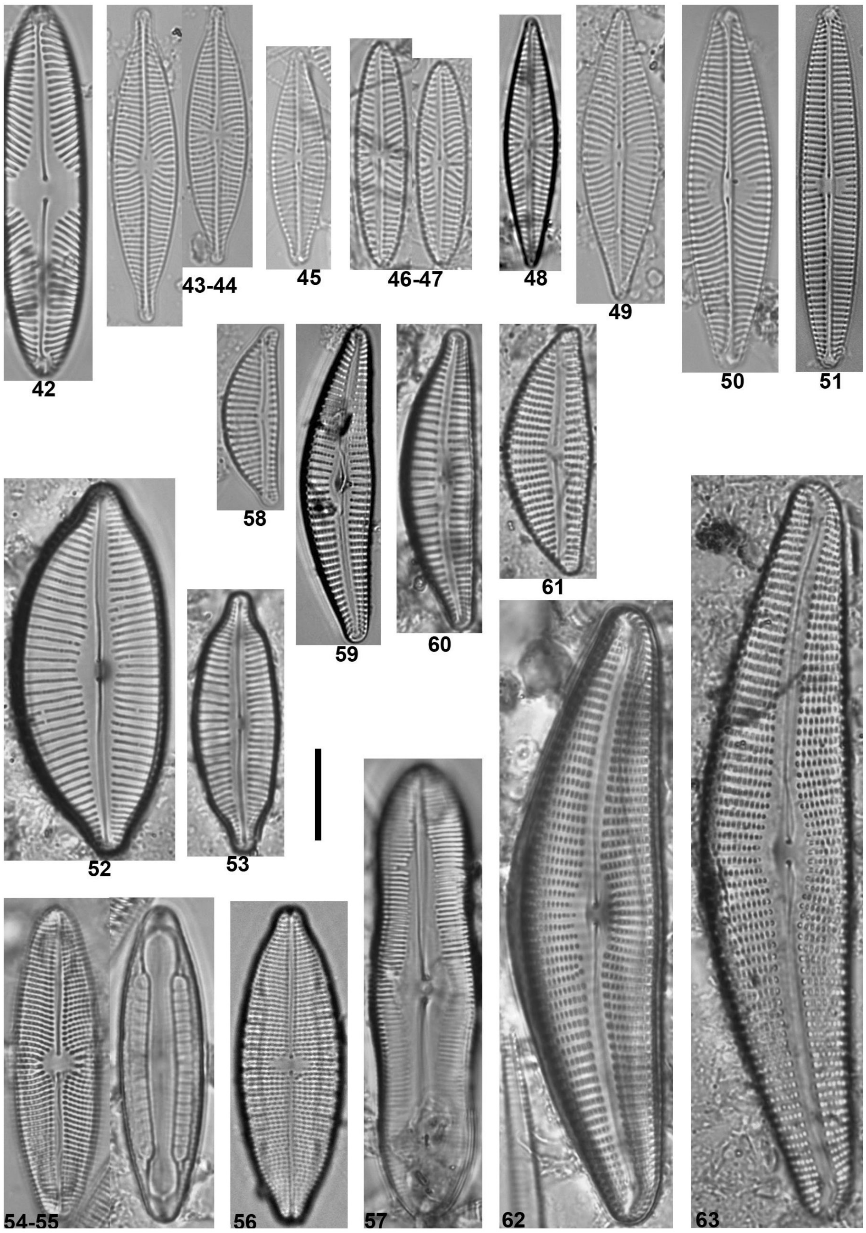

Figure 3. **42.** *Pinnularia brebissonii*; **43-44**. *Navicula capitatoradiata*; **45.** *N. veneta*; **46-47.** *N. cincta*; **48.** *N. cryptotenella*; **49.** *N. trivialis*; **50.** *N. exilis*; **51.** *N. tripunctata*; **52.** *Cymbopleura amphicephala*; **53.** *C. lata*; **54-55.** *Mastogloia elliptica*; **56.** *M. smithii*; **57.** *Pinnularia silicula*; **58.** *Encyonema ventricosum*; **59.** *E. vulgare*; **60.** *Cymbella affinis*; **61.** *Encyonema caespitosum*; **62.** *Cymbella lange-bertalotii*; **63.** *C. cymbiformis*. Scale bar: 10 µm.

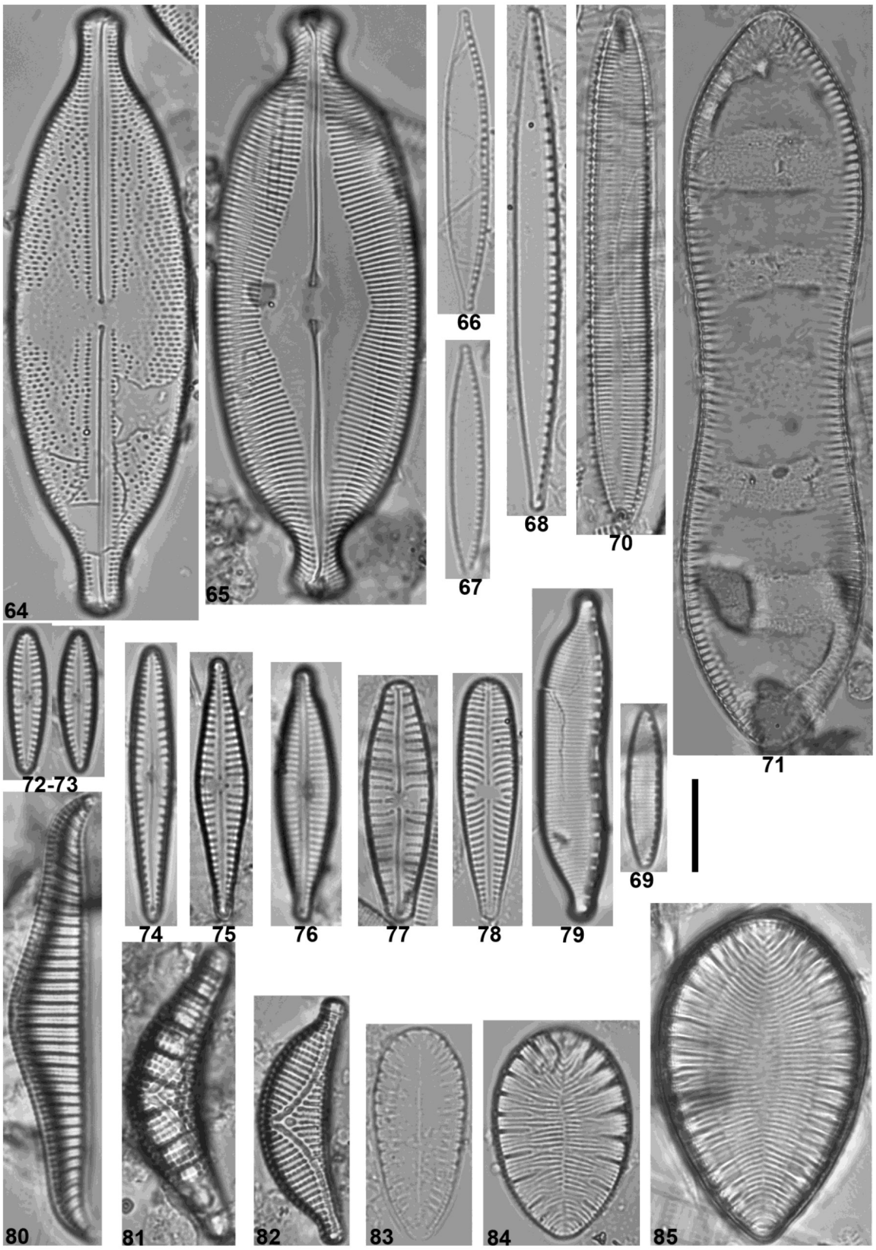

Figure. 4. **64.** *Anomoeoneis sphaerophora*; **65.** *Caloneis amphisbaeana*; **66.** *Nitzschia palea*; **67.** *N. palea* var. *debilis*; **68.** *N. recta*; **69.** *N. bulnheimiana*; **70.** *Tryblionella hungarica*; **71.** *Surirella librile*; **72-73**. *Gomphonema pumilum* var. *rigidum*; **74.** *G. pumilum*; **75.** *G. auritum*; **76.** *G. exilissimum*; **77.** *G. drutelingense*; **78.** *G. olivaceum*. **79.** *Hantzschia amphioxys*; **80.** *Epithemia gibba*; **81.** *E. smithii*; **82.** *E. sorex*; **83.** *Surirella minuta*; **84.** *S. brebissonii*; **85.** *S. ovalis*. Scale bar: 10 µm.

Discussion and Conclusion

Burdur River Basin is one of the two smallest basins out of 25 in Turkey. In this research, diatom composition and its distribution through the streams, lakes and reservoirs of the basin were studied for the first time in detail, and the diversity was found relatively higher in comparison with different river basins. Solak et al. (2018) found 94 taxa from Küçük Menderes River Basin, Çelekli et al. (2018) reported 80 taxa from the North Aegean catchment while 148 taxa reported from western Black Sea River catchment (Özer et al., 2018) and Demir et al. (2017) observed 64 diatom taxa from Akarçay River Basin. Similarly, 65 diatom taxa were found in Aras River catchment (Çelekli et al., 2019). The highest number of taxa observed in Burdur area could be the result of the variation of physico-chemical and geological characteristics of the aquatic ecosystems in the basin. The lower diversity was also detected in several diatom composition studies conducted in some lakes and rivers, Çiçek & Yamuç (2017) found out 42 diatom taxa in Eğirdir Lake, Isparta Province and Karacaoğlu & Dalkıran, (2017) detected 134 taxa in Nilüfer Stream and Şanal & Demir (2018) studied epiphytic samples of Lake Mogan and 58 diatoms species were observed. Since the samples were taken seasonally or even monthly in these studies, a more diverse diatom community was typically expected in comparison to the present study. However, focusing on one type of ecosystem (lake or river) and relatively similar characteristics of sampling points compared to whole basin studies could be the reason for detecting lower diversity of diatoms.

Diatom diversity of Burdur River Basin varied among streams (160 taxa), lakes (61 taxa) and reservoirs (131 taxa). Besides, the diatom community structure also differed among the habitats (Table 4). *Pantocsekiella ocellata, P. iranica* and *Ulnaria delicatissima* were dominant species in the reservoirs, but did not observe as dominant species in other habitats.

The seasonal changes of dominant taxa were also remarkable in the same habitat. The most distinct variation was in streams. Dominant taxa number was six in spring and declined to two species in autumn when *Navicula antonii* (37.7 %) and *Nitzschia palea* var. *debilis* (37.8 %) were dominant and constituted 74.4% of total abundance.

Navicula Bory and *Nitzschia* Hassall are generally the most diverse and widespread genera in freshwater diatoms (Karacaoğlu & Dalkıran, 2017; Kociolek et al. 2019). In Burdur River Basin, the results are consisted with this general trend and *Navicula* and *Nitzschia* species diversity were high; however, their relative abundances were low. Aside from the dominant taxa, *Navicula antonii* and *Nitzschia palea* var. *debilis*, only *Navicula capitatoradiata*, *Nitzschia dissipata*, *N. frustulum* and *N. palea* were represented over 5%.

Diatoms are essential tools for bio-assessment of aquatic ecosystems and the identification of the taxa together with its ecological requirements would contribute further to the detection of water quality. Burdur River Basin is a closed basin, which has no connection to the sea and salinity gradient of the habitats were very broad. Therefore, the species composition found in the basin comprised the species with different tolerances to the salinity. Some species prefer waters with high salinity, while others may have a wide salinity tolerance (Schröder et al., 2015). In the present study, we detected *Navicula antonii* and *Nitzschia frustulum* in all habitats with different salinity. Although *N. antonii* and *N. frustulum* species are generally defined as freshwater species, they have also been identified in Ebro Estuary which is a salt wedge estuary in the Mediterranean (Rovira et al., 2009; Costa-Böddeker et al., 2017). Our results confirmed their euryhaline characteristics based on their presence in the basin from fresh to saline habitats.

Two unique habitats related to salinity in the basin are Burdur and Acıgöl Lakes. Our observations indicated that some taxa which are present in brackish and marine waters found in these sampling areas. Specifically, the brackish species, *Halamphora coffeiformis* and *Navicula cincta* were dominant in Acıgöl Lake and their relative abundance was 69% for *N. cincta* in spring and 40% for *H. coffeiformis* in autumn. Similarly, species such as *Berkeleya* sp., *H. coffeiformis, Navicula simulata, Tryblionella apiculata*, which are known to be found in both marine and brackish waters, also showed significant presence in Burdur Lake. Some taxa like *Tabularia fasciculata* which is common in the marine coastal areas (Baytut and Gönülol, 2016) were observed in the river basin with an accomplice of a high number of taxonomically complex taxa (*Navicula*, *Nitzschia*, *Tryblionella*, *Surirella*). Species like *Halamphora coffeiformis*, *Navicula capitatoradiata*, *N. cincta*, *N. erifuga*, *N. hanseatica*, *Nitzschia tubicola*, *Tryblionella apiculata*, *T. hungarica* were assigned to marine or brackish water ecosystems and also in freshwaters with high electrolyte content (Guiry & Guiry, 2019). Furthermore, Akbulut (2010) reported *N. cincta* and *T. apiculata* in the Tuz Lake basin which is under the brackish, saline category. Species with high tolerance to salinity living in brackish waters or freshwaters with high conductivity could be found in different habitats, like some freshwater taxa observed in the marine coasts. Another taxon which generally referred to as marine species is *Berkeleya* sp. (Figure 2). This species resembles *Berkeleya fennica*, a brackish species previously reported from the Baltic Sea (Witkowski et al. 2000) and occurred mainly in brackish and a few freshwater habitats of Burdur river basin. Nevertheless, ultrastructure details are needed for the identification of the taxa.

In this research, a total of 223 taxa were found with a 11 new records for Turkey. These taxa were *Craticula buderi*, *Cymbella lange-bertalotii*, *Encyonema vulgare*, *Gomphonema auritum*, *G. drutelingense*, *G. exilissimum*, *Nitzschia bulnheimiana*, *Paraplaconeis minor*, *Placoneis anglophila*, *Placoneis clementoides* and *Stauroneis acidoclinata*. The results would contribute to the knowledge of diatom distribution in Turkey and Burdur River Basin, in particular. To determine the ecological quality of the river basins, taxonomical results would be a supplement for the physicochemical parameters for further studies. The results extend the biogeography of diatoms in Turkey and contribute to the knowledge of the diatom composition and distribution in the river basin.

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References

Akbulut, A. (2010). The diatom composition of the Salt Lake Basin and its relationship with salinity. *Ekoloji*, *159*, 150–159. doi: 10.5053/ekoloji.2010.7418

Atıcı, T. (1997). Sakarya River pollution and algae. *Ekoloji ve Çevre Dergisi*, *6(24)*, 28–32 (in Turkish).

Ács, É., Szabó, K., Tóth, B. & Kiss, K.T. (2004). Investigation of benthic algal communities, especially diatoms of some Hungarian streams in connection with reference conditions of the Water Framework Directives. *Acta Botanica Hungarica*, *46*, 225–278.

Barlas, M., Mumcu, M.F., Solak C.N. & Çoban, O. (2002). Akçapınar Deresi ve Kadın Azmağı Deresi (Muğla) epilitik alg florasının su kalitesine bağlı olarak incelenmesi. VI. Ulusal Biyoloji Kongresi. 4–7 Eylül 2002, Malatya, pp. 65–68 (in Turkish).

Baytut, Ö., & Gönülol, A. (2016). Phytoplankton distribution and variation along a freshwater-marine transition zone (Kızılırmak River) in the Black Sea. *Oceanological and Hydrobiological Studies*, *45*(4), 453–465.

Costa-Böddeker, S., Schwarz, A., Huy, H. Đ., & Schwalb, A. (2017). Diatom assemblages in surface sediments along nutrient and salinity gradients of Thi Vai estuary and Can Gio mangrove forest, Southern Vietnam. *Estuaries and coasts*, *40*(2), 479-492.

Çelekli, A., Toudjani, A.A., Lekesiz, H. & Çetin, T. (2018). Ecological quality assessment of running waters in the North Aegean Catchment with diatom metrics and multivariate approach. *Limnologica*, *73*, 20–27. https://doi.org/10.1016/j.limno.2018.09.001

Çelekli, A., Kayhan, S., Lekesiz, Ö., Toudjani, A.A. & Çetin, T. (2019). Limno-ecological assessment of Aras River surface waters in Turkey: application of diatom indices. *Enviromental Science and Pollution Research*, *26* (8), 8028–8038. https://doi.org/10.1007/s11356-019-04295-y

Çiçek, N. L., & Yamuç, F. (2017). Using epilithic algae assemblages to assess water quality in Lake Kovada and Kovada Channel (Turkey), and in relation to environmental factors. *Turkish Journal of Fisheries and Aquatic Sciences*, *17*(4), 701-711.

Dalkıran, N., Zünbülgil, B., Karacaoğlu, D., & Şükran, Dere (2016). Uluabat Gölü Epifitik Diyatomelerinin Uzun Dönemdeki Değişimi. *Journal of Limnology and Freshwater Fisheries Research*, *2*(3), 153-163 (in Turkish).

Demir, N., Çetin, T., Gök, C., & Şanal, M. (2017). First biological monitoring in the Akarçay Basin according to the water framework directive: Phytoplankton and phytobenthos. *Turkish Journal of Water Science & Management*, *1*(1), 90–105.

Guiry, M. D., & Guiry, G. M. (2019). *AlgaeBase*. World-wide electronic publication, National University of Ireland, Galway. Accessed at http://www.algaebase.org

Hofmann G., Werum, M., & Lange-Bertalot, H. (2011). Diatomeen im Süßwasser – Benthos von Mitteleuropa. Bestimmungsflora Kieselalgen für die ökologische Praxis. Über 700 der häufigsten Arten und ihre Ökologie. A.R.G. Gantner Verlag, Rugell. pp.908.

Karacaoğlu, D., & Dalkıran, N. (2017). Epilithic diatom assemblages and their relationships with environmental Epilithic diatom assemblages and their relationships with environmental variables in the Nilüfer Stream Basin, Bursa, Turkey. *Environmental Monitoring Assessment*, *189*, 1–22.

Kociolek, J.P., Balasubramanian, K., Blanco, S., Coste, M., Ector, L., Liu, Y., … Witkowski, J. (2019). *DiatomBase*. Retrieved August 31, 2017, from http://www.diatombase.org

Krammer, K. & Lange-Bertalot, H. (1986). Bacillariophyceae. 1. Teil: Naviculaceae. Ettl, H., Gerloff, J., Heynig, H. and Mollenhauer, D (Ed.). *Süsswasser flora von Mitteleuropa, Band 2/1.* Stuttgart: Gustav Fischer Verlag.

Krammer, K., & Lange-Bertalot, H. (1988). Bacillariophyceae. 2. Teil: Bacillariaceae, Epithemiaceae, Surirellaceae. Ettl, H., Gerloff, J., Heynig, H. and Mollenhauer, D. (Ed.). *Süsswasserflora von Mitteleuropa, Band 2/2.* Stuttgart: Gustav Fischer Verlag.

Krammer, K., & Lange-Bertalot, H. (1991a). Bacillariophyceae. 3. Teil: Centrales, Fragilariaceae, Eunotiaceae. Ettl, H., Gerloff, J., Heynig, H. and Mollenhauer, D. (Ed.). *Süsswasserflora von Mitteleuropa, Band 2/3*. Stuttgart: Gustav Fischer Verlag.

Krammer, K., & Lange-Bertalot, H. (1991b). Bacillariophyceae. 4. Teil: Achnanthaceae, Kritische Ergänzungen zu *Navicula* (Lineolatae) und Gomphonema, Gesamtliteraturverze ichnis Teil 1-4. Ettl, H., Gärtner, G., Gerloff, J., Heynig, H. And Mollenhauer, D. (Ed.). S*üsswasserflora von Mitteleuropa, Band 2/4*. Stuttgart: Gustav Fischer Verlag.

Kulikovskiy, M., Glushchenko, A.M., Kuznetsova, I.V., & Genkal, S.I. (2016). *Identification book of diatoms from Russia*. pp. 1-804, 165 pls. Yaroslavl: Filigran. [in Russian].

Özer, T., Erkaya, I.A., Solak, C.N., & Udoh, A.U. (2018). Diversity and ecology of algae from Melen River (Western Black Sea River Catchment) in Turkey. *Turkish Journal of Fisheries and Aquatic Sciences*. *18*, 1187–1194.

Rimet, F. (2012). Recent views on river pollution and diatoms. *Hydrobiologia*, *638*, 1–24.

Rovira, L., Trobajo, R., & Ibanez, C. (2009). Periphytic diatom community in a Mediterranean salt wedge estuary: the Ebro Estuary (NE Iberian Peninsula). *Acta Botanica Croatica*, *68*(2), 285-300.

Schröder, M., Sondermann, M., Sures, B., & Hering, D. (2015). Effects of salinity gradients on benthic invertebrate and diatom communities in a German lowland river. *Ecological indicators*, *57*, 236-248.

Solak, C.N., Çetin, T., Kaleli, A. (2018). Distribution of benthic diatom (phytobenthos) composition in Küçük Menderes River Basin. *Turkish Journal of Water Science & Management*, *2*, 2–35.

Solak, C.N., Çetin, T., Karaaslan, Y., Kaleli, A., Yılmaz, E., Duran, M., Kıvanç, G., Kirmençe, T., Koyunluoğlu, Ş.A., Çankaya, F., Yılmaz, D. (2019). Common Diatoms of Phytobenthos in Gediz River Basin. *Turkish Journal of Water Science &Management, 3*(2), 58-70.

Şanal, M., & Demir, N. (2018). Use of the epiphytic diatoms to estimate the ecological status of Lake Mogan. *Applied Ecology and Environmental Research*. *16*(3), 3529–3543.

Turkish Official Gazette. (2019). *Biyolojik İzleme Tebliği* (Publication No: 30808) Retrieved from https://www.resmigazete.gov.tr/eskiler/2019/06/20190621.pdf

Water Framework Directive. (2000). *Common implementation strategy for the water framework directive* (2000/60/EC).

Witkowski, A., Lange-Bertalot, H., & Metzeltin, D. (2000) *Diatom Flora of Marine Coasts I*. In: Iconographia Diatomologica 7. Konigstein: Koeltz Scientific Books.

Extendend Turkish Abstract
(Genişletilmiş Türkçe Özet)

Burdur Nehir Havzasındaki Farklı Habitatlarda Bentik Diyatome Kompozisyonu ve Dağılımı

Diyatomeler, sucul sistemlerdeki biyolojik izleme çalışmalarında önemli bir bileşendir. Avrupa Birliği Su Çerçeve Direktifi'nin (2000/60/AT) (SÇD) yürürlüğe girmesini izleyen yıllarda, uyum süreçleri kapsamında Türkiye'de de havza bazında izleme çalışmaları başlamıştır. Bu çerçevede, Türkiye'deki 25 havza içerisinde en küçük alana sahip iki havzadan biri olan Burdur Havzası'nın bentik diyatome kompozisyonu ilk defa bu çalışma ile detaylı olarak incelenmiştir.

Fitobentoz (diyatome) örneklemesi Nisan ve Ekim 2018'de iki dönemde gerçekleştirilmiş, fizikokimyasal parametreler ise Ocak-Aralık 2018 döneminde aylık olarak izlenmiştir. Havzada örnekleme noktası olarak belirlenen toplam 30 istasyonun 7'sinin her iki örnekleme döneminde kuru olduğunun tespit edilmesi nedeniyle 13'ü akarsu, 4'ü doğal göl ve 6'sı rezervuar olmak üzere toplamda 23 istasyonda örnekleme çalışması gerçekleştirilmiştir. 21.06.2019 tarihli Resmi Gazete'de yayınlanan Biyolojik İzleme Tebliği uyarınca fitobentoz örneklemesinde ağırlıklı olarak nehirlerde epilitik alg örneklemesi, göl ve rezervuarlarda ise epifitik alg örneklemesi yapılması gerekmektedir. Ancak, özellikle su seviyesindeki değişimler ve suyun fizikokimyasal özelliklerine bağlı olarak makrofit tespit edilemeyen göl ve rezervuarlarda ikinci sırada önerilen epilitik algler örneklenmiştir. Alınan örnekler %10 HC1 ile muamele edilmiş sonrasında H_2O_2 ile yakılarak organik maddelerin uzaklaştırılması sağlanmıştır. Diyatome örnekleri Naphrax® kullanılarak sabit preparatlar haline getirilmiştir. Sayım ve teşhisler için Zeiss Axio Observer Z1 (Carl Zeiss mikroskobu Gmbh, Jena, Almanya) mikroskobu kullanılmıştır. Fitobentoz türlerinin teşhisinde Krammer ve Lange-Bertalot (1986; 1988; 1991 a, b), Hofmann ve ark., (2011), Kulikovskiy ve ark., (2016), Guiry ve Guiry (2019) ve Kociolek ve ark., (2019) kaynaklarından yararlanılmıştır. Her bir örnekten en az 300 diyatome frustülü sayılarak, türlerin nispi bolluğu, sayılan toplam frustüllerin yüzdesi olarak ifade edilmiştir (% cinsinden nispi bolluk).

Fizikokimyasal ölçümler, havzada yer alan göl, rezervuar ve nehirlerin genel olarak alkali yapıda olduğunu ortaya koymuştur. Ortalama elektriksel iletkenlik (Eİ) genel olarak rezervuar (0,5 mS cm^{-1}) ve nehirlerde (1,7 mS cm^{-1}) düşük tespit edilirken göllerde (22.3 mS cm^{-1}) nispeten daha yüksek ölçülmüştür. Özellikle doğal göller içerisinde yer alan Acıgöl (57.2 mS cm^{-1}) ve Burdur (29.4 mS cm^{-1}) göllerinin Eİ değerlerinin yıl boyunca diğer örnekleme noktalarına göre oldukça yüksek olduğu görülmüştür. Yıl boyunca tüm istasyonlarda çözünmüş oksijen (ÇO) değerlerinde önemli değişiklikler gözlemlenmiştir. Burdur ve Acıgöl'de zaman zaman çok düşük ÇO değerleri gözlense de (sırasıyla, 2,7 mg L^{-1}; 1mg L^{-1}) ortalama değerler Burdur Gölü için 6,05 mg L^{-1}, Acıgöl için ise 7,75 mg L^{-1} olarak tespit edilmiştir. Havzada çalışılan 13 akarsu, 4 göl ve 6 baraj gölünde toplam 223 takson gözlenmiştir. İlkbaharda 136 tür tespit edilirken, sonbaharda tür çeşitliliği 174 tür olarak tespit edilmiştir. Cins seviyesinde en çok tür çeşitliliği 27 adet tür ile *Navicula* ve *Nitzschia* cinslerinde olup bunu 22 adet tür ile *Gomphonema* cinsi izlemiştir. *Navicula antonii* ve *Nitzschia palea* var. *debilis* türleri 17,1 % ve 15,5 % ile en yaygın türler olarak tespit edilmiştir. *Craticula buderi, Cymbella lange-bertalotti, Encyonema vulgare, Gomphoenma auritum, G. drutelingense, G. exilissimum, Nitzschia bulnheimiana, Paraplaconeis minor, Placoneis anglophila, Placoneis clementoides ve Stauroneis acidoclinata* olmak üzere toplamda 11 tür Türkiye sularında ilk kez gözlenmiştir. Rezervuar, göl ve akarsularda tespit edilen tür sayıları farklılık göstermiş ve en yüksek tür çeşitliliğine akarsularda rastlanılmıştır. Çeşitliliğin akarsularda 160 tür ve rezervuarlarda 131 tür arasında değiştiği görülmüş,

bununla birlikte, göllerde biyolojik çeşitlilik daha düşük bulunmuştur (61 tür). Rezervuar, göl ve akarsu istasyonlarında tespit edilen türlerin kendi içlerinde nispi bollukları hesaplanmış ve toplam nispi bolluğun %5 ve üzerini oluşturan türler ayrıca değerlendirilmiştir. Buna göre, rezervuar örneklerinde *Pantocsekiella iranica* (%14) ve *P. ocellata* (%17) ve *Ulnaria delicatissima* (%17) en bol bulunan türler olurken, göl örneklerinde bu türlerin yerini *Achnanthidium minutissimum var. jackii* (sonbaharda %20) ve *Encyonema caespitosum* (ilkbaharda %35) almıştır. Akarsu örneklerinde ise ilkbaharda *Tabularia fasciculata* (%15) en bol bulunan tür olurken sonbaharda *Nitzschia palea var. debilis* (%38) ve *Navicula antonii* (%38) dışında toplam nispi bolluğun %5'inin üzerine çıkan türün olmadığı görülmüştür. *Navicula* ve *Nitzschia* cinslerine ait türler tatlı su habitatlarında yüksek çeşitliliğe sahip ve yaygın olarak bulunan cinslerdir. Burdur Nehir Havza'sında da bu cinslere ait tür çeşitliliği yüksek bulunsa da, *N antonii* ve *N. palea* var. *debilis* türleri dışında toplam nispi bollukları düşük bulunmuştur.

Burdur Nehir Havzası, denizle bağlantısı olmayan kapalı bir havzadır ve Burdur ve Acıgöl gibi yüksek tuzluluk karakterindeki göllere sahiptir. Bu çalışma sonucunda, havzada acısu ve deniz kıyı bölgelerinde yayılış gösterdiği bilinen bazı türler tespit edilmiştir. Özellikle Acıgöl'de acısularda bulundukları bilinen *Halamphora coffeiformis* ve *Navicula cincta* türleri oldukça yüksek sayılara ulaşmıştır. Benzer şekilde, hem deniz hem de acı sularda bulunduğu bilinen *Berkeleya* sp.*, Halamphora coffeiformis, Navicula simulata, Tryblionella apiculata* gibi türlerin Burdur Gölü'nde önemli miktarda varlık gösterdiği tespit edilmiştir. Havzada tespit edilen, *Navicula capitatoradiata, N. erifuga, N. hanseatica, Nitzschia tubicola, Tryblionella apiculata, Tabularia fasciculata* gibi türlerin ekolojik tercihleri açısından deniz ve acısularda bulunduklarına dair kayıtlar mevcuttur (Guiry & Guiry, 2019). *Navicula antonii* türü ise havzada farklı tuzluluk seviyelerine sahip tüm alanlarda bulunmuştur. Bu durum türün örihalin bir tür olduğunu göstermektedir.

Burdur Nehir Havzası'nda yapılan bu çalışma bölgedeki ilk detaylı çalışmadır ve Türkiye diyatome florasına 11 yeni kayıt türün ilave edilmesini sağlamıştır. Burdur Havzası'nda gözlenenen yüksek biyoçeşitlilik havza içi sucul ekosistemlerin çeşitliliğinin bir sonucudur. Ayrıca, bu çalışmada kapalı bir havza olan Burdur Havzası'nın diğer havzalardan daha farklı diyatome kompozisyonuna sahip olduğu gözlenmiştir. Elde edilen veriler, Burdur Havzası benzeri özel alanlarda yapılacak izleme çalışmalarında önemli olacaktır.

Technical Note

Parameter Selection for Chemical Monitoring of Sediment in Lake Beysehir

Beyşehir Gölü Sedimanında Kimyasal İzleme Parametrelerinin Seçimi

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Abstract

Sediment plays a crucial role in water quality management. It acts as a potential sink for hydrophobic substances that cannot dissolve in water. With the emission of these substances to water body from the sediment, water quality can deteriorate. Since by monitoring only in the water column, it is impossible to detect these hydrophobic substances, chemical monitoring of sediment should also be conducted for water quality management Moreover, sediment monitoring provides information about historic contamination by vertical samplin of sediment. Thus, for the water bodies that have no historical water quality data, information about historical contamination can be gained. Also, the Water Framework Directive (WFD) states that environmental quality standards should be set for sediment and with the perspective of the WFD, basin based sediment management should be started. This study aimed to select the parameters that should be monitored in the sediment in Lake Beysehir. For this purpose, specific pollutants that can originate from industries around Beysehir were selected. Also, pesticides specific to the plants cultivated in Konya Basin were identified. Between these pollutants the ones that have the tendency to accumulate in the sediment were identified. The priority substances were also studied and the list of chemicals that should be monitored in the sediment was determined.

Keywords: *Sediment, chemical monitoring, Octanol-Water Partition Coefficient, Water Framework Directive*

Öz

Sediment su kalitesi yönetimi konusunda önemli bir role sahiptir. Sediment, suda çözünemeyen hidrofobik maddelerin çökelebileceği zemin görevi görmektedir. Sedimentte biriken maddelerin zaman içerisinde sedimentten su kütlesine geçişi ile birlikte su kalitesi bozulmaya uğrayabilir. Sadece su kolonunda yapılan izleme çalışmaları ile bu maddelerin tespit edilmesi mümkün olmadığından, su kalitesi yönetimi için sedimentte kimyasal maddelerin izleme çalışmaları yapılmalıdır. Ayrıca, sedimentte yapılan dikey örnekleme ile geçmişteki kirlenme hakkında bilgi sahibi olunabilmektedir. Böylelikle, geçmişe dönük su kalitesi verisi olmayan su kütlelerinde geçmişteki kirlenme hakkında bilgi edinilebilir. Ayrıca, Su Çerçeve Direktifi (SÇD), sediment için çevresel kalite standartlarının belirlenmesi gerektiğini ve havza bazlı sediment yönetiminin başlatılması gerektiğini belirtmektedir. Bu çalışma Beyşehir Gölünde sedimentte izlenmesi gereken kimyasal maddelerin seçilmesini hedeflemiştir. Bu amaç doğrultusunda, Beysehir Gölü etrafında yer alan endüstriyel tesislerden kaynaklanabilecek belirli kirleticiler ve Konya havzasında yetiştirilen bitkilere özgü pestisitler seçilmiştir. Bu kirleticiler arasından sedimentte birikme potansiyeli olanlar belirlenmiştir. Ayrıca öncelikli maddeler de çalışılarak, sedimentte izlenmesi gereken kimyasalların listesi oluşturulmuştur.

Anahtar kelimeler: *Sediman, kimyasal izleme, Oktanol-Su Ayrışım Katsayısı, Su Çerçeve Direktifi*

Introduction

In recent years, as a result of increase in population and in the number of industrial facilities, the amount of pollutants that are released to the environment is dramatically increased. Direct or indirect release of these substances into the environment causes the balance of nature to be disrupted. Additional to this disruption, reaching of these pollutants into the aquatic environment has raised the issue of water pollution. To have a solid grasp of the situation, monitoring, just only in water matrix is not sufficient for the integrated and comprehensive water quality management.

The pollutants released into the river bed have the potential of accumulation in the sediment. Especially, the hydrophobic pollutants that cannot dissolve in water precipitate in the sediment and, therefore, these hydrophobic pollutants cannot be detected by monitoring only in water matrix. In addition, no matter how much water quality is improved, the pollutants in the sediment have the potential to transport again into water matrix over time by causing water quality to deteriorate again. Moreover, for comprehensive and efficient water quality management, both present situation and historic contamination should be considered for establishing better programs of measures (Chapman, 1996). By performing vertical sampling in the sediment, it is possible to gain information about historical contamination. Thus, when dealing with water quality issues, it is crucial to evaluate the water column and the sediment in an integrated manner.

Despite being an integrated part of rivers and lakes and its importance in water quality, sediment does not significantly take place in the European Union Water Framework Directive. The WFD refers to the sediment mostly in the definitions and these definitions are mainly about water quality issues. The attributes to sediment in the WFD are as follows;

- In Article 2, Definitions, the definition of environmental quality standard (EQS) is; "the concentration of a particular pollutant or group of pollutants in water, sediment or biota which should not be exceeded in order to protect human health and the environment."

- In Article 16, Strategies against pollution of water; "The Commission shall submit proposals for quality standards applicable to the concentrations of the priority substances in surface water, sediments or biota."

- In Annex V, it is stated that "In deriving environmental quality standards for pollutants listed in points 1 to 9 of Annex VIII of WFD for the protection of aquatic biota, Member States shall act in accordance with the following provisions. Standards may be set for water, sediment or biota," (EC, 2000)

Apparently, the WFD mostly addresses to the sediment in EQS and it clearly states that EQS should be set for water, sediment or biota. Nevertheless, after the WFD has entered into force, a new perspective arose for the sediment. Instead of local sediment monitoring activities, basin based sediment management has started. Also, the European Sediment Network (SedNet) emphasizes that sediment should also take place in river basin management plans for sustainable sediment management. Therefore, it is crucial to integrate sediment monitoring activities in river basin monitoring programs (Brills, 2008).

Sediment monitoring frequency is not clearly specified in the WFD itself. However, according to 2013/39/EU Directive, it is said that for compliance with EQS, the monitoring should be conducted at least once a year (EU, 2013). This frequency for sediment can be changed due to the sedimentation rate and the hydrological regime in the water body. Also, for dynamic water bodies, the frequency of monitoring can be more than once a year (EC, 2010).

The time for the sediment monitoring should be set at the season when the sedimentation rate is maximum. Since the sedimentation rate is maximum when the flow rate of the water is slow, the sampling should be conducted in the summer season.

In Turkey, the studies on water quality monitoring are mainly carried out by some public institutions and organizations. However, except some research studies, chemical monitoring of sediment does not take place in water quality monitoring activities. Moreover, in most of these studies, the parameters monitored in the sediment are only heavy metals. However, monitoring of heavy metals in the sediment is not sufficient since hydrophobic substances can also precipitate and sink in sediment and those should be taken into account as well. These substances can be in lower concentrations in water matrix whereas they can be in higher concentration in sediment matrix, therefore; parameters that to be monitored in the sediment should be selected carefully considering their hydrophobicity.

The most fundamental criteria for the selection of chemical substances to be monitored in the sediment is the solubility of a substance in the water. As the substance become more hydrophobic, its tendency to accumulate in the sediment increases (Brils, 2008).

The hydrophobicity of a chemical substance can be determined by using Octanol-Water Partition Coefficient (K_{ow}). K_{ow} is defined as the parameter that states the solution ratio of a chemical substance in an organic or inorganic phase. For the selection of chemical substances that will be monitored in the sediment, the octanol-water partition coefficient is used. In principle, chemical substances that has log K_{ow} higher than 5, have a higher tendency to accumulate in the sediment. Therefore, these substances should be monitored in the sediment (AMPS, 2004). On the other hand, for chemical substances that has log K_{ow} higher than 3 and below 5, monitoring matrix is optional and it can be chosen as sediment or suspended material. The matrix selection for these chemical substances is due to the contamination of particular matrix (EC, 2010).

$$Kow = \frac{Coktanol}{Cwater} \quad \text{[EC, 2010]}$$

For the determination of chemical parameters to be monitored in Lake Beysehir, octanol-water partition coefficient is used and the substances that have the values of log $K_{ow}>5$ are chosen. Since chemical parameters that have log K_{ow} higher than 3 and below 5 is optional for sediment matrix and when the difficulties in analyses of these parameters for sediment is considered, for Beysehir Lake, chemical parameters that have log K_{ow} value higher than 5 is chosen.

For the metals, K_{ow} cannot be used as a criteria for the selection of those to be monitored in the sediment. When the behaviour of metals in the water is investigated, it is found that heavy metals are highly hydrophobic and they cannot be acquired in the water matrix in most of the water quality monitoring activities. Therefore, since heavy metals are very toxic and they have the high tendency to be adsorbed in the sediment, they should definitely be selected to be monitored in the sediment.

Another important issue that should be considered for the selection of metals to be monitored in the sediment is the metal concentration in the natural structure of sediment, in other words, the background concentration of the metals in the sediment. If natural background concentration of a parameter is high, it is impossible to distinguish whether the concentration of the parameter is due to the nature of the sediment or due to the contamination of the sediment.

Similar to the selection of water quality monitoring points in the water bodies, selected monitoring points should be representative of a water body for sediment monitoring. Also, the point sources are very important for the determination of location in the sediment monitoring. Generally, monitoring points should be selected downstream of an industry to analyze the contamination (EC, 2010). Especially, if it is known that there was a point source in a certain area in the past, this area should be considered as a location for sediment monitoring to detect the historic contamination.

Method

Study Area

The Konya Basin is the third largest basin of Turkey, located in the central Anatolia of Turkey. The main pressures on the basin for water quality are agriculture and livestock, and the industries in the basin are mainly metal and automotive based. Other industries are food, chemistry, textile, paper and extractive industry. The basin has a water shortage problem due to low precipitation and overexploitation of water resources.

In this study, Lake Beysehir located in Konya Basin of Turkey was selected as a pilot area by considering the industries and the agricultural activities around the lake. For this study, the pressures on the lake were investigated and it was found that there is an organized industrial zone next to the lake. The main industries in the industrial zone are machine manufacturing, automotive, weapon manufacture and chrome plating that can be seen in Figure 1. Also, there is a sugar refinery close to the lake. This factory is also considered as a pressure to the lake since its discharges can reach to the lake via the tributaries (MoFWA,2013). When the pressures on the lake and the parameters that are discharged from the industries are considered, it is possible to have contaminated sediment at the bottom of the lake. Therefore, no matter how much precaution is taken for the lake in these circumstances, sufficient water quality is not likely to be provided unless the pollution in the bottom sediment is prevented. Therefore, it is important to plan monitoring activities for the sediment in Lake Beysehir.

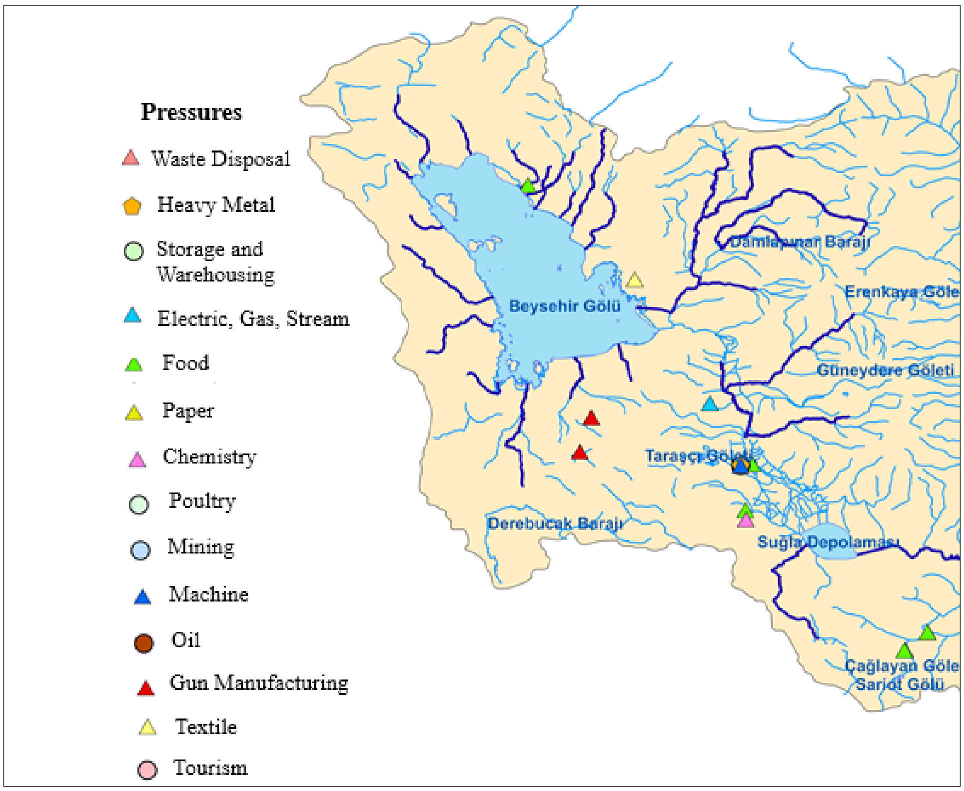

Figure 1. Beysehir Lake and the pressures around it.

Selection of Monitoring Parameters

Parameters to be monitored in the sediment for this lake were selected by using octanol-water partition coefficient for determination of hydrophobic properties of pollutants and by considering the previous monitoring activities conducted in water matrix of the lake.

In 2014, the chemical monitoring of the sediment was carried out in the Konya Basin as a part of the project called "Basin Monitoring and Determination of Reference Points" and the sampling was conducted for four times within a year in Lake Beysehir. However, in this study, the parameters monitored were only heavy metals and the monitoring results are given in Table 1.

Table 1

Results of Heavy Metals Concentration in The Sediment Of Lake Beysehir (Mofwa, 2014)

Sampling Date		28.01.2014	26.05.2014	05.07.2014	22.09.2014	Average
Parameter	Unit					
Aluminium (Al)	mg/kg	2755	4025	11598	2975	5338.2
Copper (Cu)	mg/kg	10.75	8.5	22.3	27.5	17.26
Zinc (Zn)	mg/kg	17.5	18.25	54	47.5	34.31
Chromium (Cr)	mg/kg	37.75	8.5	42	13	25.31
Cadmium (Cd)	mg/kg	<0.25	<0.25	<0.25	<0.25	<0.25
Mercury (Hg)	mg/kg	<0.25	<0.25	<0.25	<0.25	<0.25
Lead (Pb)	mg/kg	7.25	5.75	13.3	9	8.82
Arsenic (As)	mg/kg	3.25	1.3	4.25	3.5	3.07

As can be seen in Table 1, the aluminium concentration is very high compared to the other metals. However, this high concentration can be due to background concentration of sediment. Therefore, in this case, aluminium should not be selected as a monitoring par

In this project, together with sediment sampling, the monitoring in the water matrix in Lake Beysehir has been conducted and the monitoring results for heavy metals and priority substances are presented in Table 2 and Table 3.

When the monitoring results of heavy metals in the sediment and in the water column are compared, heavy metal concentration is lower in the water column while the heavy metal concentration in the sediment is considerably higher. For instance, average concentration of zinc in water column is 5.46 µg/L in water column, whereas its concentration in the sediment is 34.31 mg/kg. When the water quality is investigated, this amount of zinc corresponds to Class I, which is defined as very good quality in terms of physicochemical parameters, however, when the concentration in sediment is considered, this concentration is quite high. Therefore, it can be concluded that it is not possible to determine the heavy metal contamination just by monitoring water matrix. On the other hand, when the results of priority substances presented in Table 3 are investigated, it is obvious that most of the results are found below the environmental quality standards, in other words "passed". However, since there is hydrophobic chemicals among priority substances, their concentrations are expected to be higher in the sediment.

Table 2

Monitoring Results of Heavy Metals in Water Matrix of Beysehir Lake (Mofwa, 2014)

Parameter	CAS No	LOQ (µg/L)	Unit		1st Period 28.01.2014 (µg/L)	2nd Period 26.05.2014 (µg/L)	3rd Period 05.07.2014 (µg/L)	4th Period 22.09.2014 (µg/L)	Average concentration (µg/L)	Class I Boundry	Class II Boundry	Class III Boundry	Class IV Boundry	Quality Class of the sample
Mercury (Hg)	743 9- 97-6	0.002	µg/L	Surface	0.155	<0.002	<0.002	0.161	0.043	<0.1	0.1-0.5	0.5-2.0	>2	Class I
				Middle	0.081	<0.002	0.00327	0.05						
				Bottom	0.007	<0.002	0.00334	<0.002						
Cadmium (Cd)	744 0- 43-9	0.002- 0.04	µg/L	Surface	<0.04	<0.04	<0.04	<0.04	0.02	≤ 2	2.5	5-7	>7	Class I
				Middle	<0.04	<0.04	<0.04	<0.04						
				Bottom	<0.04	<0.04	<0.04	<0.04						
Lead (Pb)	743 9- 92-1	0.01- 0.1	µg/L	Surface	0.719	<0.1	0.171	11.1	1.217	≤ 10	10-20	20-50	>50	Class I
				Middle	0.359	<0.1	0.192	1.27						
				Bottom	<0.1	<0.1	0.456	0.147						
Cupper (Cu)	744 0- 50-8	0.01- 0.1	µg/L	Surface	1.38	0.489	0.611	7.27	1.624	≤ 20	20-50	50-200	>200	Class I
				Middle	0.92	0.4115	2.47	1.5						
				Bottom	0.457	0.334	3.14	0.501						
Nickel (Ni)	744 0- 02-0	0.05- 0.3	µg/L	Surface	3.13	1.52	1.42	7.89	2.94	≤ 20	20-50	50-200	>200	Class I
				Middle	2.07	1.1295	1.1	12.2						
				Bottom	1.01	0.739	1.78	1.29						
Zinc (Zn)	744 0- 66-6	0.2-2	µg/L	Surface	13.9	<2	<2	22.1	5.46	≤ 200	200-500	500- 2000	>2000	Class I
				Middle	6.95	<2	<2	13.2						
				Bottom	<2.0	<2	2.41	<2						

Table 3

Monitoring Results of Priority Substances in Beysehir Lake in Water Matrix (MoFWA, 2014)

Parameter	CAS No	1st Period 28.01.2014	2nd Period 26.05.2014	3rd Period 05.07.2014	4th Period 22.09.2014	Average concentration (µg/L)	AA-EQS (µg/L)	Evaluation of the sample (Passed or Failed)
Fluoranthene	206-44-0	<0.01	<0.01	<0.01	<0.01	0.005	0.0063	Passed
Alachlor	15972-60-8	<0.02	<0.02	<0.02	<0.02	<0.02	0.3	Passed
Anthracene	120-12-7	<0.01	<0.01	<0.01	<0.01	<0.01	0.1	Passed
Atrazine	1912-24-9	<0.02	<0.02	<0.02	<0.02	<0.02	0.6	Passed
Benzene	71-43-2	<0.1	0.11	<0.1	<0.1	0.065	10	Passed
Brominated diphenylethers	not applicable	<0.031	<0.031	<0.031	<0.031	<0.031	-	-
Cadmium and its compounds	7440-43-9	<0.04	<0.04	<0.04	<0.04	<0.04	-	-
Carbon-tetrachloride	56-23-5	<0.1	<0.1	<0.1	<0.1	<0.1	12	Passed
Chloroalkanes, C10-13	85535-84-8	<0.2	<0.2	<0.2	<0.2	<0.2	0.4	Passed
Chlorfenvinphos	470-90-6	<0.02	<0.02	<0.02	<0.02	<0.02	0.1	Passed
Chlorpyrifos (Chlorpyrifos-ethyl)	2921-88-2	<0.012	<0.012	<0.012	<0.012	<0.012	0.03	Passed
Aldrin	309-00-2	<0.02	<0.02	<0.02	<0.02	<0.02	Σ=0.01	-
Dieldrin	60-57-1	<0.02	<0.02	<0.02	<0.02	<0.02		-
Endrin	72-20-8	<0.02	<0.02	<0.02	<0.02	<0.02		-
Isodrin	465-73-6	<0.02	<0.02	<0.02	<0.02	<0.02		-
Total DDT	not applicable	<0.02	<0.02	<0.02	<0.02	<0.02	0.025	Passed
4,4' - DDT (p,p'-DDT)	50-29-3	<0.02	<0.02	<0.02	<0.02	0.01	0.01	Passed
1,2-dichloroethane	107-06-2	<0.1	<0.1	<0.1	<0.1	<0.1	10	Passed
Dichloromethane	75-09-2	<0.1	1.925	0.31	<0.1	0.58	20	Passed
Di(2-ethylhexyl)phthalate (DEHP)	117-81-7	0.17	<0.1	<0.1	0.08	0.087	1.3	Passed
Diuron	330-54-1	<0.04	<0.04	<0.04	<0.04	<0.04	0.2	Passed
Endosulfan	115-29-7	<0.02	<0.02	<0.02	<0.02	<0.02	0.005	-
Hexachlorobenzene	118-74-1	<0.02	<0.02	<0.02	<0.02	<0.02	-	-
Hexachlorobutadiene	87-68-3	<0.1	<0.1	<0.1	<0.1	<0.1	-	-
Hexachlorocyclohexane	608-73-1	<0.02	<0.02	<0.02	<0.02	<0.02	0.02	Passed

Parameter	CAS No	1st Period 28.01.2014	2nd Period 26.05.2014	3rd Period 05.07.2014	4th Period 22.09.2014	Average concentration (µg/L)	AA-EQS (µg/L)	Evaluation of the sample (Passed or Failed)
Isoproturon	34123-59-6	<0.04	<0.04	<0.04	<0.04	<0.04	0.3	Passed
Lead and its compounds	7439-92-1	0.376	<0.1	0.273	4.17	1.22	1.2	Failed
Mercury and its compounds	7439-97-6	0.081	<0.002	0.0025	0.0388	0.031	-	-
Naphthalene	91-20-3	<0.1	<0.1	0.071	<0.1	0.055	2	Passed
Nickel and its compounds	7440-02-0	2.07	1.13	1.43	7.12	2.94	4	Passed
Nonylphenols	not applicable	0.576	<0.05	0.487	0.485	0.393	0.3	Failed
Octylphenols	not applicable	<0.05	<0.05	<0.05	<0.05	<0.05	0.1	Passed
Pentachlorobenzene	608-93-5	<0.02	<0.02	<0.02	<0.02	<0.02	0.007	-
Pentachlorophenol	87-86-5	<0.1	<0.1	<0.1	<0.1	<0.1	0.4	Passed
Benzo(a)pyrene	50-32-8	<0.01	<0.01	<0.01	<0.01	<0.01	0.00017	-
Benzo(b)fluoranthene	205-99-2	<0.01	<0.01	<0.01	<0.01	<0.01	-	-
Benzo(k)fluoranthene	207-08-9	<0.01	<0.01	<0.01	<0.01	<0.01	-	-
Benzo(g,h,i)perylene	191-24-2	<0.01	<0.01	<0.01	<0.01	<0.01	-	-
Indeno(1,2,3-cd)pyrene	193-39-5	<0.01	<0.01	<0.01	<0.01	<0.01	-	-
Simazine	122-34-9	<0.02	<0.02	<0.02	<0.02	<0.02	1	Passed
Tetrachloroethylene	127-18-4	<0.1	<0.1	<0.1	<0.1	<0.1	10	Passed
Trichloroethylene	79-01-6	<0.1	<0.1	<0.1	<0.1	<0.1	10	Passed
Trichlorobenzenes	12002-48-1	<0.1	<0.1	<0.1	<0.1	<0.1	0.4	Passed
Trichloromethane	67-66-3	<0.1	7.86	1.35	<0.1	2.33	2.5	Passed
Trifluralin	1582-09-8	<0.02	<0.02	<0.02	<0.02	<0.02	0.03	Passed
Dicofol	115-32-2	<0.0033	<0.0033	<0.0033	<0.0033	<0.0033	0.0013	-
Perfluorooctane sulfonic acid and its derivatives (PFOS)	1763-23-1	<0.05	<0.05	<0.05	<0.05	<0.05	0.00065	-
Quinoxyfen	124495-18-7	<0.04	<0.04	<0.04	<0.04	<0.04	0.15	Passed
Dioxins and dioxin-like compounds (PCDD,PCDF)	not applicable	n.d.	n.d.	n.d.	n.d.	n.d.	-	-
Aclonifen	74070-46-5	<0.04	<0.04	<0.04	<0.04	<0.04	0.12	Passed
Bifenox	42576-02-3	<0.04	<0.04	<0.04	<0.04	<0.04	0.012	Passed

Parameter	CAS No	1st Period 28.01.2014	2nd Period 26.05.2014	3rd Period 05.07.2014	4th Period 22.09.2014	Average concentration (µg/L)	AA-EQS (µg/L)	Evaluation of the sample (Passed or Failed)
Cybutryne	28159-98-0	<0.04	<0.04	<0.04	<0.04	<0.04	0.0025	Passed
Cypermethrin	52315-07-8	<0.02	<0.02	<0.02	<0.02	<0.02	0.00008	Passed
Dichlorvos	62-73-7	<0.02	<0.02	<0.02	<0.02	<0.02	0.0006	Passed
Hexabromocyclododecane (HBCDD)		<0.05	<0.05	<0.05	<0.05	<0.05	0.0016	Passed
Heptachlor	76-44-8	<0.02	<0.02	<0.02	<0.02	<0.02	0.0000002	-
Terbutryn	886-50-0	<0.04	<0.04	<0.04	<0.04	<0.04	0.065	-

Note.AA=Anual Average, *n.d*=not detected.

Results

In Turkey, the specific pollutants were identified by conducting projects on hazardous chemical substances named Project on Control of Hazardous Substances Pollution, Project on the Determination of Hazardous Substances in Coastal and Transitional Waters of Turkey and Ecological Coastal Dynamics and Project on the Determination of the Water Pollution Caused by Use of Plant Protection Products and Identification of Environmental Quality Standards for Substance or Substance Groups. As a result of these projects, 117 specific pollutants and 133 pesticides are determined as specific pollutants in the surface waters in Turkey.

In this study, priority substances and specific pollutants including pesticides determined with these projects were studied and the parameters that should be monitored in the sediment matrix of Lake Beysehir were investigated. In Tables 4 and 5, the parameters that have log K_{ow} higher than 5 are shown. Since these parameters have the tendency to accumulate in the sediment, they should be monitored in sediment matrix together with water matrix.

Table 4

Priority Substances Which Have Log K_{ow} > 5

	Priority Substance	CAS No	Log K_{ow} (USEPA, 2018)
1	Cadmium and its compounds	7440-43-9	Heavy Metal*
2	Di(2-ethylhexyl)phthalate (DEHP)	117-81-7	7.60
3	Fluoranthene	206-44-0	5.16
4	Hexachlorobenzene	118-74-1	5.73
5	Lead and its compounds	7439-92-1	Heavy Metal*
6	Mercury and its compounds	7439-97-6	Heavy Metal*
7	Pentachlorobenzene	608-93-5	5.17
8	Polyaromatic hydrocarbons (PAH)	not applicable	6.44
9	Dicofol	115-32-2	5.02
10	Perfluorooctane sulfonic acid and its derivatives (PFOS)	1763-23-1	6.28
11	Dioxins and dioxin-like compounds	not applicable	6.80
12	Heptachlor and heptachlor epoxide	76-44-8/1024-57-3	6.10/ 4.98
13	Nickel	7440-02-0	Heavy Metal*
14	Trifluralin	1582-09-08	5.3
15	Brominated diphenyl ethers	not applicable	6.6
16	Chloroalkanes, C10-13	85535-84-8	4.4-8.7
17	Nonylphenols	not applicable	5.5
18	Octylphenols	not applicable	5.3

Note.*log Kow is used for organic compounds to determine hidrophobic character of compound. For heavy metals, log Kow is not used; however, heavy metals are hydrophobic and they have high tendecy to accumulate in sediment; therefore, they are selected without using log Kow value.

Table 4 and Table 5 shows that monitoring of only heavy metals in sediment is not sufficient to ensure adequate prevention of sediment contamination since there are other parameters that have the high potential to sink in sediment. According to hyrophobic charateristic properties of parameters, 18 priority substances and 16 specific pollutants should be monitored in sediment matrix.

Furthermore, when the area around Lake Beysehir is studied, agriculture activities are found to be the main pressures around the lake. Therefore, pesticides which indicate hydrophobic features, are specific to the basin, listed in Table 6, and should also be monitored.

Table 5

Specific Pollutants That are Possible to Discharge from The İndustries Near Lake Beysehir

	Parameter	CAS No	Log K_{ow} (USEPA, 2018)	Monitoring parameters chosen for Sediment Matrix
1	1,1-Dikloroetan	7534-3	1,79	
2	1,2,4-trimetilbenzen	95-63-6	3,63	
3	Aluminium	7429-90-5	Heavy metal*	x
4	Antimon	7440-36-0	Heavy metal*	x
5	Acenaphthene	83-32-9	3,92	
6	Cupper	7440-50-8	Heavy metal*	x
7	Barium	7440-39-3	Heavy metal*	x
8	Benzyl butyl phthalate (BBP)	85-68-7	4,73	
9	Benzo[a]fluorene	238-84-6	5,4	x
10	Berilyum	7440-41-7	Heavy metal*	x
11	Boron	7440-42-8	Heavy metal*	x
12	Zinc	7440-66-6	Heavy metal*	x
13	Iron	7439-89-6	Heavy metal*	x
14	dibutyltin oxide	818-08-6	5,33	x
15	Diphenyl ethere	101-84-8	4,21	
16	Free Cyanide	57-12-5	-0.182	
17	Phenanthrene	85-01-8	4,46	
18	Fluorene	86-73-7	4,18	
19	Silver	7440-22-4	Heavy metal*	x
20	Tin	7440-31-5	Heavy metal*	x

21	Cobalt	7440-48-4	Heavy metal*	x
22	Chrysene	218-01-9	5.46	x
23	Chromium	7440-47-3	Heavy metal*	x
24	n-Butyltin trichloride	1118-46-3	0,18	
25	Pyrene	129-00-0	4,88	
26	Polycyclic aromatic hydrocarbons	-		
27	Titanyum	7440-32-6	Heavy metal*	x
28	Trichloroethylene (TRI)	79-01-6	2,42	

Note. *log Kow is used for organic compounds to determine hidrophobic character of compound. For heavy metals, log Kow is not used; however, heavy metals are hydrophobic and they have high tendecy to accumulate in sediment; therefore, they are selected without using log Kow value.

Table 6

Hydrophobic Pesticides Specific to Konya Basin

	Pesticides	CAS No	Log Kow (at 20°C and pH=7), (IUPAC, 2018)
1	Bromopropylate	18181-80-1	5.4
2	Cyfluthrin	68359-37-5	6
3	Diafenthiuron	80060-09-9	5.76
4	Etoxazole	153233-91-1	5.52
5	Fenbutatin oxide	13356-08-6	5.15
6	Fenpropathrin	39515-41-8	6.04
7	Lufenuron	103055-07-8	5.12
8	Pendimethalin	40487-42-1	5.4
9	Pyridaben	96489-71-3	6.37
10	Tefluthrin	79538-32-2	6.4

Table 6 shows that there are 10 pesticides having potential to sink in sediment. In water quality monitoring programmes these pesticides are included for water matrix. However, they are not monitored in sediment matrix. Since agriculture is an important pressure for Beysehir lake, monitoring of these pesticides in sediment should also take place.

Discussion and Conclusion

In this study, the industries around Lake Beysehir were investigated and the chemicals that may originate from these industries were identified. Then, among these chemicals the ones that have the potential to accumulate in the sediment were detected by using octanol-water partition coefficient. Moreover, the pesticides specific to the basin were studied and the ones that should be monitored in the sediment were distinguished. According to these studies, it was found out that 18 priority substances, 16 specific pollutants and 10 pesticides should be monitored in the sediment.

This work revealed that monitoring of heavy metals are not sufficient to ensure the prevention of sediment contamination. There are other parameters that have high potential to sink in sediment. These parameters may act as a source of pollution to a water body. No matter how much the quality of water body is improved, with the emission of these parameters to the water body from the contaminated sediment, the quality of water body can deteriorate. Therefore, for the achievement of good water quality status, quality of sediment plays a crucial role and for the management of quality of water body, water and sediment should be considered in an integrated manner.

Moreover, by sediment monitoring, information about historical contamination can be obtained. In Turkey, there are water bodies that have no historical water quality monitoring data. With sampling in deeper sediment in these water bodies, information about historical contamination of the water body can be gained and precautions can be taken accordingly.

In Europe, by the reduction of point and diffuse pollution, sediment quality is tried to be improved. However, studies have shown that the reduction of pollution resulted in a slow and delayed response in sediment quality. Therefore, sediment quality management is a long term process and it should be conducted in basin scale. (SedNet, 2004)

In conclusion, except for some research studies, chemical monitoring of the sediment in Turkey is not included in the surveillance monitoring in line with the WFD requirements. This paper recommends that the places where the contamination of the sediment is likely to occur should be identified and the chemical parameters that have the tendency to accumulate in the sediment should be selected for monitoring in the each river basin of Turkey. Sediment monitoring programmes should be prepared for each basin in Turkey and monitoring activities should be conducted periodically in order to detect the trends of pollutants that are not soluble in water. Also, contamination of sediment by heavy metals should be further investigated in order to understand the source of heavy metals in sediment. This study is needed to distinguish the natural background concentration of heavy metals from anthropogenic impacts in sediment. Finally, it is clear that good water body status cannot be achieved without good sediment status. By emphasizing water and sediment linkage, sediment management plans should be prepared or sediment management issues should be included in river basin management plans.

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References

WFD group on Analysis and Monitoring of Priority Substances. (2004). Discussion Document on Sediment Monitoring Guidance for the EU Water Framework Directive. Retrieved from https://sednet.org/download/AMPS_sediment_monitoring_discussion_doc_v2.pdf

Chapman, P. M. (1996). Presentation and Interpretation of Sediment Quality Triad Data. *Exotoxicology, 5,* 327-339.

European Commission. (2000). *Water Framework Directive (WFD): Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy*. Official Journal of the European Parliament, L327, 22.12.2000.

European Communities. (2010). Common Implementation Strategy for the Water Framework Directive (2000/60/EC): Guidance document No. 25, Chemical Monitoring of Sediment and Biota Under The Water Framework Directive. *Office for Official Publications of the European Communities, ISBN 978-92-79-16224-4*. doi:10.2779/43586.

European Union. (2013). *Environment Quality Standard Directive: Dırectıve 2013/39/EU of The European Parlıament and of The Council of 12 August 2013 amending Directives 2000/60/EC and 2008/105/EC as regards priority substances in the field of water policy.* Official Journal of the European Union, L 226/16, 24.8.2013.

International Union of Pure and Applied Chemistry. (2019). Pesticide Properties Database. Retrived from https://sitem.herts.ac.uk/aeru/iupac/search.htm

Brils, J. (2008). Sediment monitoring and the European Water Framework Directive. *Ann Ist Super Sanità, 44(3)*, 218-223.

The Ministry of Forestry and Water Affairs of Republic of Turkey. (2013). *Basin Protection Action Plan for Konya Basin.* Kocaeli: TÜBİTAK Marmara Research Center.

The Ministry of Forestry and Water Affairs of Republic of Turkey. (2014). *Project on Basin Monitoring and Determination of Referance Points Report*. Ankara: Segal Environmental Measurement and Analysis Laboratory Engineering Consultancy Ltd.

SedNet. (2004). Contaminated Sediments in European River Basins Report. Retrived from https://sednet.org/work/contaminated-sediments-in-european-river-basins/

United States Environmental Protection Agency. (2019). ATSDR: Toxic Substances Portal Chemical List. Retrived from https://comptox.epa.gov/dashboard/chemical_lists/ATSDRLST

Extended Turkish Abstract
(Genişletilmiş Türkçe Özet)

Beyşehir Gölü Sedimanında Kimyasal İzleme Parametrelerinin Seçimi

Suda çözünmeyen hidrofobik maddelerin, zaman içerisinde çökerek dipte bulunan sedimentte birikme potansiyeli vardır. Sadece su kolonunda yapılan izleme çalışmaları ile bu hidrofobik maddelerin tespiti yapılamamaktadır. Ayrıca, su kalitesi ne kadar iyileştirilirse iyileştirilirsin zaman içerisinde bu kimyasal maddelerin tekrar su kolonuna geçerek su kalitesini kötüleştirme ihtimali vardır. Bununla birlikte, sedimentte yapılan dikey örnekleme ile geçmişe dönük kirlilik hakkında da bilgi sahibi olunabilmektedir. Bu sebeple, etkin su yönetimini sağlamak için sedimentte de kimyasal izlemeye yer verilmesi gerekmektedir.

Göllerin ve nehirlerin bütüncül bir parçası olması ve su kalitesindeki önemine rağmen, sedimentte kimyasal izlemenin önemi Avrupa Birliği Su Çerçeve Direktifinde (SÇD) yeterli şekilde vurgulanmamıştır. SÇD de, sedimente çoğunlukla Çevre Kalite Standartlarında (ÇKS) yer verilmiştir. SÇD açıkça ÇKS'lerin su, sediment ve biyota için belirlenmesi gerektiğini belirtmektedir. Ayrıca, Avrupa Sediment İzleme Ağı (SedNet) havza yönetim planlarında sedimente de yer verilmesi gerektiğini vurgulamaktadır.

SÇD'de sediment izleme sıklığı net bir şekilde belirtilmemektedir; fakat 2013/39/EU Direktifinde ÇKS'lerin uygulanabilirliği için izlemenin yılda bir kez yapılması gerektiği bildirilmektedir. Bu sıklık sediment için sedimentasyon oranı ve su kütlesinin hidrolojik rejimine göre değişiklik gösterebilmektedir. İzleme zamanı olarak ise sedimentasyon hızının en fazla olduğu, akış hızının ise en az olduğu zamanlar olan yaz dönemleri seçilmelidir.

Sediment izleme istasyonlarının seçimi, su kütleleri izleme istasyonları ile benzerlik göstermektedir. Seçilen izleme noktaları su kütlelerini temsil edici özellikte olmalı ve izleme noktaları seçilirken noktasal kirletici kaynaklar dikkate alınmalıdır.

Bu çalışma ile Beysehir Gölü sedimentinde izlenmesi gereken kimyasal maddelerin tespit edilmesi hedeflenmiştir. Öncelikle göl etrafında yer alan baskı türleri incelenmiş olup, baskıların başlıca tarım ve hayvancılık kaynaklı olduğu gözlemlenmiştir. Ayrıca, göl etrafında organize sanayi bölgesi ile gölün yakınında şeker fabrikası olduğu görülmüştür. Organize sanayi bölgesi içerisinde yer alan tesis türleri de incelenmiş, bu tesislerin çoğunlukla makina imalatı, otomotiv ve silah endüstrisi üzerine olduğu tespit edilmiştir. Bu tesislerden kaynaklanan kirleticiler incelenerek sedimentte birikme potansiyeli olanlar tespit edilmiştir. Ayrıca, Konya Havzasında yetiştirilen bitkilerde kullanılan pestisitlerin listesi ile öncelikli maddeler de incelenerek hidrofobik olanlar belirlenmiştir.

Sedimentte izlenmesi gereken kimyasal maddeler tespit edilirken oktanaol-su ayrışım katsayısından (K_{ow}) yararlanılmıştır. Kural olarak; log K_{ow} değeri 5'ten büyük olan kimyasal maddeler sedimentte izlenmesi gereken kimyasal maddeler olarak tespit edilmiştir. Sedimentte izlenmesi gereken metallerin seçiminde ise; log K_{ow} değeri, organik maddelerin hidrofobik özelliğin belirlenmesinde kullanılması sebebiyle dikkate alınmamıştır. Ağır metaller hidrofobik özelliğe sahip oldukları için sedimentte izlenmesi gereken kimyasal maddeler listesine eklenmiştir.

Beyşehir Gölü sedimentinde izlenmesi gereken kimyasal maddeler seçilirken, 2014 yılında Havza İzleme ve Referans Noktalarının Belirlenmesi Projesi kapsamında Beyşehir Gölünde dört dönem gerçekleştirilmiş olan su kalitesi izleme sonuçları ile sedimentte gerçekleştirilmiş olan ağır metal izleme sonuçları birarada incelenmiştir. Sonuçlar değerlendirildiğin de su kolonunda konsantrasyon değeri düşük olan ağır metallerin sedimentte daha yüksek konsantrasyon değerine sahip olduğu gözlenmiştir. Ayrıca, izleme sonuçları değerlendirilirken, sedimentte alüminyum konsantrasyon değerinin çok yüksek olduğu gözlemlenmiştir. Bu konsantrasyon değerinin sedimentin doğal yapısından kaynaklandığı, kirlilik göstergesi olmadığı yorumu yapılmıştır. Bu sebeple, sedimentte izlenmesi gereken kimyasal maddelerin seçiminde sedimentin doğal yapısından kaynaklanan maddelerin belirlenmesinin önemi vurgulanmıştır.

Sonuç olarak, "Tehlikeli Madde Kirliliğinin Kontrolüne İlişkin Proje", "Ülkemiz Kıyı ve Geçiş Sularında Tehlikeli Maddelerin Tespiti Projesi", "Ekolojik Kıyı Dinamiği Projesi" ve "Bitki Koruma Ürünlerinin Kullanımı Neticesinde Meydana Gelen Su Kirliliğinin Tespiti ve Madde veya Madde Grubu Bazında Çevresel Kalite Standartlarının Belirlenmesi Projesi" ile Türkiye için saptanan 117 belirli kirletici ve 133 bitki koruma ürünleri arasında Beyşehir Gölü sedimentinde izlenmesi gereken 16 belirli kirletici ve 10 pestisit belirlenmiştir. Ayrıca, 18 öncelikli maddenin de sedimentte izlenmesi önerilmiştir. Bu kimyasal maddelerin Beyşehir Gölünde yapılacak olan izleme çalışmalarına dahil edilmesi ve Türkiye için diğer havzalarda da sedimentte izlenmesi gereken kimyasal madde listesinin oluşturularak havza izleme programlarına sedimentte izlemenin de dahil edilmesi önerisinde bulunulmuştur. Ayrıca iyi su durumunun iyi sediment durumu olmadan mümkün olmadığı belirtilerek Türkiye için sediment yönetim planlarının hazırlanması ya da havza yönetim planlarına sediment konusunun dahil edilmesi önerilmiştir.

Research Article

Effects of Irrigation and Field Management Practices within Water Resources Systems

Sulama ve Arazi Yönetimi Uygulamalarının Su Kaynakları Sistemlerindeki Etkileri

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Abstract

The world population is foreseen to increase up to 9.8 billion people toward 2050, and global food and water demands can also be predicted to rise accordingly. Regarding these future demands, climate change and depletion in water resources; new approaches, management strategies, and models are needed. In this study, the AquaCrop model was used as an analytical tool to predict the effects of management practices within winter wheat, spring wheat, winter barley, and maize in a specific location, middle Guadiana sub-catchment, Spain. The primary drivers from the model were designated as actual evapotranspiration, crop yield, and water productivity. Model runs were executed within three different management strategies: irrigation technologies, irrigation strategies, and mulching practices. Thereafter, yield gaps and water productivity gaps were analyzed, and water scarcity/shortage degrees were compared. The results showed that the AquaCrop model is a versatile model to estimate actual evapotranspiration, crop yield, and water productivity parameters. Yield productions in deficit irrigation were found higher than supplementary irrigation. Full irrigation showed the highest crop yield within non-limited water conditions. However, some negative impacts of the full irrigation strategy such as salinity should be considered. Mulching practices positively affected the actual evapotranspiration reduction. Full irrigation and no mulching scenario showed the worst results on the water resources systems. Supplementary irrigation and synthetic mulching practices depicted the least deterioration of surface water resources. Deficit irrigation and synthetic mulching practices resulted in considerable water savings with fewer yield losses compared to the scenario with the highest yield production levels.

Keywords: *AquaCrop model, management practice, water productivity, yield gap, water scarcity/shortage degrees*

Öz

Dünya nüfusunun 2050'de 9,8 milyar kişiye ulaşacağı ve bu artışla eş zamanlı olarak küresel ölçekte gıda ve su taleplerinin de artacağı öngörülmektedir. Gelecekteki bu taleplere ek olarak iklim değişikliği ve su kaynaklarının tükenmesi durumları da dikkate alındığında; yeni yaklaşımlar, yönetim stratejileri ve modellerin geliştirilmesine ihtiyaç duyulmaktadır. Bu çalışmada, belirlenen bir bölgede (orta Guadiana alt havzası, İspanya) üretilen kış buğdayı, bahar buğdayı, kışlık arpa ve darıdaki yönetim uygulamalarının etkilerini tahmin etmek için AquaCrop modeli, bir analitik araç olarak kullanılmıştır. Modeldeki birincil sürücüler gerçek evapotranspirasyon, mahsül verimi ve su verimliliği olarak belirlenmiştir. Model çalışmaları sulama teknolojileri, sulama stratejileri ve malçlama uygulamaları olmak üzere üç farklı yönetim stratejisinde yürütülmüştür. Daha sonra, mahsül verimi açığı ve su verimlilik açığı analiz edilmiş, su kıtlığı/yokluğu dereceleri karşılaştırılmıştır. Bu çalışma, gerçek evapotranspirasyon, mahsül verimi ve su verimliliği parametrelerini tahmin etmek için AquaCrop modelinin kullanışlı bir model olduğunu göstermiştir. Kısıntılı sulamada mahsul üretimi, tamamlayıcı sulamaya kıyasla genellikle daha verimli bulunmuştur. Tam sulama, sınırlandırılmamış su koşullarında en yüksek verimi göstermiştir. Ancak, tam sulama stratejisinin tuzluluk gibi diğer olumsuz etkileri de dikkate alınmalıdır. Malçlama uygulamaları, gerçek evapotranspirasyon azalmasını olumlu yönde etkilemiştir. Tam sulama ve malçlama uygulanmayan senaryo, su kaynakları sistemleri üzerinde en olumsuz etkiyi göstermiştir. Tamamlayıcı sulama ve sentetik malçlama uygulamaları, yüzeysel su kaynağı üzerine en düşük etkiyi göstermiştir. Kısıntılı sulama ve sentetik malçlama uygulamaları, en yüksek üretim seviyelerine sahip senaryoya göre daha az mahsül üretimi kaybıyla dikkate değer su tasarrufu sağlamıştır.

Anahtar kelimeler: AquaCrop modeli, yönetim uygulaması, su verimliliği, verim boşluğu, su kıtlığı/yokluğu dereceleri

Introduction

The world's population is expected to reach 9.8 billion people in 2050, which is 2.2 billion more people than 2020 according to the United Nations (UN), and global food and water demands can also be foreseen to increase accordingly. The agricultural sector has a substantial water use dimension amongst other sectors with nearly 70%, and global warming originates crucial impacts on crop water productivity (Patel et al., 2017; Kang et al., 2009). Due to the increased population of the world, food demand and water use have been dramatically increasing for over several decades. Therefore, crop yields must be higher to eradicate issues on food security to ensure adaptation to different drivers like socioeconomic developments, climate change, and water resources depletion (Van Ittersum et al., 2013). Several studies have focused on irrigation management strategies to increase either crop yield or water productivity under limited available water for sustainable productions (Chukalla et al., 2015). Crop yield response to water was described by Food and Agriculture Organization of the United Nations (FAO) as optimizing rainfed and irrigated agriculture at field levels. Because of costly management practices and experiments on the field, crop development models are needed within different factors such as irrigation techniques, soil types, crop types, climatic conditions, and management strategies. Hence, a dynamic model is needed, such as AquaCrop, which provides simplicity, robustness, and accuracy including climatic, soil characteristics, and management practices for agricultural irrigation.

Crop yield (**Y**) can be described as the harvested production per harvested area unit for crop commodities (OECD, 2015). Regarding crop yield, due to increased food demand and other abovementioned reasons, water resources systems can be considered. Yield gap **(Y_g)** is an important parameter that can be described as a calculation of the differences between actual farmers` yield and potential yield without limitation from water and management practices. Yield gap analysis can be done by field experiments or simulation models to estimate yield gap at different scales (i.e. regional, national, or global) (Wart et al., 2013). According to Global Yield Gap Atlas (**GYGA**) (2017), yield gap (Y_g) analysis is one of the methods that can be applied ranging from local to a regional extent for agricultural sustainability, and described as a difference between potential yield (**Y_p**), water-limited yield potential (**Y_w**) or partially-irrigated yield potential (**Y_{pi}**), and actual yield (**Y_a**) (GYGA, 2017). In addition to this analysis, impacts of different adaptation pathways in agricultural irrigation on the water balance may have a momentous benefit for the future.

Water productivity (**WP**) is a measure of the efficiency of water resources that support rainfed and agricultural irrigation, and can be defined as how much yield output is obtained per cubic meter of fresh water abstracted (Smakhtin et al., 2004). Water productivity calculation may provide the efficiency with which water is converted to food, and which resource can be used effectively (GYGA, 2017). Besides, Water Footprint (WFP) concept is an inverted version of WP (m^3/kg). According to Hoekstra et al. (2011), the water footprint is an indicator of freshwater use that looks at both direct and indirect water uses by consumer or producer. Irrigation Water Use Efficiency can be defined as the ratio of the net irrigation water requirement and the total amount of water that needs to be withdrawn from the source (Döll & Siebert, 2002). Harvest Index (**HI**) is explained as the plant capacity to allocate biomass (**B**) into the formed reproductive parts (Wnuk et al., 2013).

Water Balance is another significant analysis which can be affected either positively or negatively by results of different artificial applications. A general equation can be described for sub-catchment scale regarding surface water body as the accumulation of the stream flow (**Q**), evaporation (**E**), abstractions and storage changes per time equal to return flows **(ΔS)**, precipitation on the Earth system (**P**) (Uhlenbrook & Savenije, 2017). After changing the agricultural management practices, irrigation water requirements might be more or less, and the effects on the systems may show differences.

Water scarcity can be described as a lack of sufficient available water resources to meet the demands of freshwater to produce food, to supply industries, and to sustain inhabitants in the world within different specific scales (i.e. regional, national or global) (Hoekstra et al., 2012). Water Scarcity Index **(WSI)** is one of the indicators that ensure assessing the water scarcity/shortage/stress degrees (Falkenmark et al., 1989). Water scarcity analysis is crucial to understand the stress on the water resources systems and might help to select the proper adaptation pathways to assess not only the climate change impacts, also to assess the yield and water productivity for specific locations.

In this study, the aim was to investigate the soil and plant interactions regarding yield and water productivity in the agricultural sector for selected locations (sub-catchments) and certain crop productions within different irrigation technologies, irrigation strategies, and mulching practices by using AquaCrop model.

Method

Study Area

The research area is sub-river basins of Guadiana river basin, the middle Guadiana and Portugal area Guadiana. The Guadiana river basin indicates the starting point of the border between Spain and Portugal, and it becomes an international river basin between two countries. According to Chukalla et al. (2015), dominant soil profiles in Badajoz are loam, sandy loam, and silty clay loam. The main reason for selecting middle Guadiana as study area was that agricultural activities were the second highest water user in this region, and it is also the starting point of the delineation of the Spain-Portugal border.

In addition to those parameters, station-specific data was collected from an online database (Tank et al., 2002) regarding meteorological data from weather station, and also data related with hydrological and water use sectors was provided from specific studies on the Guadiana river basin (GuaSEEAW, 2015). The study area in the Guadiana river basin is illustrated in Figure1 (partly taken from Camacho et al., 2014 and modified).

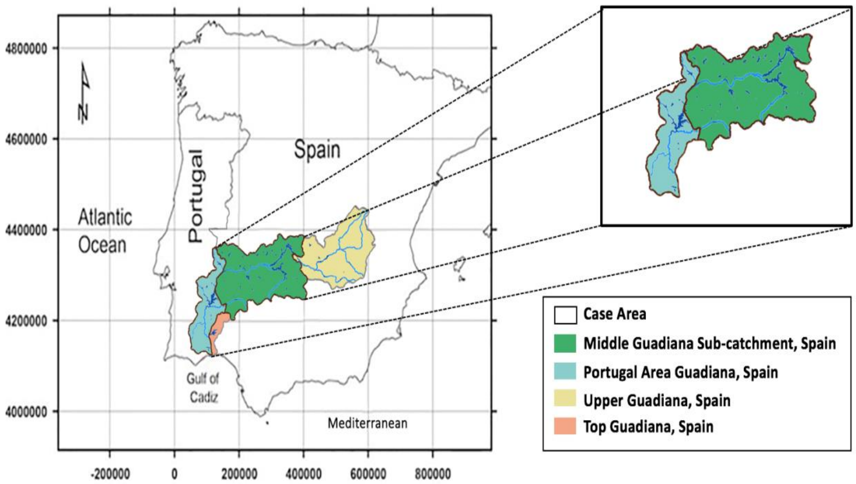

Figure 1. The study area in Guadiana river basin.

In selected area, irrigation technology was divided into three sections: sprinkler irrigation with 22%, localized irrigation with 23%, and surface flood irrigation with 54% (Aldaya and Llamas, 2008). Furthermore; dry, normal, and wet climates of the last ten years from the year of the study were found as 2015, 2009, and 2010, respectively; and model runs were executed for the normal year, 2009. According to the Spanish Ministry of Agriculture, Fisheries and Food (MAPAMA, 2010), on one hand, wheat and barley cultivations were based on rainfed irrigation (83% of the total selected area), only 2% was irrigated in 2009. On the other hand, 15% of the study area was irrigated for maize production without rainfed irrigation. In addition to this, while groundwater is dominant in the water system in the upper Guadiana, the middle Guadiana has surface water dominated areas (Aldaya and Llamas, 2008). Hence, in this study, we focused on surface water bodies more than the others while making water balance analysis. Table 1 shows the selected herbaceous crops such as maize, wheat, and barley, and irrigated crop calendars for the selected crops.

Table 1

Irrigated Crop Calendar (FAO, 2017)

	Jan	Feb	March	April	May	June	July	August	Sep	Oct	Nov	Dec
Winter Wheat												
Spring Wheat												
Maize												
Winter Barley												

Step-1: AquaCrop Model

In this study, AquaCrop model (version 6) was used. AquaCrop is a dynamic model providing a simulation on the interaction between soil and crop, which is mainly divided into two sections regarding location- and user- specific parameters (Steduto et al., 2012). Location-specific parameters are climate and soil features, and user-specific settings are crop cultivar perception, the timing of crop cycle, water management, and agronomic practices. AquaCrop model performs the simulation robustly for herbaceous crops within a single growth cycle by the calculation of biomass production and final crop yield, which is to predict the crop yield at a field (point simulations). Herbaceous crops are a strong side of AquaCrop model. The field is presumed to be uniform. Solely vertical incoming such as precipitation, irrigation, and capillary rise and outgoing (evaporation, transpiration, and deep percolation) water fluxes can be taken into account. AquaCrop uses Penman-Monteith method to calculate reference evapotranspiration (ET_0). Furthermore, water productivity was normalized in the model for air CO_2 concentrations and atmospheric demand (WP*).

The main equations of the model parameters can be seen below:

$$B = WP^{*} \times \Sigma(Tr/ET_0) \tag{1}$$
$$Yield = B \times HI \tag{2}$$
$$WP = Y/ET \text{ (kg (yield)/m}^3\text{ (ET))} \tag{3}$$
$$WP = B/\text{water applied (kg (biomass)/m}^3\text{ (Tr))} \tag{4}$$

where B= Biomass; WP= Water Productivity; WP*= normalized WP; HI= Harvest Index; Y= Yield; Tr= Transpiration; ET_0=reference evapotranspiration.

Irrigation techniques.

Irrigation is an artificial way to provide water for crop production. Irrigation methods can be varied depending on energy or pressure requirements, or the specific techniques regarding wetted areas (Chukalla et al., 2015). The AquaCrop model has different options for users, for example, irrigation technologies within their efficiency and wetted area rates, which can be adjusted in accordance with the technology. Irrigation technologies were chosen as sprinkler, drip, and furrow irrigation in this study. Some of the rates of the irrigation efficiency and wetted area can be seen in Table 2 for different irrigation techniques.

Table 2

Efficiency Rates (IE) and Wetted Areas for Different Irrigation Techniques

Techniques	Çakmak et al. (2008)	FAO (2012)[1]		Aldaya &Llamas (2008)
	IE	IE	Wetted Areas	IE
Sprinkler	70%	75%	100%	70%
Drip	90%	90%	30%	90%
Surface (Furrow)	40%	60%	80%	50%

Note. [1] Retrieved from http://www.fao.org/docrep/t7202e/t7202e08.htm

From these efficiency rates for specific techniques, distribution of the irrigation technologies was given as 23% of the irrigation for drip, 22% for sprinkler, and 54% for furrow irrigation within the middle Guadiana. Thus, the water balance analysis was conducted according to assumptions of these distributions.

Irrigation strategies.

Full irrigation **(FI)** is the application of the irrigation during plantation applying water into the system for ensuring evaporative demand to increase yield within a no water stress condition (Chukalla et al., 2015). Irrigation can be applied to a certain amount periodically or after water depletion on a certain readily available water (RAW%) depletion. The AquaCrop model helps to users for choosing RAW% threshold. When a certain crop type is selected in the model, which guides its users for selecting the correct thresholds among affected canopy expansion, stomatal closure, and senescence acceleration.

Deficit irrigation **(DI)** can be explained as when the water is limited in the area of agricultural activities; optimal water application can provide efficient amount of water according to the research and technology innovations based on optimal yield and water productivity (Hamdan et al., 2006). Unlike FI strategy, DI strategy can be applied less than evaporative demand within limited water applications among less water shortage sensitive periods of the crop development.

Supplementary irrigation **(SI)** is a method that the certain amount of water applied to increase yield and water productivity when the crop growth under insufficient rainfed conditions. During the critical stages, such as lack of soil moisture within dry periods, SI can be applied to ensure important improvements in the yield and water productivity. SI can ensure to achieve good performance when the timing is selected correctly. According to Pereira et al. (2012), SI provides irrigation which does not provide to reach crop water requirements as much as FI during insufficient rainfed conditions. At this point, less crop water is provided compared to FI and DI. When the water is limited, DI strategies can be more effective for farmers to increase WP rather than yield increase. At that point, more water can be available for more lands to be cultivated. SI is a kind of managed DI, and irrigation events can be done less than others among the stress conditions especially during the critical crop development stages to eradicate stress effects on the crop (Ewans et al., 2008). SI is a method to provide water when the dry spell occurs, and the water stress is observed during the development stage. Irrigation can be applied to increase soil moisture during dry periods for rainfed lands. Thus, SI is a remarkable strategy to increase water productivity and yield.

Mulching practices.

Using mulches in the crop cultivations provides decreases soil evaporation; besides, fewer impacts on transpiration occurs through plants. Organic, synthetic, and no mulching applications with a different surface coverage rates were considered in the study. The decrease in soil evaporation can be seen in Figure 2 by Zhang et al. (2002). Hamdan et al. (2006) refer to mulching agronomic practices as one of the evaporation reduction methods in addition to select correct timing for planting or drip irrigation. In the AquaCrop model runs, organic mulching with 100%, and synthetic mulching with 80% surface coverage were assumed for all the practices (Chukalla et al., 2015).

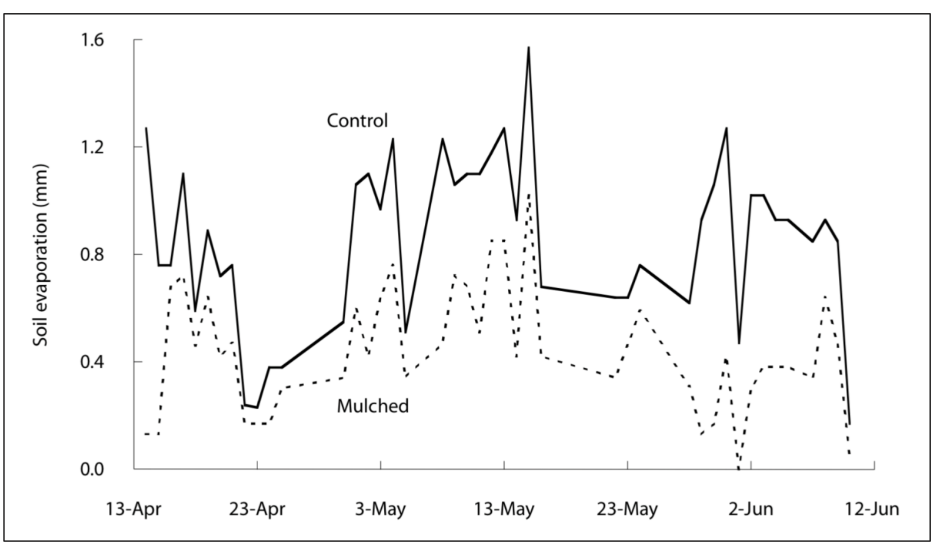

Figure 2. Effect of mulching on soil evaporation for winter wheat cultivation.

Step-2: Yield Gap (Y_g) and Water Productivity Gap (WP_g) Analysis

Yield gap (Y_g) is the difference between potential (Y_p) or water-limited yield (Y_w), and actual yield (Y_a). Y_p is the yield during the cultivation of a crop by a cultivar when the crop development is achieved by the proper climatic conditions, non-limited nutrients, and well-controlled biotic stress (Van Ittersum et al., 2013). Y_a is observed yield from the actual amounts in the field (GYGA, 2017). Y_w is more relevant to the benchmark for rainfed crops. Both Y_p and Y_w can be used as benchmarks for supplementary irrigated crops. The only difference between Y_p and Y_w is that Y_w is also dependent on soil characteristics and limited water for irrigation applications. Y_p and Y_w information can be obtained from models. In this study, the results from GYGA and AquaCrop showed the difference between two models, which have different management strategy categories and modelling characteristics. Besides, harmonization of both Y_p and Y_w could be better for yield gap analysis (GYGA, 2017). Furthermore, the difference between Y_p and Y_a came from precipitation and soil profile. To estimate WP_g, the methodology was used as same as the Y_g analysis. Furthermore, exploitable yield is defined as the difference between 80% of Y_p (or Y_w) and Y_a (Van Ittersum et al., 2013).

Step-3: Water Balance on Water Resources Systems

Water balance analysis begins with the calculation of the field level water balance within the soil water balance. The consumptive water uses (CWUs) within green (rainfed) and blue (irrigated) CWUs were upscaled among the upstream part to see impacts on both upstream and downstream later with WSI. This first step was executed through the AquaCrop model output. Reference data for hydrological information was taken from GuaSEEAW (2015) and Automatic System of Hydrological Information (SAIH) (2017). The next step was the upscaling of the field level CWUs for a sub-catchment level within harvested area-based calculations. Lastly, upstream changes in the water balance and the pressures on the downstream scale can be investigated within available data from the stations or reference sources. A visualization example of the water balance for current study at catchment level can be seen in Figure 3. Other sectoral water uses (such as domestic and industrial) were kept constant as in today`s world (GuaSEEAW, 2015). The strategies were chosen by different management practices according to the larger to smaller effects on the water resources system. Both withdrawal and consumptive water uses were considered in this study. Water withdrawal based calculations indicate uncertainties regarding the water losses in water distribution for the sectoral water demands. For this reason, the methodology was updated to use model results more accurately through CWUs. Water losses have significant importance on the estimations regarding the impacts on the defined water balance. According to Aldaya and Llamas (2008), water losses during the water distribution were assumed as approximately 30%. Instead of this average value, CWUs were assumed in this section, and WSIs were calculated by a water consumption-based analysis. Assumptions to execute water balance part are briefly given below:

- The contributions of the selected strategies to the water balance were based on the reference. The representation of the current cultivation types (rainfed winter wheat, rainfed winter barley, irrigated maize) were calculated within the consideration of other agricultural activities (i.e. olive trees, vegetables, industrial crops) among the middle Guadiana regarding the CWU. Two types of the CWUs were calculated as green CWU and blue CWU through ET_a values from AquaCrop simulations. To analyze the water resources systems, blue CWU parameters were mainly used.

- Aldaya and Llamas (2008) stated the usage of different irrigation technologies for sprinkler with 22%, furrow with 54%, and drip with 23%. Those proportions were assumed in this study, and the combinations of the different mulching (NM, OM, SM) and irrigation strategies (FI, DI, SI) were used to illustrate impacts into the water balance. The calculated and changed outflow from the upstream part was accepted as additional inflow for the downstream region.

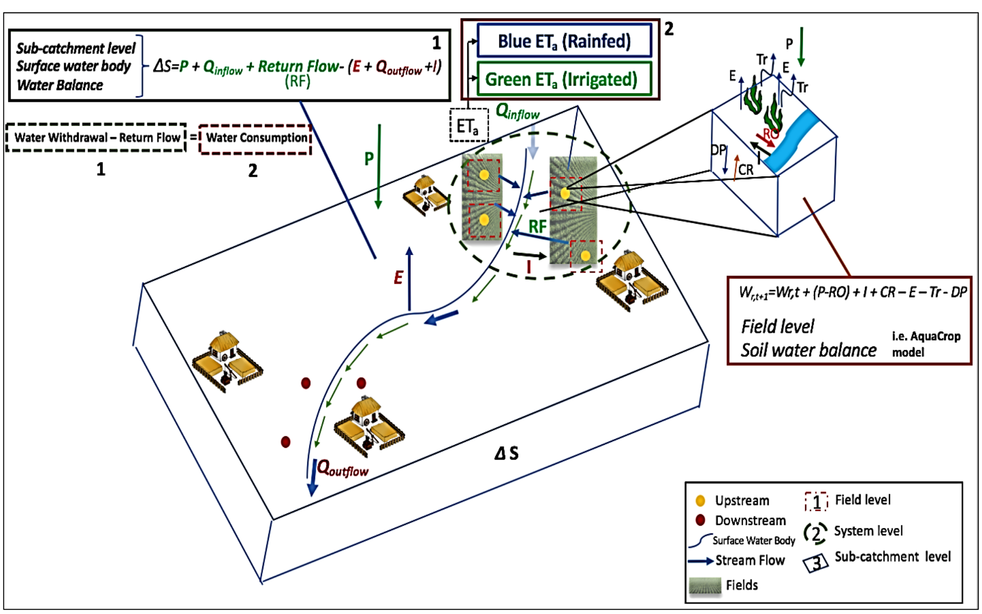

Figure 3. A visualization example of the water balance for study area at catchment level (ΔS= net change in storage; P= precipitation; E = Evaporation; Q= surface water flow; RF= Return Flow; I= Irrigation; Tr = Transpiration; ET_a= actual evapotranspiration; CC = Capillary rise; DP = Deep percolation).

Step-4: Water Stress/Scarcity Index (WSI)

Water stress/scarcity degree can be calculated within the available data for different management strategies in the agricultural system and their impacts on the water resources systems (equations 5 and 6). The illustration of the WSI application for the selected sub-basins and key flow parameters can be seen in Figure 4. It is precise that the WSI variation can be smaller regarding projections for a small proportion of the selected crops amongst all sectoral activities in the middle Guadiana. However, this analysis gives an idea regarding how different practices could depict various consequences on the water resources systems` stress degrees. The reference data in the study area was from different sources regarding other sectoral water uses and water dimensions (stream inflows) (GuaSEEAW, 2015).

Water Scarcity Index (WSI) was implemented by the equations below for upstream and downstream to compare different strategies, respectively:

$$WSI_{Spain} = \frac{\text{Water Consumption} - \Delta\,\text{Blue ETa,Spain}}{\text{Stream Inflow ,upstream}} \quad (5)$$

$$WSI_{Portugal} = \frac{\text{Water Consumption}}{\text{Stream Inflow ,downstream} + \Delta\,\text{Blue ETa,Spain}} \quad (6)$$

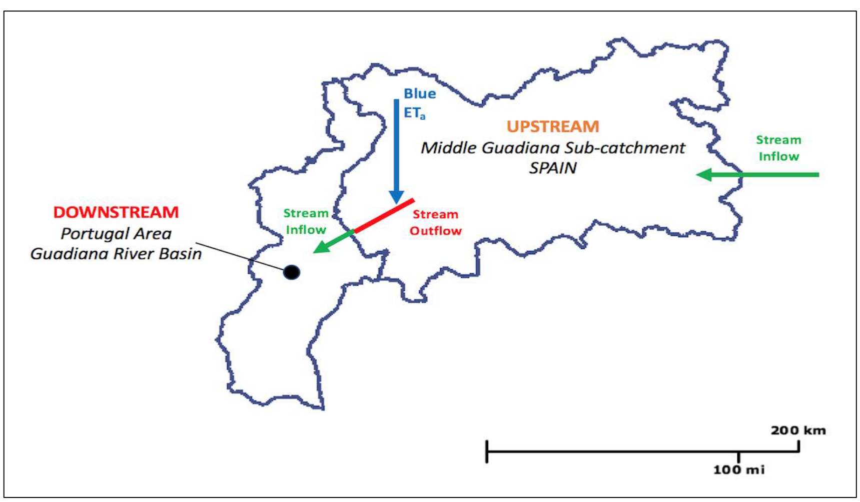

Figure 4. The illustration of the WSI application for the study area and key flow parameters.

Stepwise Approach

The stepwise approach of the study can be seen in Figure 5 including initial and main implementation phases of the research including above-stated steps.

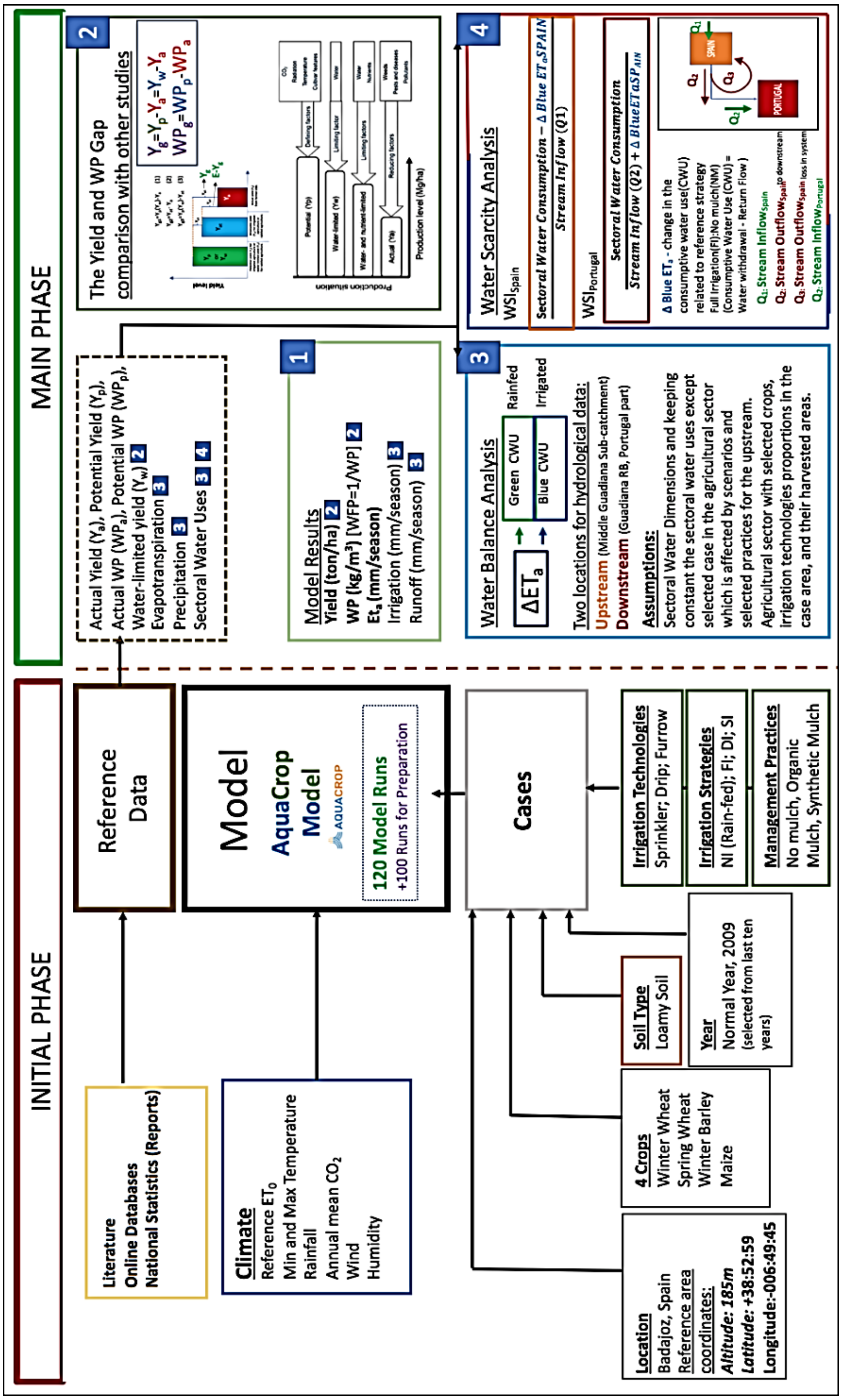

Figure 5. An overall flow diagram of the research and key parameters.

Results and Discussion

Model Results Regarding ET_a, Y, and WP within Different Management Practices by Using AquaCrop Model

In this section, actual evapotranspiration (ET_a), yield (Y), and water productivity (WP) model results were given with different illustrations according to the various purposes for selected crops which are winter wheat, spring wheat, winter barley, and maize. Y&ET_a and WP&ET_a for four crops were compared within all different management practices and significant correlations were found (Appendix). WP and ET_a results for four crops showed that the ET_a results lower than 300 mm were related to the rainfed cultivation for spring wheat and maize. The results higher than 500 mm were related to irrigated agriculture for the same crops. A total ET_a for wheat production can differ between 200-500 mm (Chukalla et al., 2015), and it was found that the range of ET_a production was in the simulated results with this scope (Appendix). The relationship between Y and ET_a showed a production curve that is increasing and leveling off with a high correlation value (0.99 R^2) for spring wheat and maize. Furthermore, the relationship between Y and ET_a; and WP and Y were found weak in winter wheat and winter barley production compared to maize and spring wheat production. Figure 6 depicts that a declining linear trend on ET_a for selected crops. Increasing trend of WP and main ordinal ranking was found as a following trend of NM, OM, SM, respectively. The effects of mulching practices depicted a decreasing trend on ET_a due to the increase of surface areas and a decrease in mainly soil evaporation values.

The reduction of transpiration values was found less affected compared to soil evaporation changes. Winter wheat and winter barley results showed less ET_a amounts compared to spring wheat and maize. The main reason of that these cultivations were rainfed based and mulching practices showed more impacts than other strategies (irrigation strategies did not exist in rainfed). However, when we look at the ranking of WP from smaller to larger, irrigated crops (maize and spring wheat) did not show a trend as found under rainfed conditions. It can be seen that leading drivers of the increasing in the WP is caused by mulching and irrigation strategies. The most significant ET_a deviations were found in the maize and spring wheat applications, whereas winter barley and winter wheat had fewer variations. Management practices without mulches depict higher ET_a values compared to organic and synthetic mulching practices as expected. The lowest ET_a value was seen in the synthetic mulching. The lower ET_a and WP values (extreme values) in both figures are related to rainfed cultivations for maize and spring wheat (Figure 6). There was an increasing trend on

WP which has less increasing trend in spring wheat, and other crops show similar increasing rates. The most substantial deviation among the mulching practices related to the WP changes was found in the rainfed maize.

As it is stated in the AquaCrop model manual (Steduto et al., 2012), yield values depict the preference of the cultivation types either rainfed or irrigated ones. It is clear that rainfed conditions were not appropriate for maize and spring wheat which illustrate extremely low yield productions under rainfed conditions. However, when we compared irrigated maize and irrigated spring wheat for a selection of more profitable options for farmers, maize production as modern producers` choices in the area, gives approximately double yield amounts. Hence, rainfed winter barley and rainfed winter wheat productions were better options as in the current situation. When any selection is needed to be done between irrigated maize and irrigated spring wheat, maize appears as an optimal selection because of its higher yield values. Different irrigation strategies had different yield responses because of the fewer water applications, from highest to lowest amounts by FI, DI, and SI, respectively, during the irrigation period. While rainfed maize was not applicable in the study area, yield from the irrigated maize was substantially more than other considered crops within the study. Spring wheat trials showed that the spring wheat production was not an efficient way for crop production compared to maize, but it still can be considered as an option for farmers whether they would like to cultivate their fields when there is an available time in addition to the present cultivations in the basin. Spring wheat might also provide strategy options in the future. It is not only for spring wheat; other crop types can be simulated in further studies for additional crop pattern alternatives.

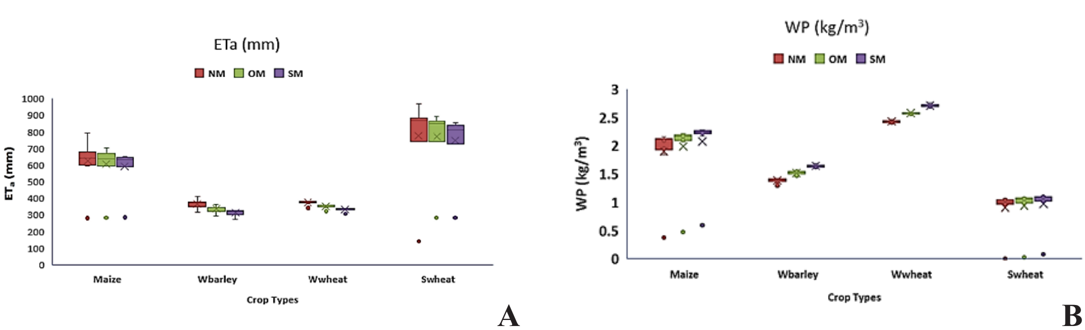

Figure 6. The actual evapotranspiration (ET_a) changes (A) and water productivity (WP) changes (B) for four crops within different mulching practices.

Figure 7 shows the different trends regarding ET_a, Y, WP, irrigation amount (I), and harvest index (HI). The highest difference between rainfed and irrigated agriculture was found for maize and spring wheat. On the other hand, there is no significant difference on yield parameter for winter barley and winter wheat. HI values showed insufficient rainfed conditions for maize and spring wheat; therefore, irrigated agriculture for those crops was inevitable. HI values were found nearly 0.5 for maize (0.5 in Steduto, 2012), 0.30-0.35 for winter barley (0.45-0.5 for modern producers in in Steduto, 2012), and 0.45-0.50 for winter wheat in this study (0.2-0.55 in Steduto, 2012). Due to the insufficient environmental conditions, lower amounts can be seen compared to literature information regarding the HI of certain crops which shows the comparability of the productions (Steduto et al., 2012). Different irrigation strategies have different yield responses because of the less water applications during the irrigation period. It is clear that fewer irrigation amounts were implemented by FI, DI, and SI in the simulations, and taken order from highest to lowest amount with FI, DI, and SI, respectively. In later sections, different strategy impacts on the water resources systems were given with selected strategies for current applications in the selected location.

The Comparison of the Yield Gap (Y_g) and Water Productivity Gap (WP_g) with Other Studies

Rainfed yield gap comparisons.

Figure 8 illustrates the yield gap comparisons of the model results with GYGA, and water-limited yields (Y_w) from the AquaCrop model and actual yields (Y_a) comparisons given to depict differences between yield gaps according to overall Y_g for a certain area in GYGA and different Y_g performances within different strategies. Only rainfed winter wheat and winter barley comparisons were given because of insufficient rainfed conditions for spring wheat and maize cultivations. For rainfed winter wheat, due to declining trend of Y_g in parallel with the estimated potential yield. According to the MAPAMA (2010), the same situation for Badajoz in modern cultavitions do exist, and only the irrigated maize production was carried out in 2009 among selected crops. When the simulated (AquaCrop) Y_ws results were compared to GYGA results (both Y_w, Y_a from GYGA), a significant decrease was found on Y_g. Synthetic mulching (SM) based field studies depict higher Y_ws for both crops, therefore, yield gap increased due to the more efficiently crop yield production from SM compared to NM.

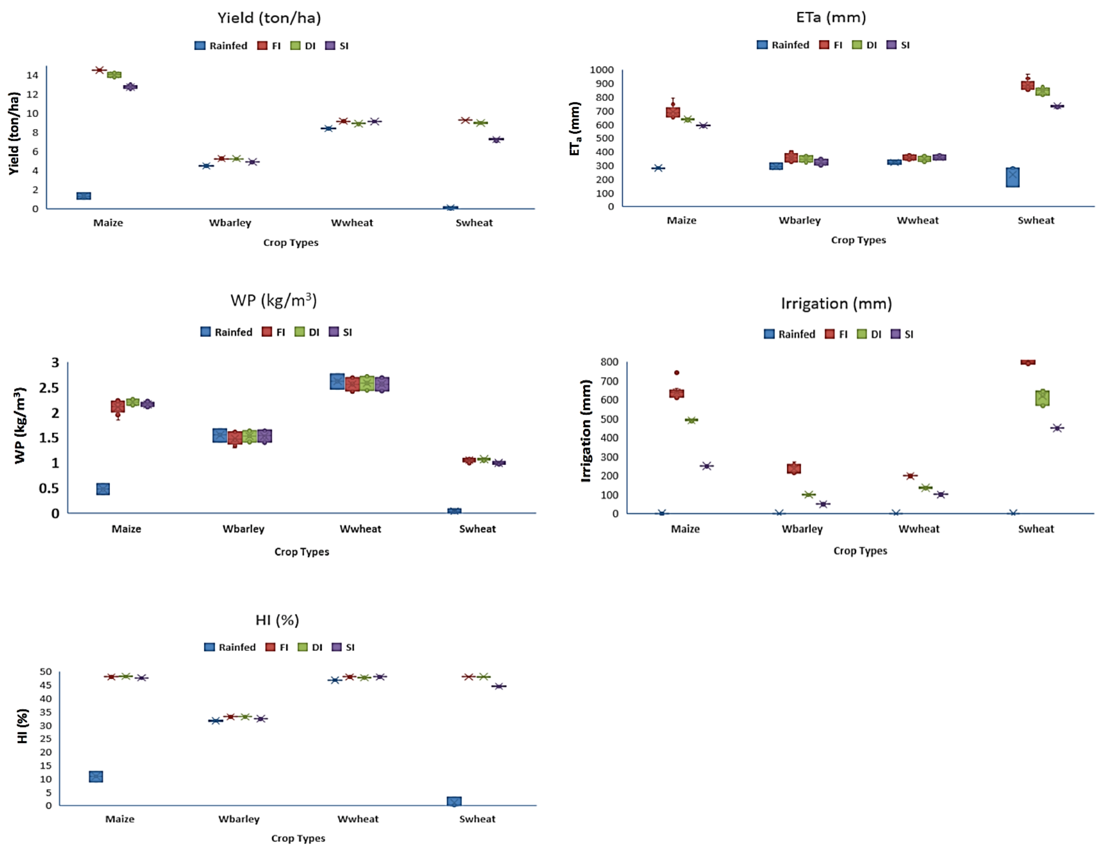

Figure 7. The yield (Y), actual evapotranspiration (ET_a), water productivity (WP), irrigation (I), and harvest index (HI) comparisons for selected crops within different irrigation strategies.

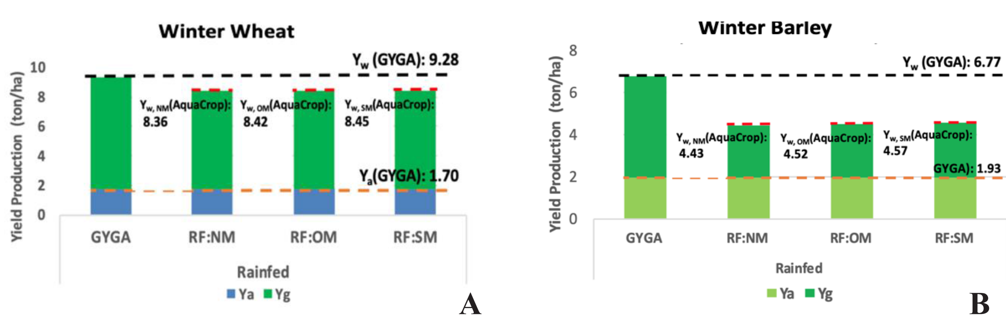

Figure 8. Changes in yield gap (Y_g) compared to GYGA within the comparison of GYGA for rainfed winter wheat (A) and winter barley (B) with different mulching trials.

Irrigated yield gap comparisons.

Figure 9 shows the yield gap changes according to different simulated management strategies for irrigated agriculture. Y_a was taken from GYGA regarding the reference column given as GYGA, and the remaining parts using the simulated yields which were selected as strategy-specific potential yields by different management applications. Potential yield (Y_p-GYGA) was taken from GYGA portal. It was found that more significant yield deviations found in the irrigated maize and irrigated spring wheat which are currently better options compared to rainfed agriculture (including only the yield quantity, economical and quality perspectives were not studied). Due to the higher proportion of the irrigated maize application, with approximately 15% highest yield decrease was found with SI application. As it is defined in the GYGA protocol, both GYGA (WOFOST model) and the findings of the current study (AquaCrop) describe the potential yields. However, Y_gs were smaller within the deficit irrigation strategies because of their less potential yield productions compared to full irrigation. Besides, large differences between AquaCrop and GYGA were because of the different model mechanisms and the data used during both studies. According to the exploitable yield gap results, SI strategy based simulations depicted a lower exploitable yield than modern maize cultivations; however, it provides savings for water resources through less water requirements in terms of crop productions.

Yield gap comparisons showed that FI strategy applied production had the highest Y_g because of its more massive potential (Y_p) compared to deficit water conditions (DI, SI, respectively, from larger to smaller yield production). As it was expected, Y_g decreases when the irrigation strategy changes from FI to SI. If yield production decreases for farmers and industrial producers like approximately 10%, the efficient use of water resources within the tendency towards less water demanded irrigation strategies (i.e. DI or SI) is inevitable. Irrigated maize yield production was in the range of 12.5-14.6 ton/ha, whereas it showed an insufficient amount of yield production under rainfed conditions with less than 0.6 ton/ha. An impressive result from this research was that yield production in irrigated maize production was almost three times higher than winter barley, and almost 0.5 times more than irrigated wheat. Hence, it is possible to reach more yield productions within maize cultivation. The critical point is that maize has less crop growth time length with five months (spring period) than other plantings with seven months (winter period). Thus, it makes possible to get benefit from those lands more, and food security issue might be undertaken more from this perspective to manage potential management opportunities.

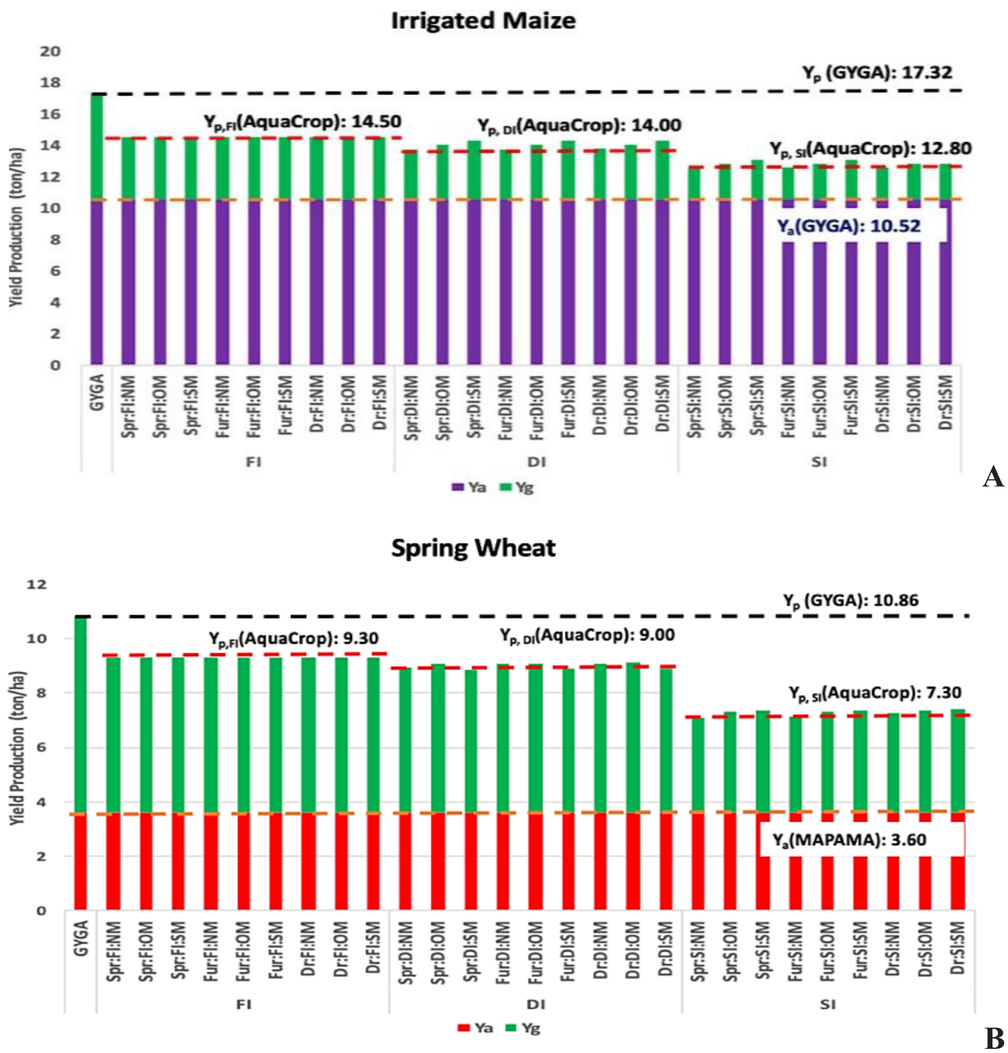

Figure 9. The yield gap changes within the comparison with GYGA (WOrld FOod STudies [WOFOST] model) for irrigated maize (A) and irrigated spring wheat (B).

To sum up, some strategies were selected considering the results from model. It was found that yield gap increases when the irrigation water uses increases because potential yields increases concurrently. Besides, no mulching had the highest consumption due to its least fruitful impact on water resources use efficiency. Furthermore, water is not the most limiting factor especially about water-limited yield formations for having fewer yield gaps also other factors like improvements on management practices would bring significant declines on yield gaps (Van Ittersum et al., 2013).

Figure 10 shows the yield gap comparisons from different studies for irrigated maize (GYGA: both Y_p and Y_a are from GYGA; for the simulated (AquaCrop): Y_p from trials within different management combinations, and Y_a from GYGA; for MAPAMA and Aldaya&Llamas (2008), Y_p from simulation runs and Y_a from sources). It can be seen that yield gap decreases from GYGA (data range for subsequent five years), MAPAMA, Simulated (AquaCrop), Aldaya and Llamas (2008), respectively. Some part of the simulated application has an overlap with GYGA (WOFOST model) estimations. Due to the larger temporal scale analysis in GYGA with 5-year, different environmental conditions (i.e. rainfall trends) show larger difference among period. In this study, year-specific actual yields were used because of the temporal scale of the research which is only for 2009. For larger temporal scale studies, using an average of 5-year actual yield was suggested by GYGA.

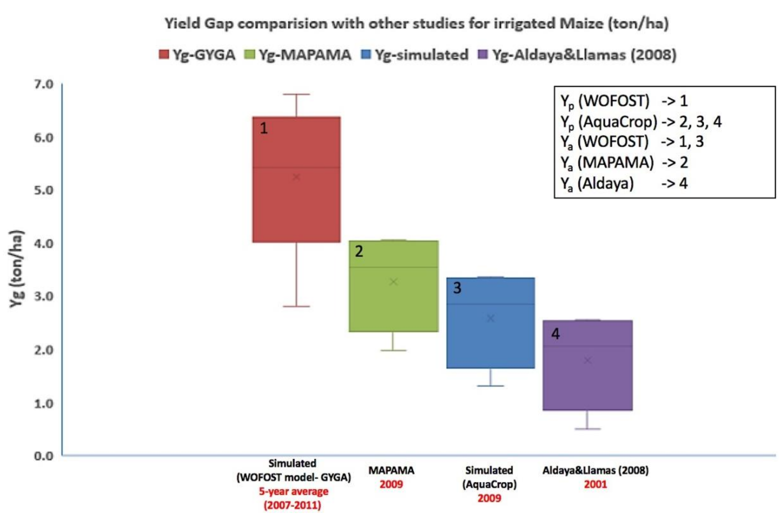

Figure 10. The yield gap comparison with other studies for irrigated maize.

Water productivity gap (WP_g).

The estimated water productivity gap (WP_g) is only available for winter wheat on GYGA. Therefore, WP_g only for winter wheat was given for various mulching practices (Figure 11). There was an increase in WP_g from no mulching to the organic and synthetic mulching due to the higher WP_p values from organic and synthetic applications by the AquaCrop model, respectively. WP_a was kept as constant from GYGA and comparisons were done with WP_p value from GYGA and possible WP values from the simulated ones. In the GYGA, only the WP_g analysis was executed for wheat production; therefore, for other crops, another reference was used (Aldaya and Llamas, 2008) from a different reference year.

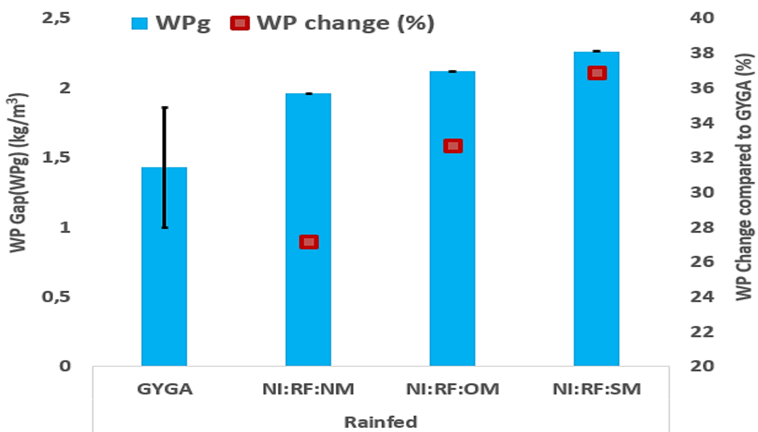

Figure 11. Strategy-specific water productivity gap (WP_g) changes over different rainfed mulching practices.

Figure 12 illustrates WP_g comparisons of the rainfed and irrigated crops. Winter wheat reachable strategy specific WP_p was calculated higher than winter barley and so the strategy specific WP_gs were higher. On the other hand, maize depicted higher WP_g for irrigated agriculture.

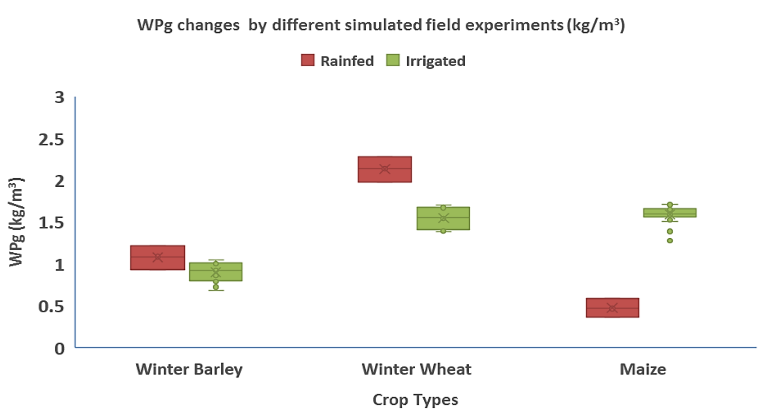

Figure 12. Water Productivity Gap (WP_g) comparisons of the rainfed and irrigated crops: winter barley, winter wheat, and maize.

Impacts of the Different Management Practices on Water Balance of the Water Resources Systems

Impacts of the different applications on the water resources systems were shown within individual strategies and then converted to the sub-catchment scale.

The reference data for this calculation was taken from GuaSEEAW (2015), which provides sub-catchment level hydrological and sectoral data among the Guadiana river basin. Besides, Portugal part was also involved in the dataset. AQUATOOL water basin management model was used by GuaSEEAW (2015), and those data were used in this research as a tool to calculate the effects of different strategies within the current case in selected sub-catchments. Stream flows were considered in this type of analysis. The order of the strategies showed a certain decrease trend from the worst-case scenario to the most efficient scenario with less water demand for agricultural production. Besides, it is necessary to consider yield changes to understand the extent of yield decreases compared to water resource efficiency increases. For irrigated maize application within strategy 9 (SI: SM), the yield decrease compared to the reference case was 12%, on the other hand, it was nearly 2% within strategy 6 (DI: SM). Integration between stakeholders is crucial to come to an agreement within a common ground. To illustrate, when it is necessary to consider both farmers' perspectives and environmentalist consideration, it might be a good scenario with less yield change and remarkable decreases in irrigation water requirements. In addition to this, mulching practices depicted an increase in yield productions for each crop.

There are two ways to interpret the analysis of water withdrawal and water consumption in water resources systems. Water withdrawal scope does also include the water conveyance losses, for instance, irrigation requirements are needed to be ensured from a water resource, and irrigation demand requires a certain amount of water conveyance line to the field from water source. However, water losses are inevitable from those water abstractions. This study was mainly focused on consumptive water uses (CWUs) as blue ET_a. Besides, CWUs were defined as ET_a which is the consumed water by a certain crop and can be derived from the AquaCrop model. The results from green CWU and blue CWU indicated that green CWU was larger than blue CWU. Due to the abstractions from surface water bodies which is the dominant water resource type in the middle Guadiana, blue CWU was selected as a critical parameter for the next steps.

Regarding sectoral water uses, it is not easy to estimate water consumptions for agricultural, domestic, and manufacturing sectors due to the substantial variabilities between soil-crop-water interactions, and water cycle complexities, human activities and different production patterns of the manufacturers. However, water consumption calculation can be executed within the use of agricultural system models (i.e. AquaCrop). These types of models are capable of simulating agricultural water demands and outflows. For example, from the manufacturing sector, recycling ratio assumption is used to estimate water consumptions as a conversion factor from water withdrawal to water consumption for global sectoral water use models (Wada et al., 2011); however, it holds an assumption behind and having some uncertainties. Mainly, implications of studies especially in emerging countries are difficult to overcome because of data limitations and economic constraints.

In conclusion, nine strategies were selected after water balance analysis to be used in WSI analysis for comparing the strategies with a reference strategy. FI and NM application was selected as reference strategy (worst case scenario regarding water resources use efficiency), and following strategies were chosen as FI:OM (**Strategy-2**), FI:SM (**Strategy-3**), DI:NM (**Strategy-4**), DI:OM (**Strategy-5**), DI:SM (**Strategy-6**), SI:NM (**Strategy-7**), SI:OM (**Strategy-8**), SI:SM (**Strategy-9**).

Impacts of the Different Management Practices on Water Scarcity Index (WSI) of the Water Resources Systems

Figure 13 shows the difference between reference strategy (FI: NM) applications within different strategies. It can be seen that WSI changes appear to be more efficient in the way of water resources management within a different strategy. Case area-specific based WSI analysis showed significant improvements in water resources use efficiency in the upstream compared to WSI analysis including all sectors. The best management option was found as SI: SM for water resource use efficiency which illustrated that WSI compared to the reference strategy decreases five times within the case-specific calculation (Figure 13).

WSI analysis can be done by using different key drivers including water withdrawal, water consumption or population (per capita) (Kummu et al., 2016). To understand the impacts of different strategies on the WSI analysis, the worst-case scenario as the reference case (FI: NM) was chosen because of its largest water demand among other strategies. The scarcity/shortage situation was decreased significantly especially in the upstream part. Due to the differences between stream inflow to downstream part, impacts were not found significant as much as upstream part. The reason is that the nominator in the equation of the WSI analysis causes more sensivity than denominator because of tremendous amount of water availability than water uses. Therefore, case area-specific analysis was preferred for the comparison of water scarcity/shortage degrees.

When we change the management practices from FI: NM to SI: SM, there were nearly five times more improvements on the system that meaning of the least negative impacts on the water resources systems. From a farmer perspective, SI does not seem to be an optimal strategy because of the 12% less yield achievement. But the Strategy 6 (DI:SM) showed 2% yield decrease and 2 times more water resources efficiency compared to reference strategy. This finding was seen from the exploitable yield gap where SI based yield production was less than actual yield. On one hand, we look at different irrigation strategies as the most efficient ones for the water resources system were SI, DI, FI, respectively for irrigated agriculture, on the other hand, for mulching practices, the most efficient one was the synthetic one.

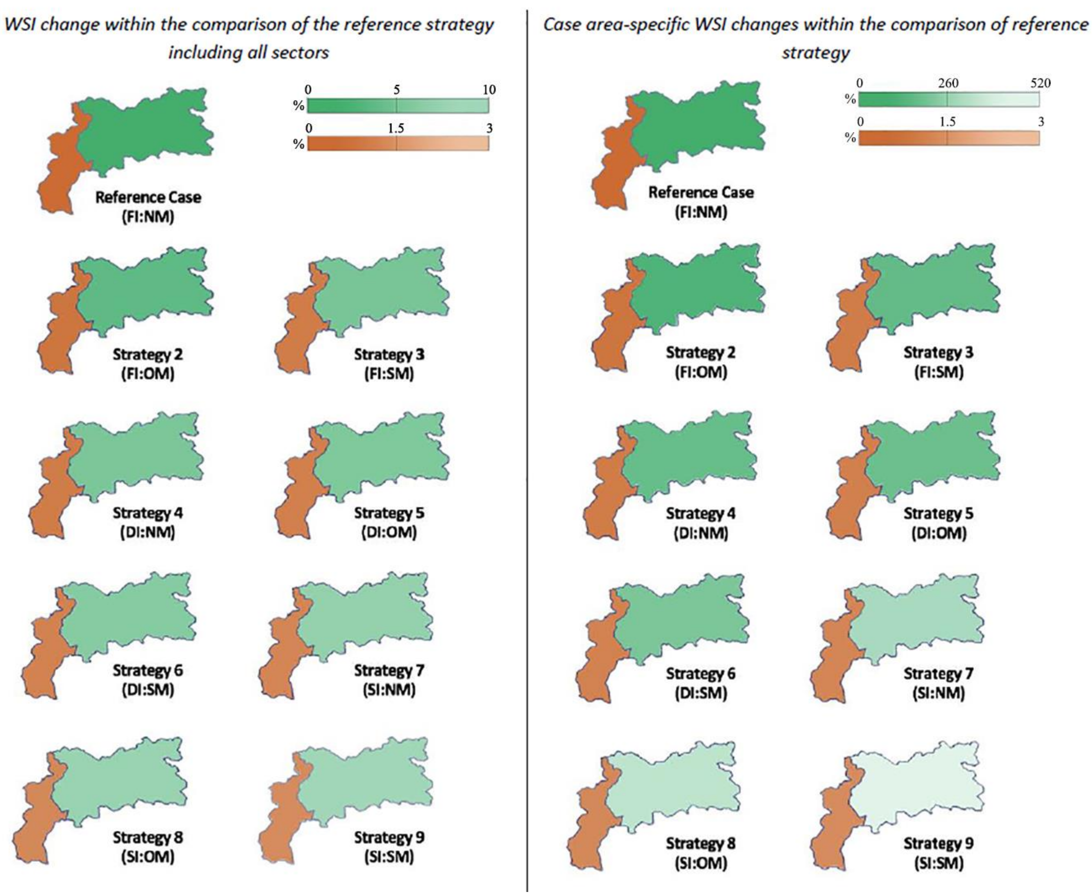

Figure 13. WSI variations of different strategies according to the reference strategy.

Conclusion

The analysis of different irrigation and management practices within the AquaCrop model indicated that model results are comparable with the particular area according to the comparisons with reported and simulated information from various sources. It was found that AquaCrop model is a sophisticated model to estimate ET_a, Y, and WP parameters. Mulching practices have positive impacts on the ET_a decreases. Besides, different irrigation strategies resulted in different yield responses, and yield productions decreased in deficit irrigation (DI) less than supplementary irrigation (SI), and full irrigation (FI) showed the highest yield within non-limited water conditions. This finding was an expected trend and decreases in water demands showed how the yields and water-related parameters are varied.

Impacts of different management practices on sub-catchment level water balance were calculated. The primary driver was selected as ET_a (CWU) change among the chosen strategies. WSI degrees were calculated by using the defined equations. The main issue regarding the water balance analysis was the scaling issues within the research and data availability concerning scales. Some assumptions were made; however, it was not possible to eradicate uncertainties entirely. To make robust and straightforward analysis, consumptive water uses from the simulated (AquaCrop) results were used to answer research questions with limited data. Water scarcity/shortage degree was estimated by water consumption amounts within including all sectors and only case-specific quantities. It is thus shown that case-specific estimations showed a clear appearance of the differences regarding the strategies compared to whole sectors. The delineation of the water scarcity/shortage degrees were more apparent in the upstream part than the downstream part because the model application was carried out for the upstream part which had the primary impacts apart from the downstream.

We address some recommendations for future studies as a next step of water resources system analysis:

1. It is possible to execute model runs for different seasons (i.e. normal, dry, wet, and future projections), in this way; results bring insights on larger temporal scales.

2. Considering grid-based soil types, other management practices (i.e. fertilizers, weed management, or salinity), and more climatic information from various meteorological stations may provide additional accuracy.

3. FI does not cause only the water resources deterioration, but also the salinity problems. Therefore, it is momentous to take into consideration quality matter too such as nonpoint source pollution (fertilizers, pesticedes etc.).

4. Although the AquaCrop model efficiency is high compared to modern observed studies, calibration and validation of the model could provide better projections for the future.

5. To calculate more detailed Y_g and WP_g estimations, data availability would bring more inputs to future studies. Such as actual yield and water productivity, potential yield and water productivity, and different management practices on the field could provide better investigations for benchmarking studies.

6. In addition to combined management practices based Y_g and WP_g analyses, strategy-specific definitions of these terms would ensure some insights on selecting the best management practices for a particular area.

7. In addition to a biophysical analysis, an integrated assessment can be done in further studies, for example, taking into consideration of economic analysis, life cycle assessment and social phenomena at the same time could be beneficial to improve integration of stakeholders and decision making process in the future.

Acknowledgements

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References

Aldaya, M. M., & Llamas, M. R. (2008). *Water Footprint Analysis for the Guadiana River Basin*. Madrid: Fundación Marcelino Botín.

Aldaya, M. M., & Llamas, M. R. (2008). *Water Footprint Analysis (Hydrologic and Economic) of the Guadiana River Basin*. Madrid: United Nations World Water Assessment Programme.

Camacho, S., Moura, D., Connor, S., Boski, T., & Gomes, A. (2014). Geochemical characteristics of sediments along the margins of an atlantic-mediterranean estuary (the Guadiana, Southeast Portugal): spatial and seasonal variations. *Journal of Integrated Coastal Zone Management, 14(1),* 129-148. doi 10.5894/rgci452.

Chukalla, A. D., Krol, M. S., & Hoekstra, A. Y. (2015). Green and blue water footprint reduction in irrigated agriculture: effect of irrigation techniques, irrigation strategies and mulching. *Hydrol. Earth Syst. Sci.*, *19*, 4877–4891.

Çakmak, B., Yıldırım, M., & Aküzüm. T. (2008). Türkiye'de Tarımsal Sulama Yönetimi, Sorunlar ve Çözüm Önerileri. 2. Su Politikaları Kongresi . Ankara: TMMOB.

Döll, P., & Siebert, S. (2002). Global Modelling of Irrigation Water Requirements. *Water Resources Research, 38 (4),* 8-1-8-10.

Ewans, R. G., & Sadler, E. J. (2008). Methods and technologies to improve efficiency of water use. *Water Resources Research,* 44, W00E04. doi:10.1029/2007WR006200.

Falkenmark, M. (1989). The massive water scarcity now threatening Africa: why isn't it being addressed? *Ambio*, 112-118.

FAO. (2017). Irrigated Crop Calendars. Retrieved from http://www.fao.org/nr/water/aquastat/water_use_agr/index2.stm.

GuaSEEAW. (2015). New developments in Water Accounts Implementation in Guadina River Basin. Technical Report. Retrived from http://ec.europa.eu/environment/water/blueprint/pdf/D%201.3.%20Final%20Technical%20Report%20v1.3.pdf

GYGA. (2017). Global yield gap atlas. Retrieved from http://www.yieldgap.org

Hamdan, I., Oweis, T., & Hamdallah, G. (Eds.). (2006). Proceedings of the Expert Consultation Meeting: *AARINENA Water Use Efficiency Network*, Syria: Aleppo.

Hoekstra, A. Y., Chapagain, A. K., Aldaya, M. M., & Mekonnen, M. M. (2011). *The Water Footprint Assessment Manual*. Retrieved from http://waterfootprint.org/media/downloads/TheWaterFootprintAssessmentManual_2.pdf

Hoekstra, A.Y., Mekonnen, M. M., Chapagain, A.K., Mathews, R.E., & Richter, B. D. (2012). Global Monthly Water Scarcity: Blue Water Footprints versus Blue Water Availability. *PLoS ONE*, *7*(2), e32688. doi:10.1371/journal.pone.0032688

Kang, Y., Khan, S., Ma, X. (2009). Climate change impacts on crop yield, crop water productivity and food security-A review. *Progress in Natural Science*, *19(2)*, 1665-1674. https://doi.org/10.1016/j.pnsc.2009.08.001

Klein Tank, A.M.G., Wijngaard, J. B., Konnen, G. P., Bohm, R., Demaree, G., Gocheva, A., … Petrovic, P. (2002). Daily dataset of 20th century surface air temperature and precipitation series for the European Climate Assessment. *International Journal of Climatology, 22*, 1441-1453. Available from http://www.ecad.eu

Kummu, M., Guillaume, J.H.A., de Moel,H., Eisner, S., Flörke, M., Porkka, M.,… Ward, P.J. (2016). The world's road to water scarcity: shortage and stress in the 20th century and pathways towards sustainability. *Scientific Reports*, Vol 6:38495.doi: 10.1038/srep38495

Spanish Ministry of Agriculture, Fisheries and Food (MAPAMA). (2010). *Agro-alimentary Statistics Yearbook*.. Retrived from http://www.mapama.gob.es/es/estadistica/temas/estadisticas-agrarias/agricultura/superficies-producciones-anuales-cultivos/

OECD. (2015). *Crop yield*. Retrieved from https://data.oecd.org/agroutput/crop-yields.htm

Patel, Y.S., Joshi, P.M., & Joshi, V.P. (2017). Analysis of Water Resources use for Agricultural Purpose in Gujarat. *Global Research and Development Journal for Engineering*. Vol 2:14-18. ISSN: 2455-5703.

Pereira, L.S., Cordery, I., & Iacovides, I. (2012). Improved indicators of water use performance and productivity for sustainable water conservation and saving. *Agricultural Water Management. 108*, 39-51. doi: 10.1016/j.agwat.2011.08.022

SAIH. (2017). *The Sytem of Hydrological Information, SAIH del Guadiana*. Retrived from http://www.saihguadiana.com/

Smakhtin, V. Y., Revenga, C., & Döll, P. (2004). Comprehensive Assessment Research Report 2: *Taking into account environmental water requirements in global-scale water resources assessments*.. Srilanka: Colombo..

Steduto, P., Hsiao, T. C., Raes, D., & Fereres, E. (2012). Crop yield response to water, Food and Agriculture Organization of the United Nations. Italy: Rome.

Uhlenbrook, S., & Savenije, H. H. G. (2017).*Hydrology of catchments, river basins, deltas: Part one-Catchment and water balance*. TU DELFT, DELFT. Retrieved from https://collegerama.tudelft.nl/Mediasite/Play/d5535e795d9d4f0b90d4da81cd1f1e9c1d?playFrom=17028&autoStart=true.

United Nations (UN). (2017). *World population prospects*. Retrieved from https://esa.un.org/unpd/wpp/Publications/Files/WPP2017_KeyFindings.pdf

Van Ittersum, M.K.V., Cassman, K.G., Grassini, P., Wolf, J., Tittonell, P., & Hochman, Z. (2013). Yield gap analysis with local to global relevance–a review. *Field Crops Research*, *143*, 4–17.

Wada, Y., Van Beek, L.P.H., & Bierkens, M.F. (2011). Modelling global water stress of the recent past: on the relative importance of trends in water demand and climate variability. *Hydrology and Earth System Sciences, 15*(12), 3785-3805.

Wart, J. W., Bussel, L. G. J. V., Wolf, J., Licker, L., Grassini, P., Nelson, A., … Cassman, K. G. (2013). Use of agro-climatic zones to upscale simulated crop yield potential. *Field Crops Research*, *144*, 44-55.

Wnuk, A., Górny, A. G., Bocianowski, J., & Kozak, M. (2013). Visualizing harvest index in crops. *Communications in Biometry and Crop Science, 8*(2), 48-59.

Zhang, L., Walker, G.R., & Dawes, W.R. (2002). Water balance modelling: concepts and applications. In McVicar, T.R., Li Rui, Walker, J., Fitz- patrick, R.W. & Changming, L. (Eds). *Regional Water and Soil Assessment for Managing Sustainable Agriculture in China and Australia* (No. 84, pp. 31–47).,Canberra.

Appendix

The Comparison of Yield & ET_a and WP & ET_a

First, yield (Y) and actual evapotranspiration (ET_a) results were compared with the all model results of the trials on a scatter plot diagram in Figure A1. Next, water productivity (WP) and actual evapotranspiration (ET_a) results for four crops are shown on a scatter plot diagram in Figure A2. The ET_a results lower than 300 mm are related to the rainfed cultivation for spring wheat, and maize. The ET_a results higher than 500 mm are related to irrigated agriculture for the same crops. The relationship between Y and ET_a shows a production curve that is increasing and level off with a high correlation (0.99 R^2) value for spring wheat and maize. On the other hand, there is no remarkable difference between rainfed and irrigated crop cultivation for winter wheat and winter barley. Furthermore, the relationship between Y&ET_a and WP&Y was found weaker in winter wheat and winter barley production compared to maize and spring wheat production. Figure A2 illustrates that the WP decreases after a moment reached on ET_a. It is mainly the reason for the full irrigation and no mulching strategies within different irrigation technologies. It can be seen that leading drivers of the decrease in the WP is caused by mulching and irrigation strategies.

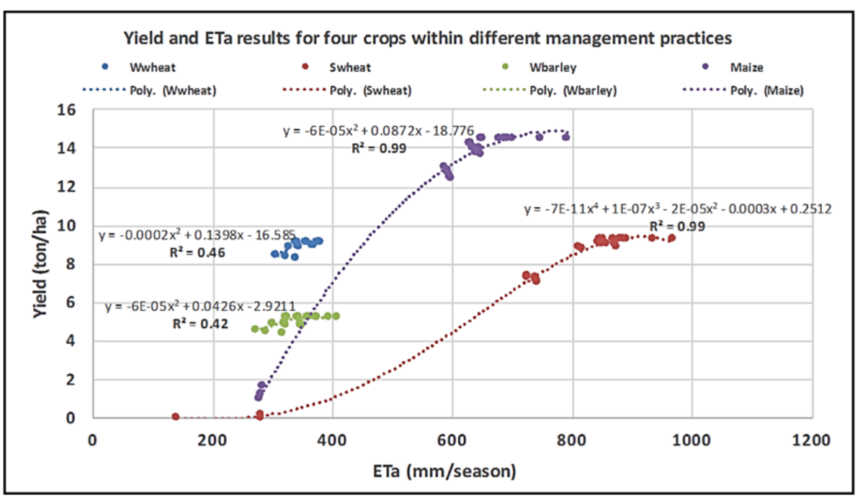

Figure A1. The comparisons of the yield (Y) and actual evapotranspiration (ET_a) for four crops (winter wheat, spring wheat, winter barley, maize) within all different management practices.

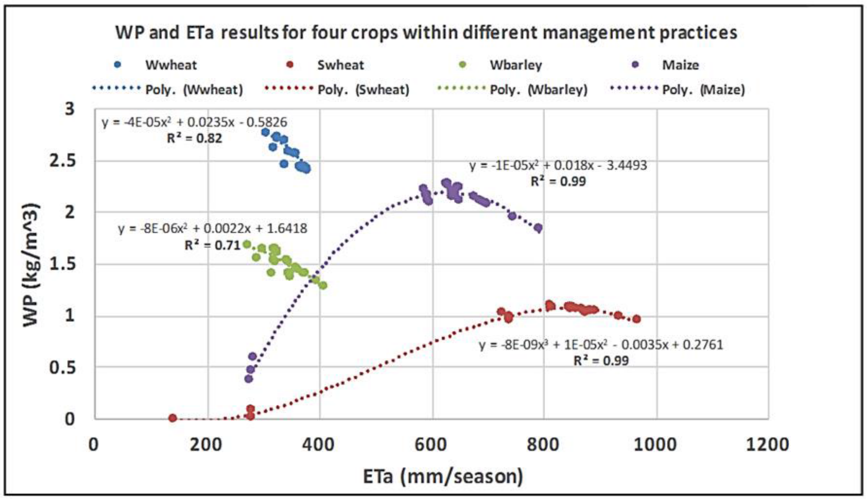

Figure A2. The comparisons of the water productivity (WP) and actual evapotranspiration (ET_a) for four crops (winter wheat, spring wheat, winter barley, maize) within all different management practices.

Extended Turkish Abstract
(Genişletilmiş Türkçe Özet)

Sulama ve Arazi Yönetimi Uygulamalarının Su Kaynakları Sistemlerindeki Etkileri

Dünya nüfusunun 2050'de 2010 yılına göre yaklaşık olarak %70 oranında artacağı tahmin edilmektedir ve bu nedenle gıda talebinin giderek artacağı ve gelecek nesillerin beslenmesinin daha kritik olacağı öngörülmektedir. Bu çalışma, farklı sulama ve arazi yönetimi uygulamalarının su kaynakları sistemleri üzerinde nasıl bir etkisi olduğunu araştırmaktadır. Gerçek sistem değerlendirmelerinin karmaşıklığı nedeniyle, modeller karmaşık biyofiziksel sistemlerin analizini ve maliyetli işleri kolaylaştırmaktadır. Bu çalışmada, AquaCrop modeli (FAO), farklı sulama ve arazi uygulamalarının su kaynakları sistemlerindeki etkileri öngörmede analitik bir araç olarak kullanılmıştır. AquaCrop modeli, mahsul fenolojisini ve toprak-su-verim ilişkilerinde çevresel değişkenliklere davranışsal tepkileri simüle eder ve çiftçilere veya karar vericilere yardımcı olmaya çalışır. Daha güçlü tahminler yapabilmek ve sistemi daha iyi anlamak için Entegre Değerlendirme (Integrated Assessment - IA) yapılmasının gerekliliği kaçınılmazdır. Bu araştırmanın amacı, tarımsal sektördeki mahsül verimi ve su verimliliğine ilişkin toprak ve bitki etkileşimleri ile belirli mahsuller için seçilen alt su havzasındaki farklı yönetim stratejilerinin etkilerinin araştırılmasıdır. Bu amaç doğrultusunda aşağıdaki araştırma sorularına cevap aranmıştır:

- Gerçek evapotranspirasyon (ET_a), mahsül verimi (Y) ve su verimliliği (WP) nedir?
- Model sonuçları, diğer çalışmalarla karşılaştırıldığında mahsül verim açığı (Y_g) ve su verimliliği açığı (WP_g) analizleri bakımından nasıl sonuç vermektedir?
- Yukarı havza - aşağı havza etkileşimi ile ilgili yönetim uygulamalarından su sistemi (su dengesi) ve su kıtlığı / yokluğu dereceleri (WSI) nasıl etkilenecektir?

Araştırma adımları, bir su kaynakları sistemlerinin analizi için ayarlanmıştır. İlk olarak, AquaCrop modeli, seçilen tarım ürünlerinin (kışlık buğday, baharlık buğday, kışlık arpa ve darı) ve belirli yerlerdeki (orta Guadiana alt havzası, İspanya; Guadiana havzası Portekiz Bölümü, Portekiz) farklı yönetim uygulamalarının sonuçlarını tahmin etmek için uygulanmıştır. Modeldeki ilk adımdaki ana faktörler, tarım sektöründeki farklı yönetim uygulamalarına gerçek evapotranspirasyon (ET_a), mahsül verimi (Y) ve su verimliliği (WP)'nin tepkileridir. Bu çalışma, üç farklı yönetim stratejisinin (sulama teknolojileri (yağmurlama, karık, damla); sulama stratejileri (tam sulama, kısıntılı sulama, tamamlayıcı sulama ve yağmura dayalı sulama), malçlama uygulamaları (malçlama yapılmaması, organik malçlama, sentetik malçlama) etkilerini incelemektedir. Toplamda, farklı yönetim stratejilerinde seçilen yıl için 120 model simülasyonu gerçekleştirilmiştir. İkinci olarak, mahsül verim açığı (Y_g) ve su verimlilik açığı (WP_g) analizleri, mukayeseli değerlendirme çalışmaları bakımından önem taşımaktadır. Mahsül verimi açığı (Y_g), potansiyel olarak ulaşılabilir mahsül verimi (Y_p) ve gerçek mahsül verimi (Y_a) arasındaki farkı ifade etmektedir. Potansiyel olarak ulaşılabilir mahsul verimi (Y_p), uygun iklim koşulları, sınırlandırılmamış nütrient sağlanması ve biyotik streslerin iyi kontrol edildiği zaman elde edilen verimdir. Üçüncü olarak, mavi ve yeşil su tüketim kullanımları her bir strateji için tarla ölçeğinden alt havza ölçeğine, seçilen alandaki ekim alanları göz önünde bulundurularak hesaplanmıştır. Su çekimlerini hesaplarken, su dağıtımı için su kayıpları %30 olarak tahmin edilmiştir (Aldaya ve Llamas, 2008). Su kıtlığı/yokluğu derecelerinin (WSI) hesaplamalarında kullanılacak olması ve su iletimi – su dağıtımındaki kayıpların belirsizliği nedenleriyle su çekimleri yerine su tüketimlerine su dengesi hesaplamalarında yoğunlaşılmıştır. Son olarak, su kıtlığı dereceleri, farklı yönetim stratejileri için karşılaştırılmıştır. Su kıtlığı dereceleri hesaplanırken su tüketimleri ve tüketimlerindeki değişimlerinin mevcut su kaynaklarının oranı ile hesaplanmış olup, bu çalışma belirli bitki deseni çeşitleri için yapıldığından ötürü, tüm sektörleri içeren WSI analizine ek olarak daha hassas sonuçları göstermesi adına çalışma özelinde hesaplamalar da gerçekleştirilmiştir.

Çalışma neticesinde elde edilen bulgulara değinilecek olursa, ilk olarak, AquaCrop modelinin ET_a, Y ve WP parametrelerini tahmin etmek için sofistike bir model olduğu bulunmuştur. Farklı sulama stratejileri farklı verim yanıtları göstermiş olup, kısıntılı sulamada (DI) tamamlayıcı sulamadan (SI) daha az mahsül verimi düşüşü gözlemlenmiştir. Tam sulama sınırlandırılmamış su koşullarında en yüksek mahsül verimini göstermiştir. Ayrıca, malçlama uygulamalarının ET_a azaltımı üzerinde olumlu etkileri gözlemlenmiştir. İkincisi, sulama ve tarla yönetimi uygulamaları Y_g'yi belirli stratejiler dâhilinde kapatmayı mümkün kılmaktadır. Bununla birlikte, bu uygulamaların olumsuz sonuçlarını ortadan kaldırmak için çevresel kaygılar dikkate alınmalıdır. Bu analizin ana bulgularından biri de tam sulama yapılan mahsullerin, kısıntılı sulama gerçekleştirilen üretime kıyasla daha yüksek potansiyel mahsül verimine ulaşmasına rağmen, mühendislik tipi modellerle su tasarrufu sağlamakla beraber mahsül eldesi düşüşlerinin minimizasyonunu hesaplamakta mümkün olmaktadır. Üçüncü olarak ise; kuvvetli ve anlaşılır bir analiz yapmak için, farklı sulama ve yönetimsel uygulamaları kullanılarak, AquaCrop modelinden elde edilen su tüketimi sonuçları ile su bütçesinin analizi ve değişikliklerin gözlemlenmesi gerçekleştirilmiştir. Son olarak, çalışılan sistemin sınırları (seçilen tarımsal ürünleri) göz önünde bulundurularak su kıtlığı derecelerinin analizi ile ilgili olarak ve tüm sektörleri içeren su kıtlığı derecelerinin farklı sulama ve yönetimsel uygulamalar bazında karşılaştırılmaları seçilen su kaynakları sistemleri için gerçekleştirilmiştir. Tam sulama (FI): Malçlama olmadan (NM) senaryosu su kaynakları üzerinde en kötü etkileri gösterirken, tamamlayıcı sulama (SI): sentetik malçlama (SM) uygulamaları çalışılan stratejiler içerisinde yüzeysel su kaynağına en düşük olumsuz etkiyi göstermiştir. Bunlara ek olarak; kısıntılı sulama (DI): sentetik malçlama (SM) stratejisinde daha az mahsül üretim kaybı ile önemli miktarda su tasarrufu gözlemlenmiştir.

Research Article

Investigating the Effect of Climate Change on Stormwater Networks: Capital Ankara Case

İklim Değişikliğinin Yağmursuyu Şebekeleri Üzerine Etkisinin Araştırılması: Başkent Ankara Örneği

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Abstract

In this study, we applied the analysis results of past and future variations of extreme precipitation and land use/cover changes in Ankara province for a newly built stormwater network of a pilot study area in Etimesgut, Ankara. We investigated the performances of the system under current and changing extreme rainfall conditions and different approaches such as stationary and nonstationary extreme precipitation assumptions. The system operated in a satisfactory state and it can be said that according to climate change projections for the extreme rainfall, the maximum volume that the system face will not exceed baseline design criteria throughout the projection period. Combination of changing climatic and land use/cover conditions also reveal a satisfactory performance for the baseline design which used 15 minutes storm duration and 2 years return period rainfall intensity and a runoff coefficient of 0,8 as design input. On the other hand, the system may fail under the loads derived separately or together with longer storm duration (such as 30 minutes or more) or higher return periods (such as 5 years and more) that is computed from stationary and nonstationary observed data analysis which is a preferred design input for such a critical facility and area.
Keywords: *Climate change, stormwater, extreme rainfall*

Öz

Bu çalışmada Ankara İlinin geçmiş dönem ve gelecekteki aşırı yağış ve arazi kullanımı/örtüsü değişim analiz sonuçları, Ankara İli Etimesgut ilçesinde yeni inşa edilen bir pilot çalışma alanının yağmur suyu şebekesi için uygulanmıştır. Sistemin performansı, mevcut ve değişen koşullar ile durağan ve durağan olmayan aşırı yağış varsayımı gibi farklı yaklaşımlar altında araştırılmıştır. Sistemin iklim değişikliği projeksiyonları altında yeterli ve beklenen servisi sağlayabileceği ve aşırı yağış için yapılan iklim değişikliği projeksiyonlarına göre sistemin temel tasarımda öngörülen maksimum kapasiteyi projeksiyon süresi boyunca aşmayacağı söylenebilir. Değişen iklim ve arazi kullanım/örtü koşulları birlikte değerlendirildiğinde sistem, 15 dakikalık yağış süresi ve 2 yıl tekerrür periyodu ile akış katsayısı olarak 0,8 kullanılan temel tasarım için tatmin edici bir performans ortaya koymaktadır. Öte yandan sistem gözlem verisi ile yapılan durağan ve durağan olmayan analizlerden elde edilen daha uzun yağış süreleri (30 dakika veya daha fazla gibi) veya daha yüksek tekerrür süreleri (5 yıl ve daha fazlası gibi) ile ayrı ayrı veya birlikte hesaplanan yükler altında beklenen servisi sağlayamayabilir ki bu süreler kritik bir altyapı tesisi ve alan için tercih edilen bir tasarım girdisidir.

Anahtar kelimeler: *İklim değişikliği, yağmur suyu, a*şırı *yağış*

Introduction

Climate change and its potential impacts gained importance due to projected changes in temperature and precipitation which can significantly affect the hydrological cycle, land use, extremes and the related infrastructure (Elshorbagy et al., 2018). The alterations in land cover and rainfall characteristics that cause urban flooding increase the need for impact assessments on design and management of stormwater networks (Hailegeorgis & Alfredsen, 2017). Stormwater networks can be sensitive to climate change, in particular to extreme rainfall events as they are one of the main variables for design. On the other hand, the design and expected performance of stormwater infrastructure become questionable with the changing climate (nonstationarity) because the conventional design (stationary assumption) may not consider the changing climatic conditions (Rosenberg et al., 2010).

Impacts studies on urban storm water drainage systems gain high attention at locations where these systems are vulnerable. Osman (2014) analyzed future rainfall characteristics and modelled the output to explore the impacts of climate change on the urban drainage system in the Northwest of England during the 21st Century. The results implied that potential changes in rainfall intensity in the future are expected to alter the performance and serviceability of the system, causing more challenges such as surface flooding and increase in surcharge level in sewers. Bahadur et al. (2016) reports that stormwater management systems will be overwhelmed by the rising intensity of rainfall, and extreme events will damage the infrastructure systems in specific regions of Asia. Hence, urging for urban climate change resilience is considered.

Urbanized areas become more vulnerable to flood hazard under conditions of high precipitation intensity (Sun et al., 2011). During the last century the number of people living in urban areas has globally increased rapidly. At the beginning of the twentieth century, only 14% of the world population lived in urban areas, today 55% of the global population resides in urban areas (United Nations [UN], 2018). This increase in urban population is expected to continue until at least 2050 and reach 68% (UN, 2018). Since urbanization is one of the main consequences of urban population growth, it is expected that increase in the urban population lead to urbanization over that period. As a consequence of urbanization impervious surface area increases and this in turn brings significant effects on the hydrological cycle in the urban areas. Increased proportion of impervious surface results in shorter lag times between onset of precipitation and end up with higher runoff peaks and total volume of runoff (Shuster, Bonta, Thurston, Warnemuende, & Smith, 2005). The conversion of pervious (permeable) land to impervious (non-permeable) surfaces changes the hydrologic characteristics of the landscape by reducing infiltration into the soil and evapotranspiration from vegetation which results in a dramatic increase in the rate and volume of stormwater runoff (Guidelines for NYC, 2012).

The rising trend of rainfall intensities as a result of changing climate or variability is a challenge for infrastructure systems that use particular return levels and periods as design parameters (Zhou, 2014). Furthermore urbanization raises the impervious areas, changes land cover types resulting in increase in discharge, volume, and frequency of floods which together with climate change induced intensified rainfalls will have amplified effects on urban stormwater systems (Thakali et al., 2016). The adaptation process is not as fast as the changing environmental conditions and this also increases exposure to floods therefore, as a result vulnerability is increased (Kundzewicz, 2003; Trenberth, 1998). In addition to extreme rainfall events and urbanization effects, inadequate investment and maintenance of infrastructure further increases exposure to flooding (Simonovic et al., 2016). While older stormwater networks have not been designed to withstand extreme rainfall events, urbanization further increases impervious surfaces and exacerbates the effect of extreme rainfall events.

In the flooding observations of last 20 years, heavy rainfall and flash flooding caused significant damages to the properties and even loss of life in Ankara, the Capital of Turkey (Supplementary Document of Official Letter of Turkish State Meteorological Service [TSMS], 2017a). Therefore, a strong need has emerged to study extreme precipitation events to reveal potential frequency and intensity alteration under changing climate conditions, reveal the effect of land use/cover change and investigate the impact of these changes on the urban stormwater networks. The current and projected IDF analysis of Ankara with transient climate change effects was already investigated in Oruc et al. (2019). However, their impact on storm sewerage system has not been carried out so far not only in the capital of Turkey, Ankara but also any other province throughout the country. It'll be a pioneer to document the response of a storm water infrastructure to changing climate condition.

In this study the results of extreme precipitation and land use/cover change analyses from Ankara province are applied for a newly built stormwater network of a pilot study area in Etimesgut, Ankara. The effect of changing climatic and land use/cover conditions are studied in pilot study area to incorporate climate change into urban stormwater network design by using stationary and non-stationary GEV models for extreme rainfall analysis together with land use/cover change. Performance of the system is investigated by the analysis in which stationary (St) and nonstationary (Nst) maximum storm intensities are converted to peak discharges using rational formula and they are conveyed from the current pipeline system under current and future conditions. As a result, the resulting fullness capacity of the system with and without revision is compared with the baseline (current). Finally, the basic economic assessment is also performed after suggested revisions to the system.

Methodology

Data and Study Area

The annual maximum precipitation data of 5, 10, 15, 30 minutes and 1, 2, 3, 6 hours duration is used for the observation period (1950-2015) while the data of 10 min, 15 min, 1 hour and 6 hour durations is used for the projection period in extreme value analysis. The future data consists of daily estimates from three global climate models (GCM) namely HadGEM2-ES, MPI-ESM-MR, and GFDL-ESM2M based on RCP 4.5 and RCP 8.5 emission scenarios. A fine-scaled regional climate model (RCM) coupled to these GCMs provides the daily precipitation of 2015-2099 period. As it is documented in Oruc et al. (2019), the maximum precipitation data for sub daily scales (10 min, 15 min, 1 hour and 6 hour) were obtained by applying a disaggregation method to projected daily precipitation values. Table 1 the location of meteorological stations and nearest RCM grids to these stations together with their altitudes are given.

Table 1

Projection Data Stations & Grids (Supplementary Document of Official Letter of TSMS, 2017b)

No	Station	Grid	Station			Grid		
			Latitude	Longitude	Altitude mt	Latitude	Longitude	Altitude mt
17129	Etimesgut Airport	2733	39,9558	32,6854	806	39,9661	32,6608	1028
17131	Ankara Güvercinlik Airport	2733	39,9343	32,7387	820	39,9661	32,6608	1028

Stationary and nonstationary rainfall return levels (in mm) for return periods 2, 5, 10, 25, 50, 100, 200 years are derived for observed and projected data for extreme rainfall time series of the sub-hourly and hourly annual maximum data.

Extreme precipitation analysis results are transformed and applied to test the runoff and hydraulic performance of an actual stormwater network of a Railway Maintenance Complex (Figure 11) located in Etimesgut, Ankara Province (Turkish State Railways, 2018).

Figure 1. Railway Maintenance Complex in Etimesgut,Ankara.

Ankara is located in the northwest of Central Anatolia. The city is like a pot surrounded by four mountains of Anatolia Plateau with an altitude of 850-1000 meters. A population of 5.3 million people (Turkish Statistical Institute [TURKSTAT], 2016) lives in the capital Ankara and 88% of the population lives in the city center (Governorate of Ankara, 2018). The growth of Ankara displayed a typical example of modernization efforts at the beginning and later (the second half of this century) it showed the uncontrollable expansion and transformation of the city with expanding squatter areas due to heavy migration. Then, the city shaped by urban regeneration projects in the last decades (Batuman, 2013).

The pilot study area is mostly flat and has an average altitude of 796 meters above mean sea level. The rail service maintenance plant has border on the northern side with the Ankara River, on the southern side with the existing railway line, on the eastern side with the Sugar Factory, and on the western side with the E89 road (Figure 2).

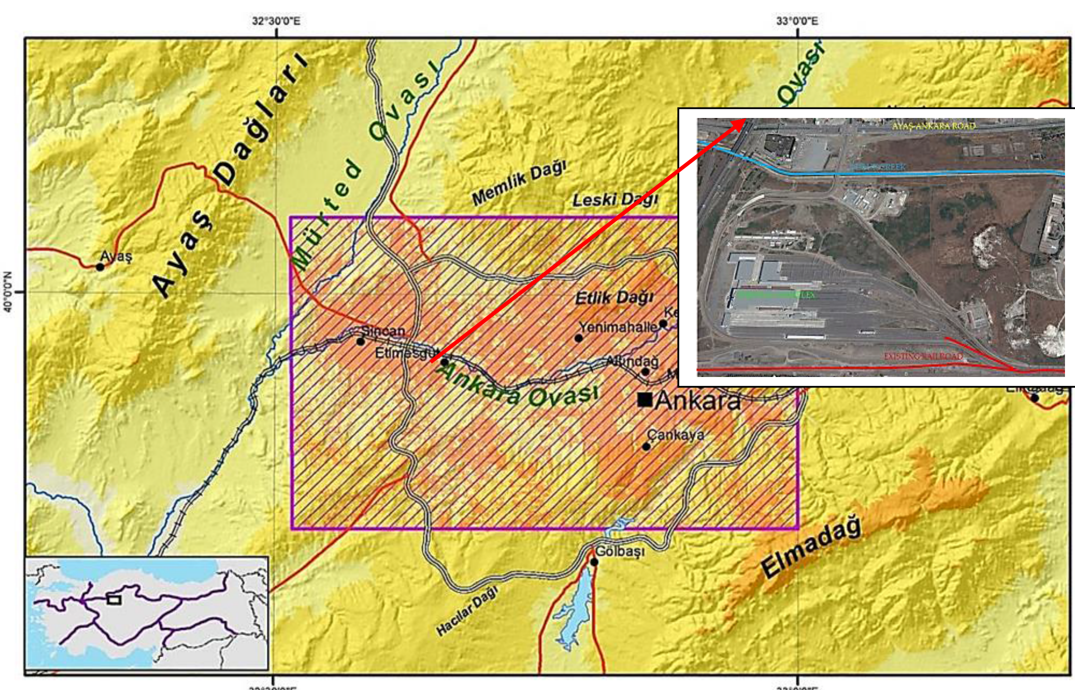

Figure 2. Location of Ankara (Yilmaz, 2013).

The new rail service maintenance plant is located approximately 20 km west of the center of Ankara. The stormwater network of this area consists of drainage channels and drainage manholes. The drainage manholes are connected by circular pipes; the drainage channels and the circular pipes discharge the flow in two main sewer pipes, in an existing sewer, or in the ditches. In addition to these solutions, some ditches have been included in the project to protect the rail service maintenance plant and the road. In these ditches is also conveyed part of the rain water of the rail service plant. The stormwater network covers a 342,000 m^2 basin area which is divided into 38 sub basins for the calculations. Single runoff coefficient of 0.8 is applied to the basin. Kutter formula is used to calculate the velocity of the discharge according to General Directorate of Ankara Water and Sewage Administration (AWSA) regulations. The stormwater network used for the calculations consists of 218 separate pipes in various length and size.

Intensity-duration-frequency analyses of maximum precipitation under stationary and nonstationary conditions were evaluated in Oruc et al. (2019) and therefore the readers are addressed to this article for further information about precipitation analysis that are used in this current study.

Land Use/Cover Analysis

Maps produced by the General Command of Mapping (GCoM) were analyzed as the most suitable data set by Özkil (2015). It has been identified that there are 4 different versions of maps (late 50s, early 80s, mid 90s and lately the year of 2013) covering the area of interest, available in the GCoM archives. Initially, the classification of land is done by extracting the image bands with the pixels grouping to each image segment. The land cover classification (e.g. urban area, green area, soil, road etc.) is performed based on the land use conditions. Polygons are created in advance during the digitizing process and then appropriate projection system is defined for polygons to calculate the geometry. Finally, urban and green areas (impervious and pervious surfaces) are calculated as the surface area per parcel.

In the pilot study area, there are several types of land cover, which are and their coverage areas are shown in Table 2. These cover types have different range of runoff coefficients (TDT, 2016; Burke & Burke, 2015) not only related with the material but also related with the slope of the cover (Table 2).

Table 2

Cover Types and Runoff Coefficient – Pilot Study Area

Type	Area m²	Runoff Coefficient
Building (Roof)	51.629	0.95
Car Park	18.500	0.95
Landscaped Area	4.650	0.50
Green Area	16.500	0.30
Asphalt-Sub-Ballast	149.000	0.95
Slab on Grade	49.000	0.95
Undeveloped	5.272	0.30
Potential Development Area	47.449	0.95
Composite Runoff Coefficient		0.90

A drainage area is composed of subareas with different runoff coefficients. The summation of the products of corresponding runoff coefficients and subareas is divided by the total area and then, a composite coefficient for the total drainage area is computed. In order to find a scenario type runoff coefficient, possible future land use types for the potential development of the undeveloped area are also considered. Finally, a composite runoff coefficient is calculated by using weighted average of the current and future land cover types. In this study, it is found as 0.9 (Table 2), which represents highly urbanized or impervious surface conditions in the future.

Hydraulic Calculations

Rational formula is used to recalculate the peak runoff of pilot study area where stormwater network information is available. The rational method makes the basic assumption that the peak rate of surface outflow from a given watershed is proportional to the watershed area and average rainfall intensity over a period of time (time of concentration) just sufficient for all parts of the watershed to contribute to the outflow (Burke & Burke, 2015). The rational formula is written as:

$$Q=CIA \qquad (1)$$

Where Q is the peak runoff, C (runoff coefficient), is the ratio of peak runoff rate to average rainfall rate over the watershed during the time of concentration (T_c), I is the rainfall intensity and A is the contributing area of watershed under consideration.

In the application of rational formula, an intensity-duration-frequency curve is used to identify intensity for the selected return period and storm duration that equals to the T_c. This is then multiplied by the drainage area and runoff coefficient to determine the peak discharge rate.

Data from the results of precipitation analyses for observation and projection periods are used to calculate the peak discharge. Time of concentration and return period are chosen 15 minutes and 2 years, respectively according to design standards of AWSA. Moreover the 15 minutes - 5 years storm depths were also used for the performance analysis of baseline storm sewer design. For the calculations, rainfall intensities (return levels in mm) are first converted to mm/minutes and then to l/sec/ha in this study. This is then multiplied by the area and runoff coefficient to determine the peak discharge rate.

Furthermore design parameters such as percent fullness, maximum and minimum velocity and minimum water height in the channels are calculated for the new intensities to observe their change. Maximum discharge (Q_{full}) and discharge of the pipe (Q_{design}) are calculated by using Q_{design}/Q_{full} ratio. V_{design} is calculated with respect to V_{design}/V_{full} ratio where V_{full} is the velocity at the full capacity. Finally, water depth (h) is calculated by using depth ratio (h/D). Kutter formula is used to calculate the velocity of the maximum discharge.

$$V = \frac{100\sqrt{R}}{m+\sqrt{R}} \sqrt{RJ} \qquad (2)$$

Where R is hydraulic radius, J is slope (hydraulic gradient) and m is Kutter constant (0.35 in this study for concrete and rainforced concrete pipes). Then, the maximum discharge (Q_{full}) is calculated using the formula below where A is the cross-sectional area of the drain:

$$Q_{full} = V_{full} \text{ x } A \qquad (3)$$

The maximum and minimum velocity and minimum water height in the channels are investigated considering the baseline design criteria (Vmin=0.5 m/s, Vmax=5 m/sand h_{min} > 20 mm) to identify whether the system is overdesigned or exceed capacity. If the system is overdesigned pipe diameter can be decreased, the slope must be increased or both can be applied in order to stay in the design ranges. In this study only diameters are changed to find a better allocation of pipes.

Analysis conducted for the following cases;

Case 1; only the design intensity has changed and the behavior of existing stormwater network is observed.

Case 2; only design runoff coefficient has changed (from 0.8 to 0.9) and the effect is observed.

Case 3; both design intensity and runoff coefficient have changed and the changes in pipe capacities (percent fullness) of existing network is observed.

Case 4; design intensity has changed and pipe diameters are optimized.

Case 5; design intensity and runoff coefficient have changed and pipe diameters are optimized.

Results and Discussion

In order to figure out the performance of current (existing) stormwater network of pilot study area rainfall return levels that are obtained from observation and projection periods with stationary and nonstationary models are used to calculate peak runoff in rational formula. In addition, land use/cover change effect is integrated as composite runoff coefficient (0.9) for the pilot study area.

Existing network design considered the time of concentration 15 minutes and the return period 2 years and therefore, the same duration and return period is used for the analyses in this study. On the other hand, the minimum return periods of 5 - 10 years were also recommended for urban stormwater networks where such a critical facility was established (Burke & Burke, 2015; Efe, M., 2006). If there is greater possibility of damage and loss, then the risk can be reduced by preferring greater frequencies. Thus, the baseline network capacity is also investigated for 15 minutes 5 years return period values. These (15 minutes - 5 years) design storm depths were obtained from the observation and projection results. However, only the results from observation period were used for the analyses because the 15 minutes - 5 years storm intensities of projections were lower than the baseline design intensity (15 minute – 2 year). The design rainfall intensity (baseline design intensity) for 15 min and 2 year frequency used in the existing network design is 116 lt/s/ha. Generally, the operational status of the stormwater network is described as unsatisfactory if the specified hydraulic criteria (e.g. excess pipe capacity, velocity is out of the range) is violated (Nanos & Filion, 2016; Gouri & Srinivas, 2015) In this study, the capacity surcharge of pipes, which is the excess of percent fullness ratio, is described as a failure.

Results from Case 1, 2, and 3

Figure 3 shows the percent of the network pipe volume as function of percent fullness for each projection model, observation and baseline design (Case 1). In addition, Figure 4 demonstrates maximum percent fullness capacity experienced by each pipe for each entry in Case 1. Considering the current climate conditions, 96% of the network pipe volume is under the 60% capacity for the baseline design as shown in Figure 4.1. Only the 4% of the network pipe volume shows a maximum pipe capacity higher than 60%. The 69% of the network pipe volume is reaching a maximum capacity lower than 20% (Figure 3).

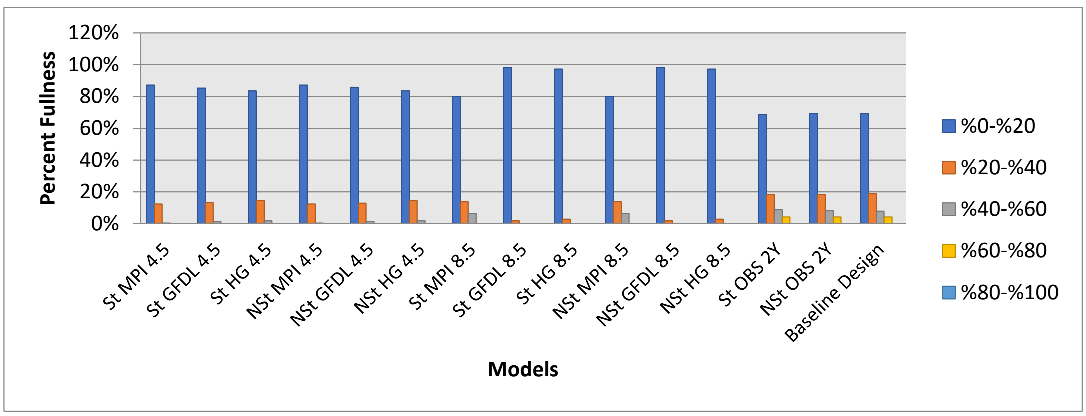

Figure 3. Percent fullness of projection model, observation and baseline design without Pipe Revision-Case 1**.**

With regard to the climate change scenario (Case 1), remarkable changes in the system performance can be observed when compared with current conditions in terms of pipe capacity ranges. Specifically, the maximum pipe capacity reached about the 85% of the network pipe volume stay within the %0-%20 range, while about %15 falls in the %20-%40 range (Figure 3). Even flows at %0-%20 capacity increases to around 98% pipe volume from GFDL and HG with 8.5 scenarios. There are no pipe flows that exceed 60% capacity for all the climate change scenarios. On the other hand observed data in stationary and nonstationary conditions exhibit a parallel capacity range with the baseline design. In Figure 4.2, observed data with and without nonstationarity provided the highest maximum fullness capacity (75%) while all scenarios showed lower than 60% maximum fullness capacity. Lowest fullness capacities occurred with RCP8.5 models.

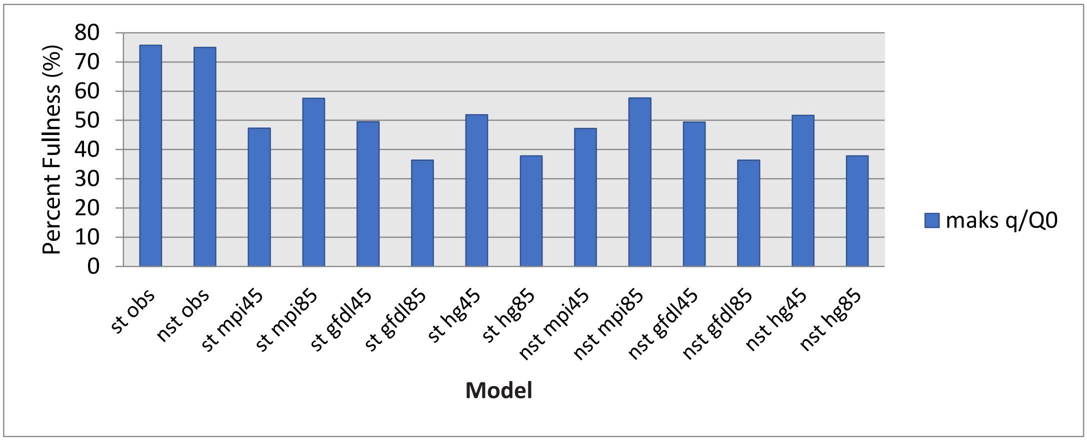

Figure 4. Maximum capacity experienced during the projection and observed period by each pipe.

Figure 5 demonstrates the percent fullness change for baseline design when only runoff coefficient is changed from 0.8 to 0.9 in Case 2. Rainfall intensity for the existing storm network remained the same. With increasing runoff coefficient, the flows at %0-%20 range slightly decrease (4%) and they contribute to higher percent fullness range (%80-%100) with 2 percent. However, the general effect of surface runoff coefficient is negligibly small for this study area. Figure 6 exhibits the pipe capacities for land use change combined with climate change scenarios without any revision for the existing network as Case 3.

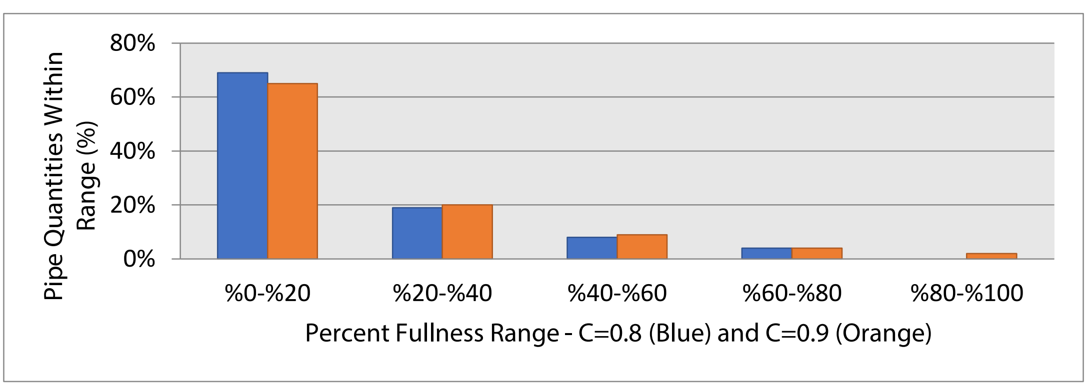

Figure 5. Baseline design percent fullness with surface runoff coefficients of C=0.8 (Blue) and C=0.9 (Orange) in Case 2.

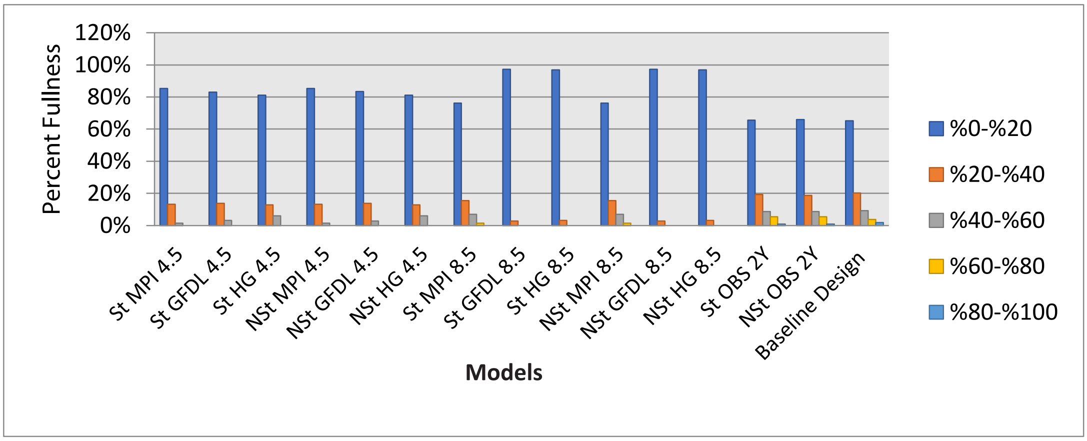

Figure 6. Percent fullness of models without pipe revision-climate change and land use change (C=0.90) – Case 3.

The system operated in a satisfactory state refers the condition that the maximum volume that the network system conveys as result of climate change projections for the extreme rainfall will decline throughout the projection period. With the combined effect (or the effect of 10 percent higher surface runoff coefficient) the flows in pipes shift to higher fullness capacity ranges with projection models and observed conditions. However, there is still no flow at fullness range of %80-%100 and even at the range of %60-%80 in any scenario models (Figure 6). Only observed stationary and nonstationary analyses show pipes with 1% within %80-%100 fullness range. But, they are still less than pipe volumes of baseline design (2%). The outputs produced from the simulations show that performance of the case study stormwater network was observed to operate in a satisfactory state for the climate change scenario experiments, where an unsatisfactory state is defined as an occurrence of the conduit/pipe capacity (percent fullness) exceeding 90%. These results indicate that system performance will be satisfactory by the end of the century.

Results from Case 4 and 5

The existing stormwater network design is observed to operate in satisfactory conditions considering the projected rainfall data however a better allocation of pipe diameters can be achieved when the percent fullness data is examined. For this reason other design parameters such as velocity and minimum depth are compared. The maximum and minimum velocity and minimum water height in the channels are presented in Table 3 for Case 1. Stationary and nonstationary projection results reveal closer values while observed period shows the highest velocity, depth and fullness percent for stationary and nonstationary conditions.

Minimum velocity values indicate that the system is over designed and pipe diameter must be decreased or the slope must be increased in order to stay in the design ranges. RCP 8.5 based results for GFDL and HG models show lower rainfall intensities so the velocity (0.1 m/s for minimum) in the pipes and the fullness (~36%) decrease for these models. Also, the water depth in the pipes reaches its lowest value (0.3 cm) for these models. With regard to models results, decreasing trend of extreme precipitation, which is the outcome of projection results, can be one of the reasons that existing system stay satisfactory over time. That means stationary assumption reveals more conservative design conditions for the future but with higher economy of scale.

Table 3

Velocity, Percent Fullness, H Minimum Results For Models Obtained For Model Driven Intensities For Baseline Design System

Model	Maximum velocity (m/s)	Minimum velocity (m/s)	Max q/Q_0 (%)	H minimum (cm)
St Obs 2 Years	3.03	0.26	75.7	1.62
Nst Obs 2 Years	3.02	0.25	75	1.62
St Mpi 4.5	2.44	0.2	47.3	0.54
St Gfdl 4.5	2.51	0.2	49.5	0.75
St Hg 4.5	2.59	0.21	51.9	0.75
St Mpi 8.5	2.76	0.23	57.6	1.5
St Gfdl 8.5	1.95	0.1	36.4	0.3
St Hg 8.5	2.03	0.1	37.8	0.3
Nst Mpi 4.5 2.43		0.2	47.2	0.54
Nst Gfdl 4.5	2.51	0.2	49.4	0.75
Nst Hg 4.5	2.59	0.21	51.7	0.75
Nst Mpi 8.5	2.76	0.23	57.7	1.5
Nst Gfdl 8.5	1.95	0.1	36.4	0.3
Nst Hg 8.5	2.03	0.1	37.8	0.3

Furthermore the existing stormwater network is redesigned in terms of pipe diameter to obtain more optimal and economical solutions and compare them with the current quantities. Figure 7 and Figure 8 exhibit the pipe capacities for only climate change effect and combined effect of climate change with land use change scenarios, respectively with revision made to the existing network (Case 4 and Case 5).

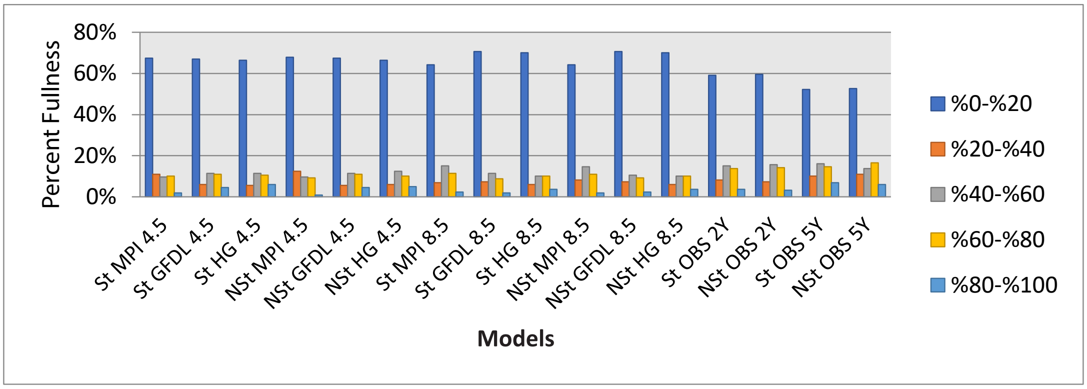

Figure 7. Percent fullness of models with pipe revision to climate change – Case 4.

In the climate change and revised system scenario (Case 4), the system performance (capacity of fullness) increases. Approximately the 15% and 25% of the network pipe volumes show a maximum pipe capacity higher than 60% and 40%, respectively for the projection scenarios. The surcharged range (80%-100%) network pipe volume is equal to 3% for RCP8.5 and %5 for RCP4.5 but these pipe volumes do not exceed the 90% capacity ratio significantly (only 1 or 2 pipes do exceed) (see Figure 7). In the 15 minutes 5 years scenario that is originated from observed stationary and nonstationary return levels, the revised system produces also a better drainage system performance, if compared with the baseline design. For example, the maximum pipe capacity within 0%-20% dropped to 52%. About 40% of the network pipe volume has a maximum pipe capacity higher than 40% capacity ratio and almost the 16% of the pipes is within 60%-80% capacity range (Figure 7).

In addition to climate change scenario, also the new composite runoff coefficient (0.9) is applied to the design (Case 5). The revised system performance simulated under these new conditions and its results are given in Figure 8. The surcharged range (80%-100%) of network pipe volume is further increased and there are pipe volumes that exceed the 90% (1-6% for projections and 4-7% for observation) and 100% (1% for projections and 3% for observed 5-year stationary frequency) capacity ratio significantly. On top of climate change effect, adding runoff coefficient increases the pipe volumes 4 percent for projections and 6 percent for observed 2-year and 5- year frequency at the capacity range of %80-%100 (see Figure 7 and Figure 8). Prior to revision applied to the system, either climate change effect or joined effect has no significant change on the system performance but after the revision it becomes critically important. For the 15 minutes 5 years return levels, increase in the rainfall intensity cause system failure; about 10% of the pipes excess capacity for the baseline (existing) network.

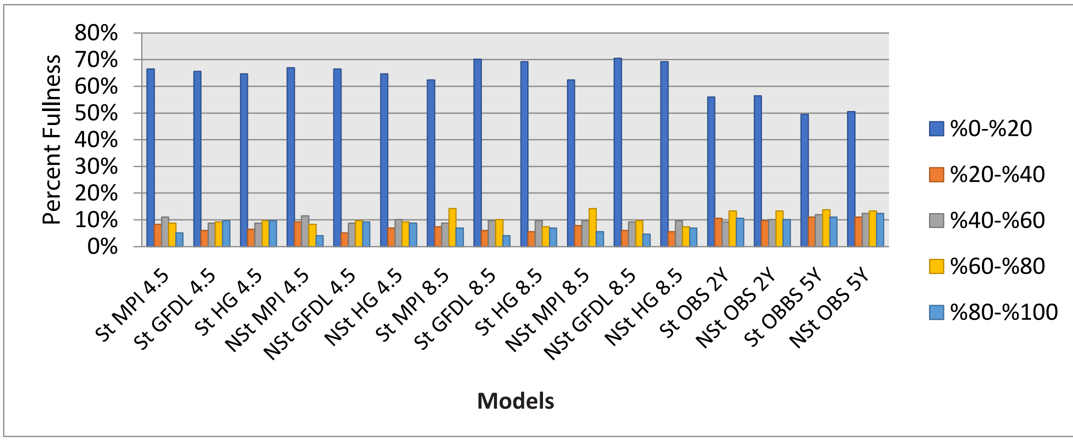

Figure 8. Percent fullness of models with pipe revision to climate change and land use – Case 5.

The hydraulic performance of the baseline and revised system for the observed and projected rainfall data has been compared in terms of the pipe capacity ratio associated with various pipe capacity range (0-20%, 20-40%, 40-60%, 60-80% and 80-100%) for five cases. It can be seen that system continuity can be satisfied with various design conditions, including climate change and land use, for all return level results. The remarkable point is that the system can perform with lower pipe diameters than it is designed. Figure 9 shows quantities of small size and large size pipes in meters for every diameter (200 mm to 1600 mm) and total quantities of small (200 mm to 600 mm) and large (800 mm to 1600 mm) size pipes for each model entry and baseline design. The revision of the system by pipe diameter, which is represented in Figure 9, results in a reduction of the oversized network.

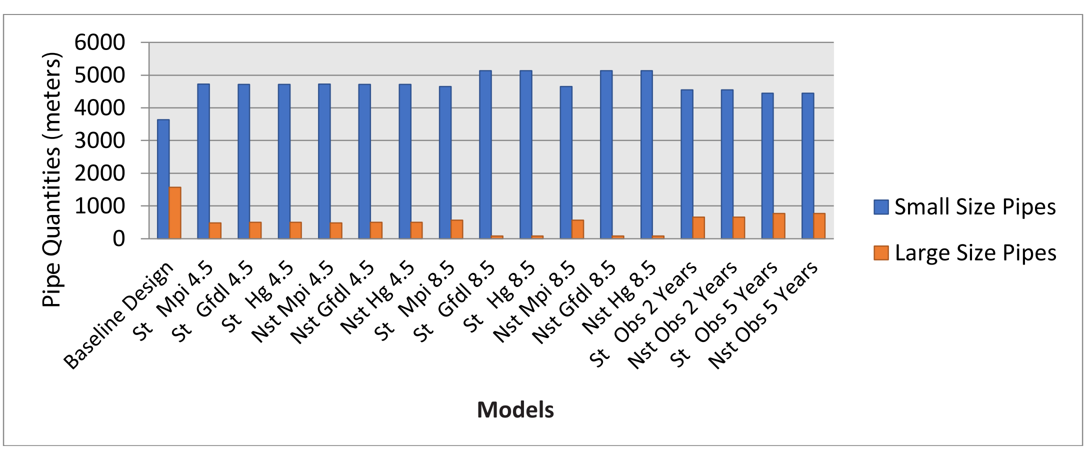

Figure 9. Small vs. large size pipe quantities – Baseline and revised system pipe lengths in meters - Small (200-600 mm) and Large (800-1600 mm).

Table 4 exhibits the % change in pipe length for every diameter after revision compared with baseline design. The negative and positive signs show decrease and increase in the pipe length change, respectively. Figure 9 also shows the total small and large size pipe quantities (lengths in meters) of baseline (existing) design and revised network's small and large size pipe quantities due to climate change scenarios. The current system also can perform well under the 15 minutes 5 years return period loads with revisions in pipe diameter such as increase in large size and decrease in small size quantities considering the revised system which can be seen from Figures 7, 8 and Table 4. The revised network for 2 years - 15 minutes rainfall intensities can also be designed with several changes in pipe diameter so that 15 minutes - 5 years return period loads can be tolerated which at the end result increase in small size pipes and decrease in large size pipes with respect to existing (baseline) design pipe quantities (see Figure 9 and Table 4). The largest decrease (95%) and increase (41%) in pipe sizes occurred with model projections of GFDL 8.5 and HG 8.5 for stationary and nonstationary conditions. Changes (22-25% and 51-58%) in observed cases with 2-yr and 5-yr return periods stayed below the changes (>28% and >64%) for all projected cases after revision.

Economical Analysis

Furthermore the effect of design revision due to changing climatic conditions is reflected in terms of cost in Figure 10. Only the cost associated with pipe diameter is calculated by using actual project unit price. The pipe diameters (small size or large size group) determine the choice of concrete or reinforced concrete for which unit prices are different. Therefore, it directly affects total cost, which decreases with model results. In other words, with the revision large size pipes (reinforced concrete) are generally replaced with small size pipes (concrete) and this reduces the total cost.

The models that reveal lower rainfall intensity have the lowest total costs such as stationary GFDL with RCP8.5 or HG with RCP8.5. In general, RCP4.5 results reveal higher total cost than RCP8.5 results probably because of the less warming that may cause higher precipitation, besides MPI model stationary and nonstationary results. Observed data driven design alternatives have the higher cost among the all alternatives, yet they are still lower than the existing design. Almost 400K euro the baseline design, which was designed with 15-min and 2-yr storm is more costly and even increasing the return period to 5-yr it is still more expensive. In baseline design, the cost for small size pipes (300-600 mm) is almost four times lower than the cost for large size pipes (800-1200 mm) but in all projected designs the cost for small diameter pipes is higher. The cost distribution for observed data driven designs (2-yr and 5-yr) is similar to the baseline design but with very low total cost.

Table 4

Pipe Diameter Change for the Models Compared with Baseline Design

Model	Ø300	Ø400	Ø500	Ø600	Total Change (Small Ø)	Ø800	Ø1000	Ø1200	Total Change (Large Ø)
St Mpi 4.5	37%	-52%	469%	-51%	30%	-58%	-95%	-100%	-69%
St Gfdl 4.5	37%	-2%	245%	-54%	30%	-59%	-90%	-100%	-69%
St Hg 4.5	35%	-19%	335%	-41%	30%	-64%	-77%	-100%	-69%
Nst Mpi 4.5	32%	-29%	469%	-51%	30%	-65%	-77%	-100%	-69%
Nst Gfdl 4.5	35%	7%	245%	-54%	30%	-59%	-90%	-100%	-69%
Nst Hg 4.5	35%	-23%	357%	-41%	30%	-64%	-77%	-100%	-69%
St Mpi 8.5	31%	-45%	496%	-50%	28%	-58%	-77%	-100%	-64%
St Gfdl 8.5	55%	0%	229%	-66%	41%	-95%	-95%	-100%	-95%
St Hg 8.5	55%	0%	229%	-66%	41%	-95%	-95%	-100%	-95%
Nst Mpi 8.5	29%	-36%	496%	-50%	28%	-58%	-77%	-100%	-64%
Nst Gfdl 8.5	55%	0%	229%	-66%	41%	-93%	-100%	-100%	-95%
Nst Hg 8.5	55%	0%	229%	-66%	41%	-95%	-95%	-100%	-95%
St Obs 2 Years	26%	-18%	190%	36%	25%	-48%	-77%	-100%	-58%
Nst Obs 2 Years	26%	-20%	190%	36%	25%	-48%	-77%	-100%	-58%
St Obs 5 Years	24%	-62%	91%	139%	22%	-70%	-6%	-48%	-51%
Nst Obs 5 Years	24%	-56%	64%	138%	22%	-70%	-6%	-48%	-51%

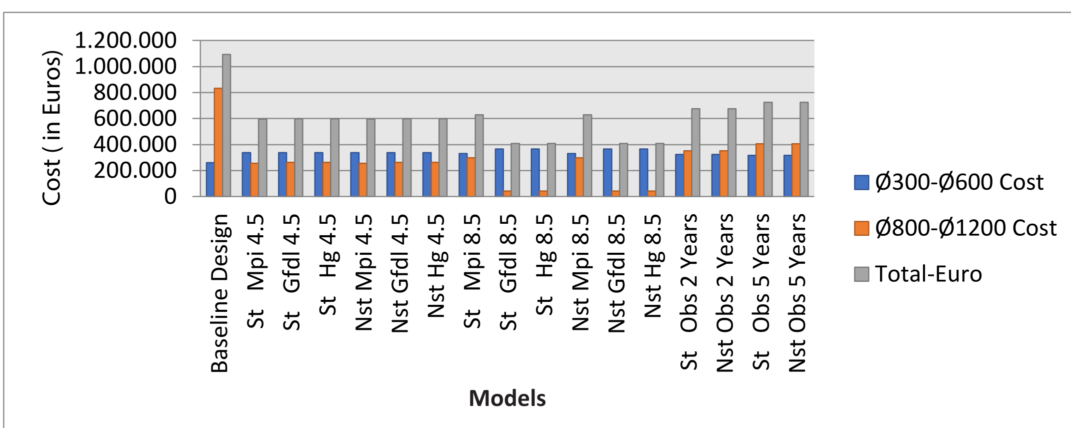

Figure 10. Cost Comparison of Models after Revision (in Euros).

Conclusions

The impact of climate and land use changes on the capacity of a storm water drainage system in Etimesgut, Ankara of Turkey was investigated. The capacity of the system was evaluated under the current condition using the data from 1950 to 2015 and future condition using the data from three different GCM results with two different emission scenarios. The rainfall intensity of 15 min and 2 year frequency from all data entries was conveyed through the system. The resulted capacity (percent fullness) was compared with the capacity available from existing network that also used 15 min – 2 year rainfall frequency. The higher load obtained from 15 min – 5 year storm for current period was also added to the analyses. The nonstationarity effect in maximum rainfall depth was also considered in this impact study. Finally, alternative solutions (reduction in pipe sizes) were applied to the existing sewerage system and their implications were evaluated.

The system operated in a satisfactory state totally under current and future conditions when climate change (case 1) and land use change (case 2 case 3) were considered. It can be concluded that according to climate change projections for the extreme rainfall with and without nonstationarity, the maximum volume that the system face will not exceed baseline design criteria throughout the projection period. Combination of climate change and land use change conditions also stay satisfactory for the baseline design which used 15 minutes storm duration and 2 years return period as intensity input. Increase in runoff coefficient from 0.8 to 0.9 did not yield important changes in flow amount but joining it to climate change produced significant flow loads in the current system towards higher depth ratio values. The current network system can perform appropriately under the loads from 1950-2015 and 2015-2098 periods that are delivered by stationary and nonstationary assumption. The stationary assumption reveals more conservative design conditions for the future but with higher cost. The capacity of the system with both stationary and nonstationary is expected to be not overload in the future. On the other hand the system may fail under the loads derived separately or together by longer storm duration (over 15 minutes and more) or higher return periods (such as 5 years and more) for such a critical facility and area. Under these conditions, selecting the design parameters that satisfy the demand and requirements gains importance. Dikici (2018) indicates the importance of proper and optimum solution for water sewer system of the major urban areas and draw attention to updated design criteria considering the increasing population and urbanization.

Overall, the total cost results of alternative design options indicate that the storm sewer system can be built at a lower total cost not only for all climate change and land use options but also for the 2 and 5 years 15 minutes observed data driven return level results. On the other hand the unit cost of pipes are unique to the project (regarding this, information that have been presented in this study cannot be used or reproduced without permission) due to tender method and cannot be generalized.

By contrast with this study, the general outcome of the future period studies with regard to stormwater networks in literature is that current systems will probably fail and cannot withstand considering the future climate conditions. For instance, Thakali et al. (2016) specify that the present capacity of most urban drainage systems is expected to be overload in the near future and the analysis of the present stormwater facilities of the Flamingo and Tropicana watershed showed that these facilities are unable to sustain their performance under the loads resulting from the projected climate scenario. Furthermore, Osman (2014) also showed that in the future the urban drainage system of the area that he studied could react differently in terms of increase in number of surcharged sewers and from manholes surface flooding. Larsen et al. (2008) pointed out that a 100-year event in the control period for Sweden will be shorten due to climate change scenarios and damages caused by urban flooding will probably occur more frequently.

With regard to stormwater network design in Ankara, there are two important aspects; one is selection of the design parameters and the other is application of these parameters in construction phase in an appropriate environment. For instance selecting an appropriate design load (e.g. storm duration and return period) is important for the network design on the other hand if the runoff cannot be routed correctly to the system then surface flooding occurs. Infrastructure is relatively a long term investment and conditions may change during the proposed design life such as decrease in pervious land, population growth etc. lasting with increasing load exposure for the system, so monitoring is an essential part for a vital stormwater management process.

Selecting the optimum parameter is another issue, for the changing environmental conditions together with urbanization bring out the need for a new approach. Design parameters cannot be assumed stationary for such a long term design lives so temporal, spatial or other changes must be considered for the design process which makes involvement of multiple bodies to the design process necessary. While selecting the design approach and parameters also a risk based approach should be applied; for this not only physical damage but also environmental and social cost of the event must be considered. The cost of designing a network for 10 minutes - 10 years return period rainfall will probably be higher than a 15 minutes - 2 years return period rainfall based design as the design intensity increases which brings out the larger pipe diameters. Nevertheless decision making mechanism must take into account not only the extra cost derived by the design parameter but also the cost of loss of life, reputation, interruption of business etc. Because Ankara is the capital of Turkey, center of the bureaucracy, transportation hub for high speed rail and host of many entities that are determining bodies of economic and social state of affairs design process of stormwater network must consider the above mentioned conditions.

Acknowledgement

Based on the article, two presentations were published at the 3rd International Electronic Conference on Water Sciences (ECWS-3) which was held online, promoted by the open access journal *Water* http://www.mdpi.com/journal/water; one presentation is presented and one full-text is published at 10th National Hydrology Congress, Muğla Sıtkı Koçman University, The presentations that are published at ECWS-3 can be found at:
https://sciforum.net/paper/view/conference/5807
https://sciforum.net/paper/view/conference/5808

References

Bahadur, A., Tanner, T., & Pichon. F. (2016). *Enhancing Urban Climate Change Resilience: Seven Entry Points for Action* (No.47). Manila: Asian Development Bank.

Batuman, B. (2013). City profile: Ankara. *Cities, 31,*. 578–590. doi:10.1016/j.cities.2012.05.016

Burke, C. B., & Burke, T. T. (2015). Stormwater Drainage Manual 2015. *Indiana Local Technical Assistance Program (LTAP) Publications*. Retrieved from https://docs.lib.purdue.edu/inltappubs/100

Dikici, M. (2018). Megakentlerin Kanalizasyon Hatlarının Etkili İşletmesi ve İstanbul Örneği. *International Green Capitals Congress*.Konya.

Efe, M. (2006*). Review of modelling softwares used in design and operation of sewerage and storm water collection systems*, (Master's Thesis, Graduate School of Natural and Applied Sciences, Istanbul Technical University). Retrieved from http://hdl.handle.net/11527/8941

Elshorbagy, A., Lindenas, K., & Azinfar, H. (2018). Risk-based quantification of the impact of climate change on storm water infrastructure. Water Science, 32(1), 102-114.

Hailegeorgis, T. T., & Alfredsen, K. T. (2017). Analyses of extreme precipitation and runoff events including uncertainties and reliability in design and management of urban water infrastructure. *Journal of Hydrology, 544*, 290-305.

Kundzewicz, Z. W. (2003). Proceedinss of symposium HS02b: *Extreme Precipitation and Floods in the Changing World, Water Resources Systems—Hydrological Risk, Management and Development*. Sapporo.

Larsen, A. N., Gregersen, I. B., Christensen, O. B., Linde, J. J., & Mikkelsen, P. S. (2008). 11the International Conference on Urban Drainage: *Future Development in Extreme One-Hour Precipitation Over Europe Due To Climate Change* (Vol. CD-ROM). Sheffield, UK: University of Sheffield.

NYC Environmental Protection. (2012). *Guidelines for the Design and Construction of Stormwater Management Systems*. New York: New York City Department of Environmental Protection Printing Office.

Osman, Y. (2014). Monitoring the future behaviour of urban drainage system under climate change: a case study from north-western England. *Open Engineering, 5*(1). doi:10.1515/eng-2015-0003

Oruc, S., Yucel, I., & Yilmaz, A. (2019) *Investigation of The Effect of Climate Change on Extreme Precipitation: Capital Ankara Case* (Manuscript submitted for publication).

Özkil, A. (2015). *Searching for the Crumbles of Nature in the Metropolitan Ankara* (Unpublished manuscript). Middle East Technical University, Ankara.

Rosenberg, E. A., Keys, P. W., & Booth, D. B. (2010). Precipitation extremes and the impacts of climate change on stormwater infrastructure in Washington State. *Climatic Change, 102*, 319. Retrieved from https://doi.org/10.1007/s10584-010-9847-0

Shuster, W. D., Bonta, J., Thurston, H., Warnemuende, E., &. Smith, D. R. (2005). Impacts of impervious surfaces on watershed hydrology: A review. *Urban Water Journal, 2*(4), 263–275.

Simonovic, S. P., Schardong, A. Sandink, D., & Srivastav, R. (2016). A Web-based tool for the development of intensity duration frequency curves under changing climate. *Environmental Modelling & Software Journal, 81,*136-153.

Sun, Z., Guo, H., Li, X., Huang, Q., & Zhang, D. (2011). Proceedings of the ASPRS 2011 Annual Conference: *Effect of LULC change on surface runoff in urbanization area*. Milwaukee: Wisconsin.

Texas Department of Transportation . (2016). Hydraulic Design Manual.

Thakali, R., Kalra, A., & Ahmad, S. (2016). Understanding the effects of climate change on urban stormwater infrastructures in the las vegas valley. *Hydrology, 3*, 34.

Trenberth, K. E. (1998). Atmospheric moisture residence times and cycling: Implications for rainfall rates with climate change. *Climatic Change, 39*, 667-694.

United Nations, (2018). World urbanization prospects: The 2018 revision. New York.

Yilmaz, E. (2013). *Ankara Şehrinde Is*ı *Adas*ı *Oluşumu*(Doctoral dissertation).Retrieved from file:///C:/Users/s.ozturk/Downloads/10055965.pdf

Zhou, Q. (2014). A review of sustainable urban drainage systems considering the climate change and urbanization impacts. *Water, 6*(4), 976-992.

Extended Turkish Abstract
(Genişletilmiş Türkçe Özet)

İklim Değişikliğinin Yağmursuyu Şebekeleri Üzerine Etkisinin Araştırılması: Başkent Ankara Örneği

Kentsel su altyapı sistemleri, örneğin yağmur suyu sistemleri ve barajlar gibi taşkın kontrol yapıları ekstrem yağış özelliklerine göre tasarlanırlar ve bu yağış özellikleri de Şiddet-Süre-Tekerrür eğrileri ile ifade edilir (Peck vd., 2012; Hosseinzadehtalaei vd., 2017). Şiddet-Süre-Tekerrür eğrileri, farklı yağış sürelerinde belirli bir şiddette bir yağışın meydana gelme sıklığını ölçmektedir. Şiddet-Süre-Tekerrür eğrileri, genel olarak geçmiş döneme ait yağış analizleri ve istatistiklerine dayanmaktadır. Altyapı tasarımı da uzun süreden beri zaman içinde ekstrem olayın sıklığında bir değişiklik olmadığını varsayarak oluşturulmuş kriterlere dayanmaktadır. Ancak ekstrem yağışların sıklığı ve şiddetinin değişiklik göstermekte olduğu ve bu değişikliklerin gelecekte de muhtemelen devam edeceği anlaşılmıştır (Cheng ve Aghakouchak, 2014). Ayrıca, değişen frekansın ihmal edilmesinin ekstrem olayları hafife alan Şiddet-Süre-Tekerrür eğrileri ile sonuçlandığı görülmüştür (Cheng ve Aghakouchak, 2014). Sarhadi vd. (2017) de yaptıkları çalışmalarda Cheng ve Aghakouchak (2014) ile benzer sonuçlar ortaya koymuştur. Sonuçlara göre durağan yani zaman içerisinde değişikliğin ihmal edildiği yaklaşımın ekstrem yağış olaylarını hafife alabileceği dolayısıyla da değişen koşullarda Şiddet-Süre-Tekerrür eğrilerinin ve tasarım kriterlerinin güncellenmesinin gerekliliği ortaya konmuştur. Hosseinzadehtalaei vd. (2017), ekstrem yağışların özelliklerinin değişmekte olduğunu vurgulamış ve zamanla herhangi bir değişiklik göstermeyen Şiddet-Süre-Tekerrür eğrilerine dayanan mevcut tasarım standartlarına dikkat çekmiştir.

Altyapı sistemlerini, özellikle de yağmur suyu sistemlerini, etkileyen önemli bir faktör de kentleşmedir. Kentleşmenin sonucu olarak geçirimsiz alanların artış göstereceği, bunun da değişen yüzey örtüsü ile birlikte deşarj hacminde artışa neden olacağı, iklim değişikliğinin de etkisi ile yağışların frekansı ve şiddeti göz önüne alındığında kentsel yağmur suyu sistemlerinin daha yoğun etkilere maruz kalacağı belirtilmiştir (Thakali vd., 2016). Son veriler, Avrupa'daki tarım arazilerinin, kentsel alanlara ve altyapı tesislerine kalıcı olarak dönüştürüldüğünü ve kentsel genişlemenin büyüklüğünü teyit etmektedir (AÇA, 2017). Arazi örtüsü ve kullanımı ile iklim değişmekte olup; arazi kullanımı ve özelliklerinde değişiklikler ile kentsel taşkınlara neden olan ekstrem yağış olayları birlikte ele alındığında bu değişikliklerin yağmur suyu şebekelerinin tasarımı ve yönetimi üzerindeki etkilerinin değerlendirilmesine duyulan ihtiyaç ortaya çıkmaktadır (Hailegeorgis ve Alfredsen, 2017).

Türkiye'de yıllık ekstrem olay sayısı, Meteoroloji Genel Müdürlüğü 2017 İklim Değerlendirme Raporu'na (2018) göre 1940-2017 döneminde artış eğilimi göstermektedir. 2017 boyunca en tehlikeli ekstrem olayların şiddetli yağmur / sel (% 31), rüzgar fırtınası (% 36), dolu (% 16), yoğun kar (% 7) ve yıldırım (% 4) olduğu görülmektedir. Türkiye'nin başkenti olan Ankara, Sakarya ve Kızılırmak Havzaları içerisinde yer almaktadır. Ankara yarı kurak bir iklime sahiptir ve sürekli bir nüfus artışına dolayısıyla da geçirimsiz yüzeylerin artmasına neden olan yüksek bir kentleşme oranına sahip olup iklim değişikliğinden önemli ölçüde etkilenmiştir. Bu da kentsel altyapı üzerinde artan bir baskıya neden olmaktadır. Ayrıca, son 20 yıldaki sel olayları gözlemlendiğinde, yoğun yağış ve ani selin, mülklere çeşitli zararlar vermiş olduğu ve hatta kentte can kaybına neden olduğu görülebilir. Bununla birlikte, tüm bu olumsuz olaylara rağmen, özellikle iklim değişikliği etkisinde ekstrem yağışta meydana gelebilecek değişikliklere ve Şiddet-Süre-Tekerrür eğrilerinin bu değişikliklerden nasıl etkileneceğine odaklanan detaylı çalışmalar Türkiye'nin Başkenti Ankara için eksik kalmıştır

İklim değişikliği ve arazi kullanımından kaynaklı değişikliklerin Ankara ili, Etimesgut ilçesindeki bir yağmur suyu drenaj sisteminin kapasitesi üzerindeki etkileri incelenmiştir. Sistemin kapasitesi 1950›den 2015›e kadar olan gözlem verileri ile iki farklı emisyon senaryosuna sahip üç farklı iklim modeli sonucundaki projeksiyon verileri kullanılarak değerlendirilmiştir. Etki analizi yapılırken tüm model ve gözlem sonuçlarından elde edilen 15 dakikalık süreli ve 2 yıllık tekerrüre sahip yıllık maksimum yağış verileri kullanılmıştır. Elde edilen kapasite verileri (doluluk oranı), mevcut şebekenin tasarımında kullanılan kapasite oranları ile karşılaştırılmıştır. Analizlere ayrıca gözlem periyoduna ait 15 dakikalık-5 yıl tekerrür süresine sahip yağış şiddeti de eklenmiştir. Bu çalışmada hidrolik hesaplamalarda kullanılan yıllık maksimum yağış yüksekliğine durağanlığın etkisi de dikkate alınmıştır.

Bir drenaj alanı, farklı akış katsayıları olan alt alanlardan oluşur. Alanların kullanımı / arazi örtüsü özelliklerine karşılık gelen akış katsayıları ve bu alanların akış katsayıları ile çarpımlarının toplamı, toplam alana bölünür ve daha sonra toplam drenaj alanı için bir bileşik akış katsayısı hesaplanır. Çalışma alanında gelecekte meydana gelebilecek akış katsayısını bulmak için, herhangi bir düzenlemeye maruz kalmamış alanların gelecekteki olası arazi kullanım tipleri dikkate alan potansiyel maksimum gelişim senaryosu ile alanların arazi örtüleri belirlenmiştir. Son olarak, mevcut ve gelecekteki arazi örtüsü tiplerinin ağırlıklı ortalaması kullanılarak bileşik bir akış katsayısı hesaplanmıştır. Çalışma alanı için gelecekte yüksek oranda kentleşmiş ve geçirimsiz yüzey koşullarını temsil eden akış katsayısı 0.9 olarak bulunmuştur. Son olarak, mevcut yağmursuyu sistemine alternatif çözümler (boru çaplarında varyasyonlar) uygulanmış ve etkileri değerlendirilmiştir.

Tüm analizler göz önüne alındığında, mevcut sistemin, durağan ve durağan olmayan yaklaşımla tespit edilen 1950-2015 ve 2015-2098 dönemleri arasındaki yükler altında uygun şekilde çalışabildiği görülmüştür. Gelecek dönem için tespit edilen yükler göz önünde bulundurulduğunda durağan varsayım ile yapılan tasarımın daha konservatif olduğu anlaşılmıştır ancak bunun maliyetinin daha yüksek olacağını da unutmamak gerekir. Hem durağan hem de durağan olmayan koşullar için sistemin kapasitesinin gelecekte aşırı yüklenmeye maruz kalması beklenmemektedir. Öte yandan, sistem, böyle kritik bir tesis ve alan için ayrı ayrı veya birlikte daha uzun yağış süreleri (15 dakika ve üzeri) veya daha yüksek tekerrür periyotları (5 yıl ve daha fazlası gibi) ile elde edilen yükler altında istenen ve beklenen performansı gösteremeyebilir. Bununla birlikte, modellerin yağış şiddeti ve sıklığı gibi tahminlerindeki farklılıklarının, mevcut tasarım parametrelerini en yeni veriler ve yaklaşımlarla güncelleme ihtiyacını desteklediğine dikkat edilmelidir. Ortaya çıkan farklılıklar ayrıca iklim modellerinden elde edilen gelecek dönemlere ait verileri de kullanarak analiz yapma ihtiyacını ortaya koymaktadır.

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