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Seawater quality at the brine discharge site from two mega size seawater reverse osmosis desalination plants in Israel (Eastern Mediterranean)

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ABSTRACT

Two mega-size seawater desalination plants, producing 240 Mm^3/y freshwater, discharge brine into the Mediterranean coast of Israel through two marine outfalls, located 0.8 km apart. Six years monitoring brine discharge have shown almost no impact on seawater quality. The brine dispersed near the bottom following its initial mixing, and was not detected near the surface. Maximal excess salinity at the salty layer ranged from 4.3 to 9.1% over the reference and the affected area was highly variable (2 km^2 - >13 km^2), with maximal plume size from 1.75 to more than 4.4 km. Brine increased seawater temperature by up to 0.7 °C near the outfalls. It had no impact on oxygen saturation, turbidity, pH, nutrients (except for total organic phosphorus (TOP)), chlorophyll-a and metal concentrations. TOP, from the polyphosphonate-based antiscalant discharged with the brine, was correlated with excess salinity.

It is unknown if the results of this short term study represent a steady state, with temporal variability, or the beginning of a slow incremental impact. Israel is planning to more than double desalination along its 190 km Mediterranean coast by 2050. A long term, adaptable, program, in conjunction with specific research and modeling, should be able to assess and predict the impact of large scale brine discharge on the marine environment.

1. Introduction

Desalination is recommended by the United Nations, through the goals of Agenda (2030) for Sustainable Development, as an essential tool to provide clean water and sanitation to the world's population. (https://www.un.org/sustainabledevelopment/development-agenda, accessed December 9, 2019). The increasing need for freshwater and the rapid technological development generated an annual growth rate in desalination of 14–17% per year from 1997 to 2012 (74.8 Mm^3/d). Since 2012 the growth rate decreased but the installed capacity is still increasing: 86.5 Mm^3/d in 2015 and 95.4 Mm^3/d in 2018 (Gude, 2016; IDA, 2018; Lior, 2017). The main global desalination effort (47.5%) is concentrated in the Middle East – North Africa (MENA) region. Desalination produces, in addition to freshwater, brine that needs to be treated or disposed of. A recent synthesis on the global state of desalination suggests that 142 Mm^3/d of brine were being produced in 2018 by close to 16,000 desalination plants (Jones et al., 2019).

Seawater (SW) desalination accounts for 61% of the produced freshwater globally. Brine, originating from SW desalination, is usually discharged at sea either at the shoreline, through open systems, alone or commingled with other discharges, or through submerged marine outfalls (Missimer and Maliva, 2018; Purnama, 2015; Voutchkov, 2011). Brine may include chemicals used during the desalination process that are also discharged at sea. Among them are the coagulants, antiscalants, biocides, neutralized acids and bases used for cleaning the membranes, excess lime used to remineralize the product water (Kress, 2019 and references therein). However, in contrast to the vast number of publications on desalination processes, economics and energy use, that have been growing exponentially since 1980 (Jones et al., 2019), less than 2000 publications addressing environment and desalination were published from 1960 to 2017. Out of them, only 194 addressed the marine environment (Kress, 2019). Most publications specified only the potential, theoretical impacts, without actual observational data. Even now, in 2019, the number of studies presenting actual observed impacts, if any, in the marine environment is lacking.

The Mediterranean coast of Israel is an ideal location to research

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the impact of desalination brine discharge on the marine environment: Five seawater reverse osmosis (SWRO) desalination plants operate along the coastline, four within a 40 km stretch (Fig. 1). The plants are among the largest in the world, producing close to 590 Mm^3/y freshwater, about 80% of the total domestic and industrial needs of Israel. Two plants (Ashkelon and Hadera) are co-located with power stations and dispose of the brine at the shoreline, next to or mixed with the power stations' cooling waters. Three plants (Palmachim, Soreq and Ashdod) discharge the brine through marine outfalls equipped with diffusers at 20 m water depth. All use a submerged intake system for feed water supply (Kress et al., 2017). A tender for the construction and operation of a sixth plant in the Palmachim area, to be the largest SWRO in the world (200 Mm^3/y), was issued in October 2018, and a seventh plant, to be located at the northern shore, is at the planning stages. The Israeli experience on desalination, mainly concerning energy use and carbon footprint, has been recently described (Tal, 2018).

In Israel, effluents, including desalination brine, can be discharged at sea only after a permit is issued by an inter-ministerial committee chaired by the Ministry of Environmental Protection.

As part of the brine discharge permitting process, the plants are required to perform biannual compliance monitoring surveys to assess the effects of brine discharge on the receiving marine environment and report the findings to the Ministry of Environmental Protection. The goal of this study was to characterize the spatial and temporal distribution of the brine plume during real operations, as opposed to modeling results and determine its impact on seawater quality (salinity, temperature, dissolved oxygen, fluorescence, turbidity, dissolved oxygen, nutrients, chlorophyll-a, suspended particulate matter, pH, and metals). The specific aims of the study were: 1) To determine the variability of the brine dispersion, 2) To compare the field and modeling results, providing necessary ground true data for modeling improvement, and 3) To assess the quality of the environmental monitoring and provide tools to regulators to improve brine management. The importance of seawater quality for plant operations is discussed as well. Data from 14 surveys, four prior to brine discharge, serving as the reference conditions and 10 after the start of operations were used to achieve these aims.

2. The desalination plants

The Palmachim and Soreq SWRO plants are located at the southern Mediterranean coast of Israel, ca. 17 km south of Tel-Aviv (Fig. 1). The Palmachim plant started to operate in 2007 with an installed capacity of 30 Mm^3/y that was expanded gradually. Since 2013 the installed capacity is 90 Mm^3/y. Seawater is supplied to the plant through two intake heads located 0.94 km from the shoreline. The bottom depth at the site is 15 m and the intake heads are located 4 m above the bottom. From 2007 up to March 2014, the brine was discharged through a submerged marine outfall (hereafter outfall) equipped with diffusers at 10 m bottom depth (0.6 km from shore) (Kress and Galil, 2012). Since April 2014 the brine has been discharged through a new outfall located at 20 m bottom depth (1.4 km from the shoreline and 0.67 km westwards from the intake). The new submerged outfall is equipped with a diffuser system consisting of 3 discharge ports of 0.8 m diameter, placed 6 m from each other. The ports are pointed towards 315° azimuth, 6 m above the bottom and with a 45-degree angle in relation to the horizon. The average discharge rate is ca. 15,500 m^3/h.

The Soreq plant has been fully operational since September 2013 with an installed capacity of 150 Mm^3/y. Seawater is supplied to the plant through two intake heads located 1.3 km from the shoreline. Bottom depth at the site is 14 m and the intake heads are located 5–8 m above the bottom. Brine is discharged at 20 m bottom depth (1.9 km from the shoreline and 0.6 km westwards from the intake) through an outfall equipped with a diffuser system with 4 discharge ports of 0.8 m diameter, placed 2.5 m from each other. The ports are

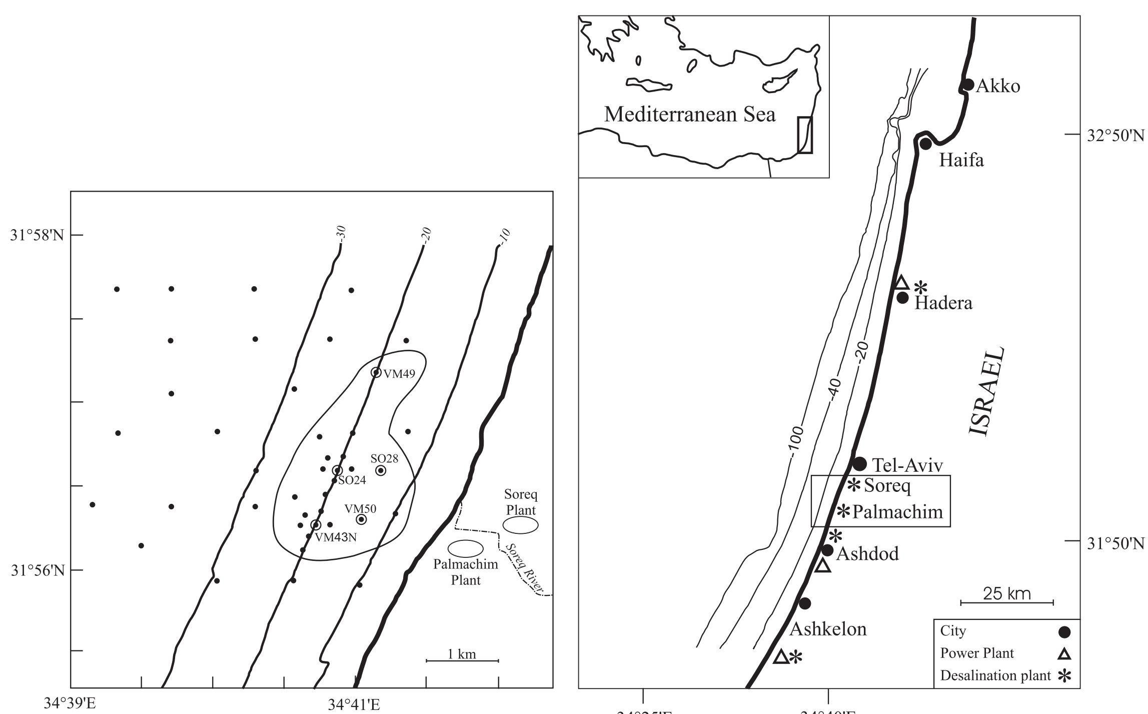

Fig. 1. Desalination plants along the Israeli Mediterranean coast. Inset, map of sampling stations (filled circles) occupied during the May 2017 survey. The Soreq and Palmachim plants intakes (SO28 and VM50, respectively), the plant's outfalls (SO24 and VM43N, respectively) and the reference station VM49 are marked. The distance between the planned and actual occupied station position was ca. 20 m. The area with the stations that were occupied during each survey is encircled. The additional stations varied according to the findings during the survey. Survey-specific stations locations are presented in Fig. S1 (supplementary material).

arranged parallel to the bathymetry, 2 towards the NNE (24° azimuth) and two towards the SSW (204° azimuth) alternately, 4 m above the bottom and with a 45° angle in relation to the horizon. The average discharge rate is ca. 27,700 m^3/h. The outfall is located 0.8 km NNE to the Palmachim plant brine outfall. Since September 2017, electricity for the plant is provided by the new I.P.P. Soreq Delek Ltd power plant, located on site. The plant uses natural gas as fuel and part of the desalination brine (and/or seawater) as cooling waters, that are discharged with the desalination brine. The temperature of the combined discharge is higher than ambient seawater by 3–4 °C.

The desalination process at both plants have a recovery of about 50%, and the brine's salinity (about 80 psu) has about twice the salinity of the seawater. Both the Palmachim and Soreq plants use polyphosphonate based antiscalants, that are discharged at sea with the brine. The Soreq and Palmachim plants discharged 53.2 and 21-ton P in 2018, respectively; and 50.4 and 17.7-ton P in 2017, respectively. Iron salts are used as coagulants in the feedwater pretreatment step only in the Soreq plant. About 90% of the iron is disposed of inland and only 10% is discharged at sea with the brine. The Fe loads discharged at sea in 2018 and 2017 were 0.87 and 0.97 ton, respectively (Kress et al., 2019b).

An Environmental Impact Assessment (EIA) was performed during the planning stages of the plants. It included, among others, modeling of the brine dispersion, and pre-construction marine surveys. Modeling, to optimize the design of the outfalls to achieve maximum dilution, was performed at the coastal and marine engineering research institute (CAMERI) of the Technion, Israel Institute of Technology (Sladkevich et al., 2012, improved in 2015). Near field simulations were performed with the UM3 model from Visual Plume package developed by the United States Environmental Protection Agency (EPA) and the far-field simulations were performed with the CAMERI3D/HD-ST model. Four pre-construction marine surveys were conducted in 2008–2009 and in 2012 (Table 1). The aim of these surveys was to characterize the natural spatial and temporal variability of seawater quality parameters to serve as a reference against which to estimate the impact of the plants at the operational stage.

3. Experimental and methods

3.1. Field sampling

Monitoring cruises were performed twice a year on board the R/V Etziona or the R/V Mediterranean Explorer. A total of 14 surveys were conducted in the area, four prior to the construction of the 20 m depth outfalls, at the EIA stage and 10 at the operational stage (Table 1). The surveys were timed to take place during maximal production of the plants and hence maximal brine discharge, usually at night, when electricity costs are lower. The sampling scheme consisted of 19 planned stations, for discrete seawater sampling, and additional stations to follow the spatial dispersion of the brine plume. The number and location of these additional stations were based on the actual brine dispersion pattern encountered during the survey (Table 1, Fig. 1). Station VM49, located 1.4 and 2.2 km from the Soreq (SO24) and Palmachim (VM43N) outfalls, respectively, towards the north-north west (22° azimuth) was assigned as the reference station (Fig. 1). The distance between the planned and actual occupied station position was ca. 20 m. Based on the monitoring results, the sampling scheme was changed in 2017 to emphasize the near bottom compartment where brine dispersed (Table 1).

During each survey, continuous depth profiles of salinity, temperature, dissolved oxygen, turbidity and seawater fluorescence were acquired at all stations using a SeaBird electronics CTD model SBE 19plusV2. The depth profiles were acquired down to the bottom, until the CTD touched the sediments lightly to prevent resuspension. Although salinity was measured with the CTD as a conductivity ratio and has no physical units (Millero, 1993, 2010), it is reported here with the common unit of psu for clarity.

Surficial seawater samples (0.5 m below the surface) and near bottom seawater samples (0.5 m above the sediment) were sampled at the planned stations using a FLOWJET membrane pump connected to the CTD. Seawater for metal determination by ICP-MS was sampled using a peristaltic pump (Cole-Parmer, a MasterFlex E/S Portable Sampler). Seawater for nutrient analysis were collected into 15-mL acid-washed plastic scintillation vials, immediately frozen (−20 °C) and kept frozen until analysis. Samples for Chlorophyll-a (Chl-a) determination were filtered on board on GF/F filters and immediately frozen. A known volume of seawater (usually 2 L) for duplicate samples for suspended particulate matter (SPM) determination was pre-filtered through a 63 μm plastic sieve to remove large debris and then filtered on pre-weighted 0.45 μm polycarbonate filter and immediately frozen. Seawater for the determination of metals by ICP-MS was sampled into LDPE bottles, acidified to pH 2 and kept in the dark until analysis. Seawater for Hg determination was sampled into plastic vials containing acidified BrCl (EPA Method 1631). Samples were kept refrigerated in the dark until analysis. Samples for pH determination were collected in 50 ml Sarstedt test tubes and kept refrigerated until analysis.

Table 1
Details of the surveys performed at the brine outfall areas of the Palmachim and Soreq desalination plants. Brine has been discharged from the Soreq plant since September 2013 and from the Palmachim plant since April 2014.

Date	Number of stations		Comments
	CTD and seawater sampling	CTD only	
October 2008	7 (surface and near bottom samples)		EIA for 20 m depth outfalls. (Palmachim outfall at 10 m operational)
May 2009			
May 2012			
September 2012			
October 2013	10 (surface and near bottom samples)	4	SO outfall area only
November 2014	19 (surface and near bottom samples)	13	Both outfalls operational
May 2015		11	
September 2015		13	
May 2016		21	
September 2016		19	
May 2017	15 (10 surficial samples and 15 near bottom samples)	21	Change of seawater sampling scheme
October 2017		26	
May 2018		21	
October 2018		21	

3.2. Laboratory analysis

Nutrients (nitrate + nitrite, phosphate, silicic acid, total Nitrogen (TN) and total Phosphorus (TP)) were measured with a Seal Analytical AA-3 system using the methods suggested by the manufacturer and by Kress et al. (2014a). Total Organic Phosphorus (TOP) was calculated by subtracting the phosphate concentration from TP. Chl-a was measured fluorimetrically following extraction with 95% acetone (Holm-Hansen et al., 1965). SPM concentration was calculated after the filters were freeze-dried and re-weighted. One replicate filter was digested with HF in Teflon vials in a MARS 6 CEM microwave for Al, Fe, Mn, Zn, Cu determination by flame atomic absorption spectroscopy (Agilent Technologies 280 FS AA) and Cd, Cr and Pb determination by graphite-furnace atomic absorption spectroscopy (Agilent Technologies GTA 120). Hg was determined in the second replicate by atomic fluorescence (PSA Analytical Merlin Millennium System), following digestion with HNO_3 in the MARS 6 CEM microwave. Metals and Se in seawater samples, except for Hg, were determined by ICP-MS at the Geological Survey of Israel (GSI). Hg was determined by us by atomic fluorescence (PSA Analytical Merlin Millennium System). pH in seawater samples was measured on board with a WTW pH/ion Meter Multi 3430 or in the laboratory with a Radiometer PHM240 pHmeter.

3.3. Calculation and mapping of excess salinity over reference conditions

The reference salinity for each survey was set as the seawater salinity measured near the bottom at the reference station, VM49. Excess salinity over reference (hereafter excess salinity) was calculated as a percentage from the reference salinity for each data point. The calculation, performed for each survey, generated a normalized parameter, factoring out the natural seasonal and inter-annual variability of salinity, and thus enabling comparison among the surveys and to data from elsewhere (Kress et al., 2017).

Depth profiles and dispersion maps of the seawater properties measured were generated with the Ocean Data View Software, (Schlitzer, R., Ocean Data View, odv.awi.de). The Sigma Plot 2002, version 8.02 from SPSS Inc. was used to draw some of the figures.

3.4. Statistical analysis

Numerical summary of the data (number of data points, range, mean, standard deviation of the mean, median, median absolute deviation) (Table S1), and comparisons (*t*-test, Mann-Whitney a-parametric test) were performed using Addinsoft's XLSTAT2013 add-on to the Excel software at the 5% significance level.

4. Results

The desalination brine discharged through the two outfalls was saltier and denser than the ambient seawater, negatively buoyant, and dispersed near the bottom (See section 4.1). Therefore, the results of the discrete measurements, performed at the surface and at the near bottom (nutrients, Chl-a, SPM, metals in seawater and in SPM) were classified by sampling depth and by sampling season (spring, end of summer, fall). Moreover, the near bottom samples were divided into two groups based on the excess salinity: the affected samples, with $\geq$1% excess salinity and the non-affected samples, with <1% excess salinity. The results are summarized in Table S1. The decision to use the 1% excess salinity as the cutoff between affected and non-affected areas was based on modeling results.

4.1. Salinity

During all surveys, surficial seawater salinity ranged from 38.94 to 40.02 psu with slight seasonal and temporal variations (Table S1). Brine was not detected at the surface in any of the surveys. In the absence of brine, bottom salinity ranged from 39.19 to 40.21psu, with slight seasonal and temporal variations (Table S1). The water column was either mixed or slightly stratified, with salinity increasing towards the bottom, the latter at the deeper stations, with no seasonal dependence. In the presence of brine, salinity increased sharply near the bottom, usually with a gradient of 1.5 psu within 2 m. Representative salinity depth profiles at stations with and without brine presence are depicted in Fig. 2A, that emphasizes the sharp salinity increase at the affected stations. Generally, the brine dispersed towards the open sea (west- north west (WNW)), forming a thin saline layer with excess salinity $\geq$1% near the bottom as shown in a representative depth section (Fig. 2B). Similar profiles and sections were observed during all 10 surveys conducted following the start of operations. The maximal extent of the dense thin layer (L) was greater than 4.4 km while the maximal excess salinity observed near the bottom ranged from 4.3 to 9.1% (Table 2, Fig. 2). The near bottom area affected by the brine ranged from 2 km^2 to more than 13 km^2 (Table 2, Figs. 2C and S1). The sampling scheme was detailed enough to delimit the whole affected area only during 3 out of the 10 surveys.

The brine discharged from both outfalls combined to form one area with $\geq$3% excess salinity at some of the surveys while at others, one or two separate areas were found (Table 2, Fig. S1). The maximal area with excess salinity over 3% was found in September 2015 (2.6 km^2, Table 2, Fig. S1). The areas with excess salinity over 5% were small (usually less than 0.1 km^2) and confined to the vicinity of each outfall (Table 2). Excess salinity over 7% near the bottom was detected during 2 surveys only, at the Soreq outfall (Tables 2 and S1). Noteworthy is the fact that the bottom salinity at the reference station in May and October 2018 was higher than expected (by 0.04 psu, 0.1% excess salinity) seemingly due to brine presence. Therefore, a new reference station was set up and used in the surveys conducted in 2018. The new reference station is located 0.5 km north-north east from the previous reference station, at the same water depth.

While brine was usually observed at the 1–2 m layer above the bottom, during 3 surveys, it was detected at the outfall starting at 10 m above the bottom (Fig. 2, Table 2). The maximal excess salinity, 12.8%, was observed in May 2018 at 13 m depth (Table 2). These mid-depth excess salinities were larger than the ones calculated for the bottom layer, probably due to their proximity to the discharge ports (See section 5.1).

4.2. Temperature

Seawater temperatures were natural for the area, with seasonal and temporal variations as expected (Table S1). Depth profiles showed usually a mixed water column, or slight stratification, with temperature decreasing with increasing water depth. Only within 200 m from the outfalls, in the presence of brine, seawater temperature was higher than ambient by 0.3–0.7 °C (Fig. 2). Unusual depth profiles of temperature, mirroring salinity, occurred at the outfall stations when brine was detected at mid-depths (Fig. 2).

Since October 2017, the temperatures near the Soreq outfall were slightly higher (by 0.3 °C) than those measured at the Palmachim outfall at the same salinity (Fig. 2D). This is due to the co-discharge of the Soreq brine with cooling waters following the start of operations of the adjacent power plant (see section 2). This difference did not exist prior to the operation of the power plant (Kress et al., 2019b).

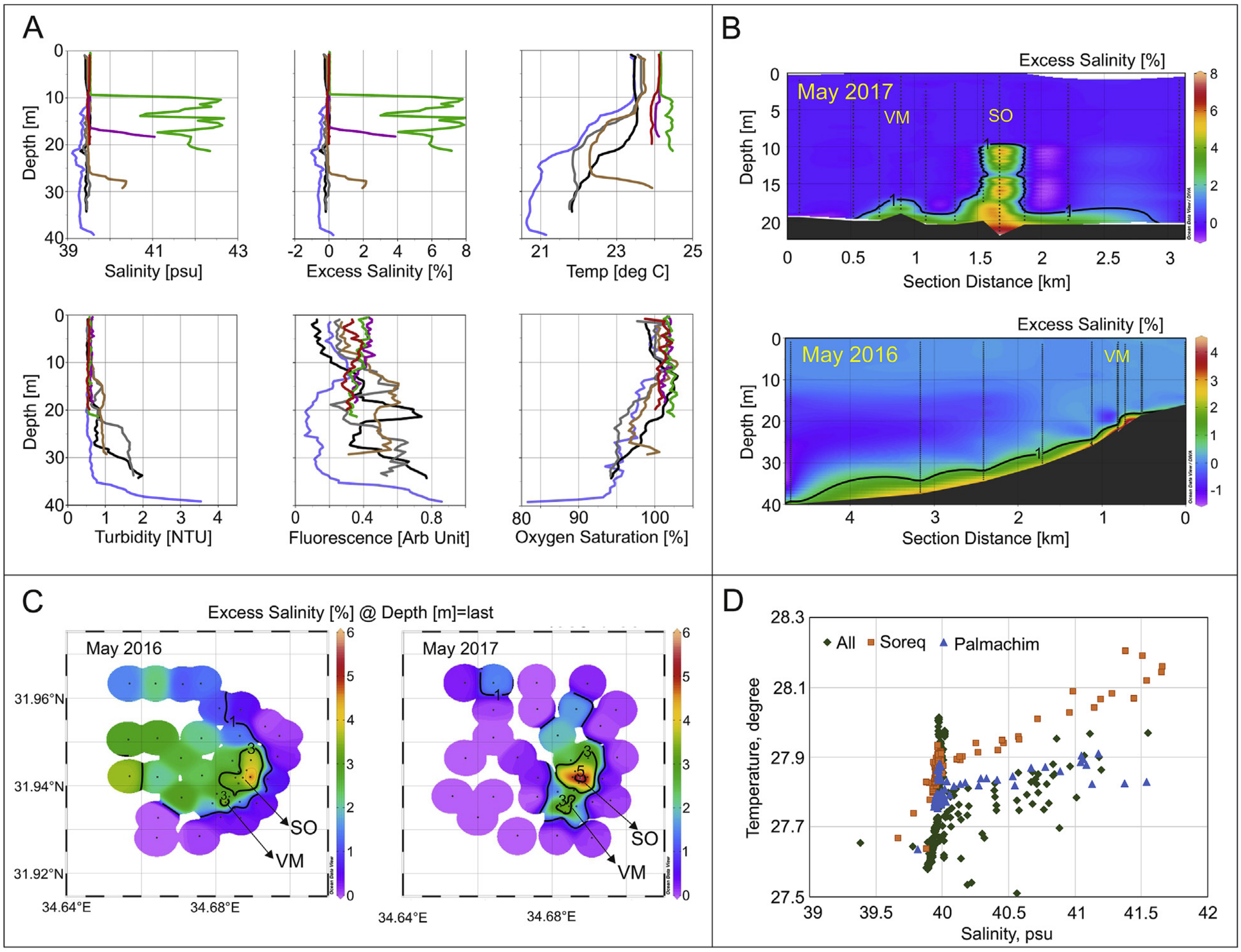

Fig. 2. Depth profiles, depth sections, brine dispersion near the bottom and TS diagram at representative stations and surveys. The Soreq and Palmachim outfalls are abbreviated as SO and VM, respectively.

A. Depth profiles of salinity, excess salinity, temperature, turbidity, fluorescence and dissolved oxygen saturation at selected stations (May 2017). SO and VM outfalls in green and purple, respectively; reference station in red. Other colors, typical stations.

B. Depth sections of excess salinity through the outfalls (May 2017) and westwards towards the open sea (May 2016).

C. Excess salinity in seawater near the bottom in May 2016 and 2017. Points denote the sampling stations (Fig. 1) and the right panel, the color coded scale of excess salinity. The near bottom dispersions for all the surveys are presented in Fig. S1.

D. Temperature vs Salinity (TS) diagram for the October 2018 survey. Orange squares - Soreq outfall and nearby stations; Blue triangles - Palmachim outfall and nearby stations; Green diamonds, other stations.

Table 2

Extent of the brine affected areas, maximal excess salinity and maximal plume size (L, the distance between two maximally remote points of the plume) found during the 10 surveys following plant operations. Modeling results for the worst case scenario (calm seas) are also shown. Brine discharge through the 20 m depth outfalls started in September 2013 (Soreq, SO) and April 2014 (Palmachim, VM).

Excess salinity	Area (km^2)				Maximal excess salinity (%)	L, m (1%)	L, m (5%)
	≥1%	≥3%	≥5%	≥7%			
Model simulation (CAMERI)	3.5	0.5	0.01 (VM) 0.02 (SO)	0.002(VM)		2395	189 (VM) 286 (SO)
October 2013[&]	ND	>0.3	<0.1	0	5.7	ND	400
November 2014[#]	3	1	0.25 (SO)	0.07 (SO)	9.1	>1400	500
May 2015	>2	1.1 (SO)	<0.1 (SO)	0	6.1	>2370	ND
September 2015	>5	2.6 (SO), 0.2 (VM)	<0.1(VM)	0	5.1	>3530	ND
May–June 2016	>13	1.3	<0.1 (SO)	0	5.2	4300	330
September 2016	>7	0.4	<0.1 (SO)	0	5.7	4000	~200
May 2017	4	0.7 (SO), <0.1(VM)	<0.1 (SO)	Localized (SO)	7.2 (8.0*, SO-14.5 m)	3167	~200
October 2017	5	0.1 (SO), <0.1(VM)	0	0	4.7 (7.0*, VM – 10 m)	1769	0
May 2018	>10	1.3 (VM), Localized (SO)	0	0	4.8 (12.8*,SO- 13 m)	>4300	0
October 2018	>12	0.7	0	0	4.3	>4400	0

*Mid water column, at discharge port, [&]Only the Soreq outfall operational, [#]Plants operating and 75–80% full capacity. ND – could not be determined.

4.3. Dissolved oxygen, turbidity and fluorescence

Representative depth profiles of dissolved oxygen, turbidity and fluorescence are depicted in Fig. 2. Brine did not affect these seawater quality parameters. Surficial seawater was saturated with dissolved oxygen, with similar or slightly lower saturation near the bottom, as expected. Turbidity usually increased slightly near the bottom at all stations, unrelated to the presence of brine. Fluorescence depth profiles, a proxy of Chl-a concentration, were seasonally and temporally dependent.

4.4. Nutrients, Chl-a, SPM and pH

Summary statistics of nutrients, Chl-a and SPM concentrations are presented in Table S1. Brine presence did not affect the concentrations of the nitrogen (N) species, silicic acid and phosphate. Surface and near bottom concentrations of NO_X, NH_4 and TN were similar (average values of 0.22 ± 0.38 μM, 0.24 ± 0.28 μM, and 6.50 ± 1.01 μM, respectively), with no seasonal nor temporal dependence. No differences were found among the silicic acid concentrations in the surface and near bottom samples in the spring nor in the summer, however the concentrations in the spring were lower than in the summer (1.55 ± 0.74 μM and 2.09 ± 0.94 μM, respectively, $p < 0.0001$).

Phosphate concentrations in the spring were lower than the summer concentrations, both at the surface (0.029 ± 0.023 μM and 0.061 ± 0.046 μM in spring and summer, respectively, ($p < 0.0001$)) and near the bottom (0.047 ± 0.032 μM and 0.077 ± 0.056 μM in spring and summer, respectively ($p < 0.0001$)). Phosphate concentrations at the surface in the spring were lower than the bottom concentrations ($p < 0.0001$) while in the summer, the concentrations were similar at both sampling depths. There was a large temporal variability in the concentrations but no specific trend.

Brine affected only the concentrations of TP. In the presence of brine, the concentrations of TP near the bottom (average of 0.27 ± 0.12 μM in the spring and of 0.33 ± 0.17 μM in the summer) were more than twice of those found at the near bottom samples without brine (Table S1). As phosphate concentrations were not affected by the brine presence, this increase in TP is attributed to the presence of total organic P (TOP). TOP probably originated from the polyphosphonate-based antiscalants used by both desalination plants and discharged at sea with the brine (See section 2). Moreover, TOP correlated significantly to excess salinity (Fig. 3). The TP concentrations found at the surface and near the bottom in the absence of brine were similar, and no differences were found between spring and summer.

Chl-a concentrations during the surveys ranged from 0.07 to 1.65 μg/L, exhibiting a high temporal variability and well as seasonality. The concentrations in the spring were lower than in the summer, in particular at the surface (Table S1). Both in the spring and summer, the Chl-a concentrations at the surface were lower than the concentrations measured near the bottom. Chl-a concentrations were not affected by brine presence.

SPM concentrations were natural for the area, ranging from 0.38 to 2.95 mg/L at the surface, and slightly higher at the bottom as expected due to the proximity to the sediments (maximal value of 4.91 mg/L). No seasonal differences nor brine impact on SPM concentrations were detected (Table S1).

Seawater pH were natural for seawater, 8.26 ± 0.08. No seasonal differences nor brine impact were detected.

4.5. Metals in seawater

Metals in seawater were measured in two different ways: directly in a seawater sample by ICP-MS (except during the 2018 surveys) and in SPM collected from a known volume of seawater during all surveys. The metals analyzed were chosen based on existing Israeli guidelines for seawater quality (Table 3). Special emphasis was given to Fe, used as a coagulant in the pre-treatment stage at the Soreq plant and to Cu and Zn, reported to accumulate at brine discharge sites of thermal desalination plants ie (Sadiq, 2002; Saeed et al., 2017). Although not expected to be of concern with RO plants, metals determination was included in the monitoring program as a precautionary measure.

All metals concentrations measured directly in seawater, excluding Hg and Al, were below the detection limits, including the samples with brine (Table 3). Hg concentrations were very low, ranging from <5 to 48 ng L^{-1} and Al ranged from 5 to 50 μg/L. All concentrations were below the proposed Israeli water quality guidelines. No correlation to the presence of brine, nor spatial or temporal variations in the concentrations were found.

The concentration ranges for metals in seawater, calculated from the SPM analysis, are given in Table 3. No correlation was found between metal concentrations and brine presence in the samples, only seasonal and temporal variations were detected. This is shown by the linear correlation of Fe and Cu plotted as a function of Al (Fig. 4), the latter normalizing for seasonal and temporal variations (Herut and Sandler, 2006). Some outliers were identified, in particular for Cu, with no temporal nor spatial pattern.

5. Discussion

Brine discharge did not impair the surficial seawater. Salinity and temperature were as expected for the area and exhibited natural seasonal and temporal variability. The concentrations of nutrients, Chl-a, and SPM were in agreement with the reference values derived for the area (Kress et al., 2019). Brine affected the water quality mainly near the bottom (2 m layer) but from the middle of the water column (starting at about 10 m depth) in the close vicinity of the discharge ports. Therefore, the following discussion will concentrate on the near bottom seawater quality.

5.1. Variability of excess salinity and of the spatial extent of brine presence

Brine dispersion in the marine environment is highly dependent on the site's geographical settings, hydrographical conditions, desalination technology, plant production, mode of discharge and on the final brine density following discharge (Kress, 2019 and references therein). Near field mixing is governed by the specifics of the discharge system, while far field dispersion is highly dependent on the hydrographic conditions (Missimer et al., 2015). Everything being equal, as in this study, the dispersion pattern observed during a specific survey will depend on the actual hydrographic conditions encountered during it and on the rate of brine discharge (plant operations) during the survey. Quiet seas will reduce mixing and dispersion, while rough seas will increase it. In this study, the surveys took place during calm seas (less than 1 m wave height with winds up to 20 knots), but the conditions were not identical. Moreover, the maximal production of the plants occurs during the night, when electricity costs are lower.

Seawater salinity exactly at the outfall was measured during 3 surveys (Table 2). Brine impacted the water column starting from 10 m water depth at the outfall stations, causing a "noisy" depth profile of salinity, temperature and dissolved oxygen, probably due to sampling at the brine jets and not in the spreading brine layer. The maximal excess salinity, 12.8%, was observed at 13 m water depth at the Soreq outfall in May 2018 (Fig. 2, Table 2). Although observed during only 3 surveys, similar depth profiles are assumed to occur near the outfalls, and only the difficulty to reach the exact

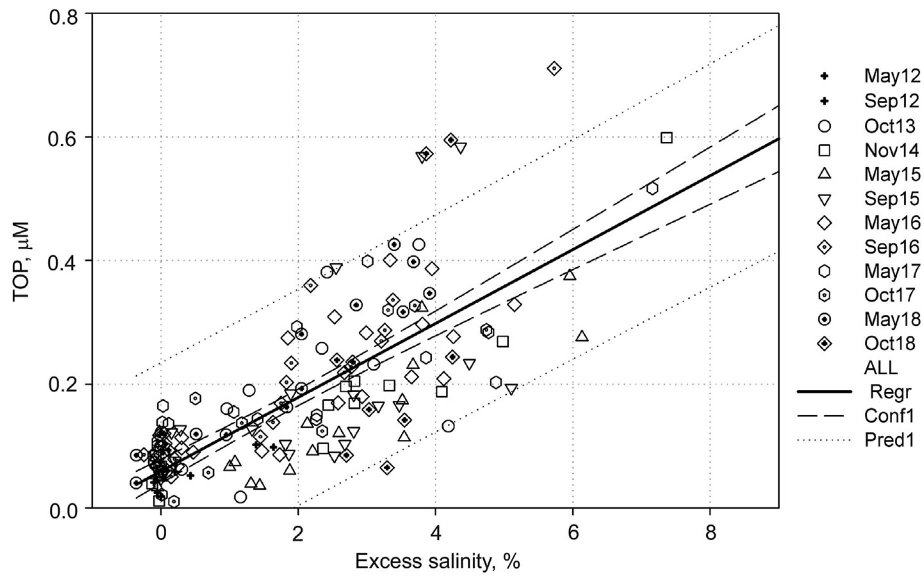

Fig. 3. TOP vs excess salinity at the near bottom samples from all surveys. The regression line is drawn as a solid line ($r^2 = 0.6056$). The confidence intervals (dashed line) and the predicted intervals (dotted line) were calculated at the 95% level.

Table 3
Ranges of metal concentration in seawater at the study area measured from 2013 to 2018. The proposed Israeli guidelines for seawater quality are presented as well.

	Hg	As	Se	Ag	Cd	Cu	Pb	Cr	V	Al	Mn	Fe	Ni	Zn
	ng/L	μg/L												
ICP-MS	<5-48	<7	<7	<0.1	<0.1	<1	≤0.1	<10	<10	<5-50	≤1	<50	≤1	<2
SPM														
Min	<0.01	NM	NM	NM	<0.0001	<0.001	<0.01	<0.01	NM	3.3	0.02	2.3	NM	<0.02
Max	35				0.09	0.48	0.67	2.10		335	8.93	290		7.37
Israel's seawater quality guidelines														
Average	160	36	60	3	0.5	5	5	10	50				10	40
Maximal	400	69	150	7	2	10	20	20	100				50	100

NM- Not measured.

discharge point during the cruises prevented us from detecting it in all surveys. The initial mixing was aided by the orientation of the diffuser heads taking advantage of the predominant alongshore currents (Rosentraub and Brenner, 2007). Following this initial mixing, the brine sank and dispersed near the bottom towards the west and north-west (Figs. 2C and S1). The design of the diffuser systems prevented dispersion eastwards towards the coast and the intake heads, except for about 200 m mainly from the Soreq outfall.

The extent of the near bottom area affected by the brine was highly variable with no seasonal nor temporal variations (Table 2): the size of the area with ≥1% excess salinity ranged from 2 km^2 to more than 13 km^2 and the area with ≥3% excess salinity ranged from about 0.1 to 2.6 km^2. Areas with excess salinity ≥5% were either confined to the vicinity of the outfalls (<0.1 km^2) or absent, with maximal plume size (L, Table 2) of 500 m. Although variable in extent and position, the impact of the brine on seawater salinity was detected during all surveys, with similar maximal excess salinity (Table 2). In most surveys, the higher excess salinity was measured in the vicinity of the Soreq outfall compared to the Palmachim outfall, probably due to the1.7 times larger production capacity of the Soreq plant.

The brine plume dispersing near the bottom started usually as a 2 m deep layer, narrowing with increased distance from the outfalls (Fig. 2B). When the area exactly at the discharge ports was sampled, the vertical width of the brine plume was ca. 10 m. The brine formed a density current towards the open sea, its full extent not characterized yet. The bottom at the study area is homogenous with a gentle slope towards the open sea, with no known bathymetric barriers (such as rock outcrops or reefs) that may block the dispersion of the brine and cause its accumulation near the outfall. The maximal length of the area with excess salinity ≥1% was greater than 4.4 km. For comparison, when the Palmachim plant discharged the brine at 10 m depth, excess salinity ≥1% was observed up to 1 km only from the outfall. However, the maximal excess salinity reached 15%, when the production was at 90 Mm^3/y (Kress and Galil, 2012; Kress et al., 2014b).

Comparison among the modeling results and the field observations showed that the simulated area with excess salinity ≥1% was usually smaller than the actual observations (Table 2, Fig. S1). The same is true for the size of the area with >3% excess salinity, that was common to both outfalls in the model, but observed at times as two separate areas around the outfalls. Both the model and the in situ observations showed a small, usually <0.1 km^2, area with ≥5% excess salinity. A similar underestimation of the model was found for the Ashdod SWRO plant, where the simulated area with >1% excess salinity was smaller by factor of 19 and 2 in the spring and fall surveys conducted in 2018 (Shoham-Frider et al., 2019).

An additional modeling of the brine dispersion in the area, using the MIKE3-HD hydrodynamic (Biton et al., 2019) found that the maximal plume size in which excess salinity exceeded 1% was confined to 2 km, similar to value found by the CAMERI simulation and smaller than the lengths observed in the field. The brine

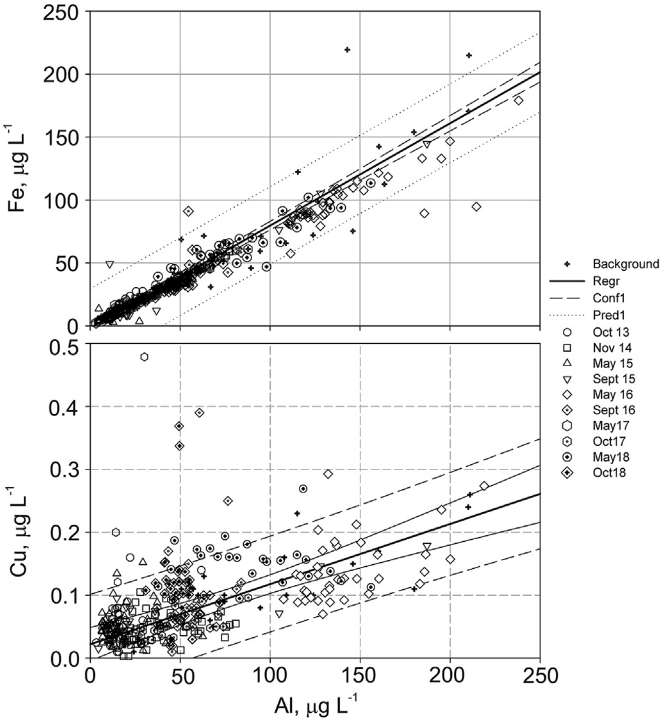

Fig. 4. Fe and Cu as a function of Al in seawater as determined in SPM samples. The linear regression line (solid black) was drawn using background data collected during the EIA study. The confidence intervals (dashed line) and the prediction intervals (dotted line) were calculated at the 95% confidence level. Two data points with Al concentration of 328 and 335 µg/L are not presented in the figure for the sake of clarity.

dispersion in the winter was downslope towards the open sea while in the summer, the simulation showed a propagation along the coastline. Excess salinity of 0.01% was simulated to cover a large part of the coast. Although not significant ecologically, they speculated that brine presence may impact coastal water dynamics.

5.2. Comparison to other areas and to environmental guidelines

In general, field studies on SWRO plants brine discharge found excess salinity mostly in the 1–7% range, although at some instances much higher values were reported. Brine was detected up to 5 km from the discharge site, but most studies reported a return to ambient salinity within tens to a few hundred meters from the discharge site (Kress, 2019). As expected, large plants discharging brine with low initial dilution had a larger impact on salinity than smaller plants or plants discharging brine through optimized outfalls. For example, hypersalinity (3% maximal excess salinity) was measured for a few kilometers from the outfall of the San Pedro del Pinatar SWRO plant (Spain) prior to the installation of a diffuser while no increase in salinity was detected following its installation (Fernandez-Torquemada et al., 2009). In Algeria, brine from the Mostaganem plant, discharged through a submerged outfall equipped with a diffuser, increased seawater salinity by up to 9% and brine was detected up to 200 m from the outfall. At the Beni Saf plant (Algeria), with the same production capacity and brine discharge at the same water depth without a diffuser, the maximal excess salinity measured was 72% and brine was detected up to 1.5 km from the outfall (Belatoui et al., 2017). SWRO brine, co-mingled with cooling waters from power stations is discharged at the shoreline, through open outfalls, at Hadera (Israel) and Carlsbad, CA (USA), among others. At Hadera, the brine plume was observed up to 2.5 km from the outfall and the maximal excess salinity measured was 10% while in Carlsbad, the brine plume was observed up to 1 km from the discharge and the maximal excess salinity observed was 13% (Petersen et al., 2019; Shpir and Ben Yosef, 2017a). Table 4 compiles additional examples from the literature.

In Israel there are no guidelines for the permissible increase of salinity at the discharge site nor on the compliance point distance (Safrai and Zask, 2008). Globally there are a few, usually requiring less than 5% excess salinity at the edge of the regulatory mixing zone (50–300 m from the discharge) (Jenkins et al., 2012; Uddin et al., 2011; Viskovich et al., 2014). The California Ocean Plan (2015) limits the increases of salinity to two psu over ambient (about 6% excess salinity) within 100 m from the discharge point. Brine discharge near sensitive areas, such as near *Posidonia oceanica* seagrass mats in Spain, may elicit stricter guidelines (Fuentes-Bargues, 2014; Palomar and Losada, 2010). Comparison of the results of this study to the prevalent 5% excess salinity guideline showed that this guideline was exceeded during all surveys except one (October 2018, Table 2). However, the exceedance area was small, mostly <0.1 km^2. Only during the November 2014 survey the impacted area was 0.25 km^2 and the maximal plume size 0.5 km (Table 2). Therefore, it can be carefully concluded that the excess salinity found in this study is mostly within the guidelines proposed elsewhere and that the design of the outfall systems is efficient for diluting the brine. However, the extent of the brine dispersion should be carefully followed, in particular as brine was present at the reference station in 2018.

5.3. Seawater quality

In addition to salinity increase and hence changes in seawater density, it has been generally hypothesized that brine discharge could affect seawater quality by: increasing temperature, reducing oxygen solubility and concentration, increasing turbidity, changing pH, increasing the ambient concentration of chemicals discharged with the brine originating from the desalination process (coagulants, antiscalants, biocides, cleaning solutions) and metals from plant corrosion (Kress, 2019). Changes in seawater quality may in turn affect the biota in general, and the pelagic microbial communities in particular (Belkin et al., 2018 and references therein) and affect Chl-a concentrations. Almost none were found in this study. The results showed that brine had no effect on oxygen concentration and saturation, nor on turbidity, pH, nutrients (except for TOP and hence TP), Chl-a and metal's concentration in seawater.

Brine increased the seawater ambient temperature by up to 1 °C at a limited, small area around the outfalls. This slight increase in temperature is not expected to influence the biota directly but it may give a natural edge to the establishment of non-indigenous species (NIS) in the presence of saline brine. The Israeli coast is known for its large amount of Erythraean NIS introduced through the Suez Canal. Most of the NIS identified in the area of study are the macrobiota (fish, crustaceans, polychaeta), while smaller size taxa, as the phytoplankton and bacteria, that could influence the parameters studied here, are largely unrecognized and undetected (Galil et al., 2018; Zenetos et al., 2005).

Brine increased also the concentration of TOP in seawater compared to the reference values (Fig. 3). This increase is attributed to the presence the polyphosphonate-based antiscalants used by both desalination plants and discharged at sea with the brine. Fig. 3 depicts the relationship of TOP to excess salinity. To the best of our knowledge this is the first time that such a correlation has been shown except along the Israeli coast (Kress, 2019; Kress et al., 2017). A similar correlation can be found also at the other desalination plants operating along the Israeli Mediterranean coast that also use

Table 4
Excess salinity reported in *in situ* studies at the outfalls of Seawater Reverse Osmosis (SWRO) desalination plants. NR – not reported, outfall (submerged outfall system), MCWPP – mixed with cooling waters from power plant, D-diffuser. The actual production during the study is given as the percentage from the total capacity when different from it and if reported.

Location or Plant name	Start Operations	Total capacity (Actual Production)	Discharge			Ambient Salinity	Impact		Reference
			Method	Distance from the shoreline	Bottom depth		Maximal excess salinity	Maximal distance	
		m^3/day (%)		km	m	psu	%	m	
KAUST (Kingdom of Saudi Arabia), Red Sea	NR	40,000 (50%)	Outfall	2.8	18	40.9	15	NR	van der Merwe et al., (2014)
Platja de Mitjorn, Formentera Island (Spain)	1985	2000 (50%)	Outfall	0.02	0.9	37.5	11	50	Gacia et al., (2007)
Javea, Alicante (Spain)	2002	28,000 (25–50%)	Open, with pre-dilution	0		37	13–16	300	Fernandez-Torquemada et al. (2009)
Alicante I (Spain)	2003	68,000 (74–88%)	Open, with pre-dilution	0		37	8	2000; 500 when diluted	Fernandez-Torquemada et al., (2009)
Alicante II (Spain)	2009	65,000	Open, with pre-dilution	0		37.7 –38.3	2.6–5.3	750	Garrote-Moreno et al., (2014)
New Channel of Cartagena, San Pedro del Pinatar, Murcia (Spain) 2 plants	2005 –2006	134,000 (see references for details)	Outfall, with D since 2010	5	38	37	3	800- few km, 0 (after D)	Fernandez-Torquemada et al. (2009)
				5	38	37	6.8–32	250, 0 (after D)	Del-Pilar-Ruso et al. (2015) de-la-Ossa-Carretero et al. (2016)
Mostaganem (Algeria)	2011	200,000	Outfall, with D	1.4	8	36.5	9	200	Belatoui et al. (2017)
Beni Saf (Algeria)	2009	200,000	Outfall	0.5	8	36.5	72	1500	Belatoui et al. (2017)
Bou Ismail (Algeria)	2004	5000	NR			38–39	10–13	NR	(Belkacem et al. 2016, 2017)
Bousfer, Oran Bay (Algeria)	ND	5500	Open channel	0		35.2	24	NR	Benaissa et al. (2017)
Ashqelon (Israel)	2005	329,000	Open MCWPP	0		39.5	10	3000	Shpir and Ben Yosef (2017b)
Palmachim (Israel)	2007	82,000–247,000	Outfall with D	0.6	10	39	15	1000	Kress and Galil (2012)
Palmachim and Soreq (Israel)	2007 2013	247,000 411,000	Two Outfalls with D	1.9	20	39	12.8	<4400	(Kress et al., 2016, 2017) This study
Hadera (Israel)	2010	348,000	Open MCWPP	0		39	6	2500	Shpir and Ben Yosef (2017a)
Ashdod (Israel)	2016	274,000	Outfall with D	1.8	22	39	8.4	5500	Shoham-Frider et al. (2019)
Perth, Kwinana, Cockburn Sound (Australia)	2006	143,700	Outfall with D	0.5	10	33–37	3	350	(Bonnelye et al., 2017; Holloway, 2009)
Gold Coast, Tugun (Australia)	2009	133,000 (57%, 28%, 7%)	Outfall with D	1.2	NR	37	0	0	Viskovich et al. (2014)
Adelaide, Gulf of St. Vincent (Australia)	2011	274,000	Outfall with D	NR	20	35.9 –37.4	1.5–3	100	(Ayala et al., 2015; Kämpf and Clarke, 2013)
Southern, Binningup (Australia)	2012	274,000	Outfall with D	0.75	10	36	1.4	50	Anon (2017)
Penghu County (Taiwan), two plants	2003, 2008	15,500 trial	Outfall	3	NR	34	Somewhat increased		Lin et al., (2013)
Maspalomas II, Gran Canaria (Spain)	1988	25,000	Outfall, with D since 2011	0.3	4	36.8	6.5, <3 with D	700	Portillo et al. (2014)
Carlsbad, California (USA)	2016	180,000	Open MCWPP	0		33.2	13	1000	Petersen et al. (2019)

NR – not reported, MCWPP – mixed with cooling waters from power plant, D-diffuser. Actual production during the study, given as the percentage from total capacity.

polyphosphonate based antiscalants in the desalination process (Shoham-Frider et al., 2019; Shpir and Ben Yosef, 2017b).

Higher TOP, a possible P source, did not increase the phytoplankton community, as indicated by Chl-a concentrations, a proxy for phytoplankton. This could be due to the existing natural excess of PO_4–P over N availability in the area (Kress et al., 2019a; Rahav et al., 2018). The average N:P ratio computed from all samples in this study was 6.7 ± 12.6 (median 2.7) and similar, 5.4 ± 8.6 (median 3.3), in the samples with excess salinity ≥1% (Table S1). These ratios are much lower than the Redfield ratio of 16:1 found in phytoplankton (Redfield et al., 1963), indicating N limitation of productivity in the area. An additional reason may be that the P in the phosphonate bond (C–P) is not bioavailable to the phytoplankton although it has been hypothesized that *Prochlorococcus*, the globally important marine primary producer may have the ability to utilize it (Feingersch et al., 2012).

The pelagic microbial community was not part of the monitoring program. However, in a related research, brine discharge was shown in situ to change the relative composition of the community at the Mediterranean coast of Israel (Belkin et al., 2017). Moreover, at the shallow Palmachim outfall (10 m depth), operational until March 2014, autotroph productivity was reduced by 30% in the summer compared to reference conditions. At the Ashqelon open, shoreline outfall, phytoplankton and heterotrophic bacteria biomass were reduced, while productivity fluctuated between seasons. An earlier study at the same site found a reduction in phytoplankton densities and their productivity per Chl-a, when iron salts, used as coagulants, were still discharged in pulses with the continuous brine discharge (Drami et al., 2011). In contrast, no impact of brine discharge was found in microbial abundance in the Red Sea (van der Merwe et al., 2014).

5.4. Plant operations and seawater quality

Seawater quality, in spite of its importance to desalination plant operations, is seldom addressed in environmental studies. Only harmful algal blooms (HAB) have been extensively studied as they reduce production causing even temporary plant shutdowns (UNESCO and IOC, 2017). News services have reported on jelly fish clogging intakes and reducing freshwater production (https://www.zavit.org.il/en/uncategorized/jellyfish-outbreaks-might-cost-israel-millions-of-euros-each-year/ for example, accessed December 9, 2019). In this study, brine was not detected at the intake heads, what could have increased the feedwater salinity and thus energy costs for desalination. The engineering design of the discharge systems was able to prevent it. However, occasional polluted riverine discharge from the Soreq river, its mouth located opposite the intakes, and pollution from the Ashdod port (Fig. 1) have caused problems for plant operations. On March 2016, poorly treated domestic effluents discharged to the Soreq river and subsequently to the sea initiated a bloom of the diatom *Asterionellopsis glacialis* (E. Rahav, unpublished results) and the Palmachim plant was temporarily closed by order of the Ministry of Health (A. Hermoni, CEO Palmachim plant, personal communication). On May 2017, both the Soreq and Palmachim plants temporarily stopped operations due to oil pollution origination from the Ashdod Port.

5.5. Conclusions and implications for environmental management

Seawater desalination will continue to increase globally and in Israel. In 2017, seawater desalination in the Mediterranean supplied 582 Mm^3/y freshwater out of the 750 Mm^3/y planned to be provided by 2020. In 2012, the long term master plan for the national water section was redrawn and the goal is to provide 1750 Mm^3/y of freshwater by 2050 (92% of the forecasted need for urban and industrial use). (http://www.water.gov.il/Hebrew/Planning-and-Development/Planning/MasterPlan/DocLib4/MasterPlan-en-v.4.pdf, accessed December 9, 2019).

The increase in desalination effort will in turn increase marine discharge of brine in the already replete Mediterranean Sea coast. The results of this study, describing the seawater quality at the outfalls of two adjacent, mega-sized SWRO plants, can serve as a basis for future management and regulatory acts. The results showed that the models underestimated the extent of brine dispersion. Brine discharge did not impair the seawater quality, except for a highly variable area near the bottom with excess salinity $\geq$1%, ranging from 2 to >13 km^2 with a plume size from <1.4 to >4.4 km. Seawater temperature was elevated near the outfalls and TOP higher than ambient in the presence of brine. However, this is a short term study encompassing 6 years since brine discharge started. It is still unknown if the results of the study represent a steady state, with temporal variability, or the beginning of a slow incremental impact. One evidence to the expansion of brine presence in the area may be its presence at the reference station in the 2018 surveys. Marine monitoring should continue for as long as the plants are operational and its findings critically reviewed. The monitoring program should be re-evaluated periodically for its frequency, sampling stations and parameters measured, and adapted when necessary. Based on the results of this study, primary and bacterial production would be a useful additional biological parameter to characterize seawater quality. Moreover, regulators should revise the environmental requirements in light of the technological advances in the desalination industry. For example, phosphorus discharge to the marine environment could be reduced or stopped by: 1) improving pretreatment of seawater and thus reducing the use of the polyphosphonate-based antiscalant; 2) replacing it with green antiscalants, biodegradable compounds that do not include phosphorus; 3) using chemical-free methodology (Dayarathne et al., 2019; Giwa et al., 2017; Pervov et al., 2017) among others. Excess salinity could be decreased by reducing brine discharge implementing: 1) hybrid processes; 2) new or improved membranes; 3) zero liquid discharge and water and salts recovery from the brine (Amy et al., 2017; Buonomenna, 2013; Tong and Elimelech, 2016) among others. Clearly, implementing changes in the established desalination processes is a difficult task, in particular for mega size desalination plants. However, finding ways to implement new technologies to promote sustainability should be a goal for the fast growing desalination industry.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: The monitoring surveys were funded by the Soreq and the Palmachim desalination plants as part of their requirements to be issued a permit by the government for the marine discharge of desalination brine. The data presented in the manuscript were reported by us in compliance monitoring reports, written in Hebrew, and submitted yearly to the desalination plants and to the Israel Ministry for Environmental Protection.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.watres.2019.115402.

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