ICES VIEWPOINT BACKGROUND DOCUMENT: IMPACT FROM EXHAUST GAS CLEANING SYSTEMS (SCRUBBERS) ON THE MARINE ENVIRONMENT (AD HOC)

VOLUME 2 | ISSUE 86

ICES SCIENTIFIC REPORTS

RAPPORTS SCIENTIFIQUES DU CIEM

ICES INTERNATIONAL COUNCIL FOR THE EXPLORATION OF THE SEA

CIEM COUNSEIL INTERNATIONAL POUR L'EXPLORATION DE LA MER

International Council for the Exploration of the Sea
Conseil International pour l'Exploration de la Mer

H.C. Andersens Boulevard 44-46
DK-1553 Copenhagen V
Denmark
Telephone (+45) 33 38 67 00
Telefax (+45) 33 93 42 15
www.ices.dk
info@ices.dk

ISSN number: 2618-1371 I

ICES Scientific Reports

Volume 2 | Issue 86

ICES VIEWPOINT BACKGROUND DOCUMENT: IMPACT FROM EXHAUST GAS CLEANING SYSTEMS (SCRUBBERS) ON THE MARINE ENVIRONMENT (AD HOC)

Recommended format for purpose of citation:

Hassellöv, I.M., Koski, M., Broeg, K., Marin-Enriquez, O., Tronczynski, J., Dulière, V., Murray, C., Bailey, S., Redfern, J., de Jong, K., Ponzevera, E., Belzunce-Segarra, M.J., Mason, C., Iacarella, J.C., Lyons, B., Fernandes, J.A. and Parmentier, K. 2020. ICES Viewpoint background document: Impact from exhaust gas cleaning systems (scrubbers) on the marine environment (Ad hoc).
ICES Scientific Reports. 2:86. 40 pp. http://doi.org/10.17895/ices.pub.7487

Authors

Ida-Maja Hassellöv • Marja Koski • Katja Broeg • Octavio Marin-Enriquez • Jacek Tronczynski • Valérie Dulière • Cathryn Murray • Sarah Bailey • Jessica Redfern • Karen de Jong • Emmanuel Ponzevera • Maria Jesus Belzunce-Segarra • Claire Mason • Josephine C. Iacarella • Brett Lyons •
Josean A. Fernandes • Koen Parmentier.

ICES International Council for the Exploration of the Sea
CIEM Conseil International pour l'Exploration de la Mer

Contents

i Executive summary

Shipping is a diverse industry that connects the world. The distribution and intensity of commercial shipping is increasing and there is a growing need to assess and mitigate the impacts of vessel activities on the marine environment.

New global standards on sulphur content in marine fuels have led to an increasing number of ships installing exhaust gas cleaning systems (EGCS), also known as scrubbers, to reduce their emissions of sulphur oxides to the atmosphere. Ships equipped with a scrubber can continue to use heavy fuel oil, and the process results in discharges of large volumes of acidified water that contain a mix of contaminants, such as heavy metals, polycyclic aromatic hydrocarbons (PAHs), oil residues, and nitrates. For the most common type of scrubber, open loop, this polluted water is directly discharged back to the sea, trading reductions in air pollution for increased water pollution. The scrubber discharge mixture has demonstrated toxic effects in laboratory studies, causing immediate mortality in plankton and exhibiting negative synergistic effects. The substances found in scrubber discharge water are likely to have further impacts in the marine environment through bioaccumulation, acidification and eutrophication. The impacts of scrubber discharge water can be completely avoided through the use of alternative fuels, such as distilled low sulphur fuels. Distilled fuels have the added benefit that they remove the threat of heavy fuel oil spills from shipping activities. If the use of alternative fuels is not adopted, and scrubbers continue to be considered an equivalent method to meet the sulphur emissions limits, then there is urgent need for:

1) significant investment in technological advances and port reception facilities to allow zero discharge closed loop scrubber systems;
2) improved protocols and standards for measuring, monitoring and reporting on scrubber discharge water acidity and pollutants;
3) evidence-based regulations on scrubber water discharge limits that consider the full suite of contaminants.

1 Global use of scrubbers on ships and contaminants within scrubber discharge water

Global regulatory limits on maximum allowable sulphur content in marine fuels were reduced from 3.5% m/m (mass by mass) to 0.5%[1] as of 1 January 2020 by the International Maritime Organization (IMO 2008). To comply with these limits, ships must switch to a fuel with lower sulphur content or install an exhaust gas cleaning system (EGCS), also known as a scrubber. Installation of a scrubber allows for continued use of lower cost residual fuels (heavy fuel oil) that have higher sulphur content. Within the scrubber, the exhaust gas passes through a fine spray of alkaline water which readily dissolves sulphur oxides (SO_X), nitrogen oxides (NO_X) and numerous other contaminants so that levels are sufficiently reduced in air emissions. The resulting scrubber discharge water is a chemical cocktail of acidifying, eutrophying and contaminating substances and elements (Figure 1).

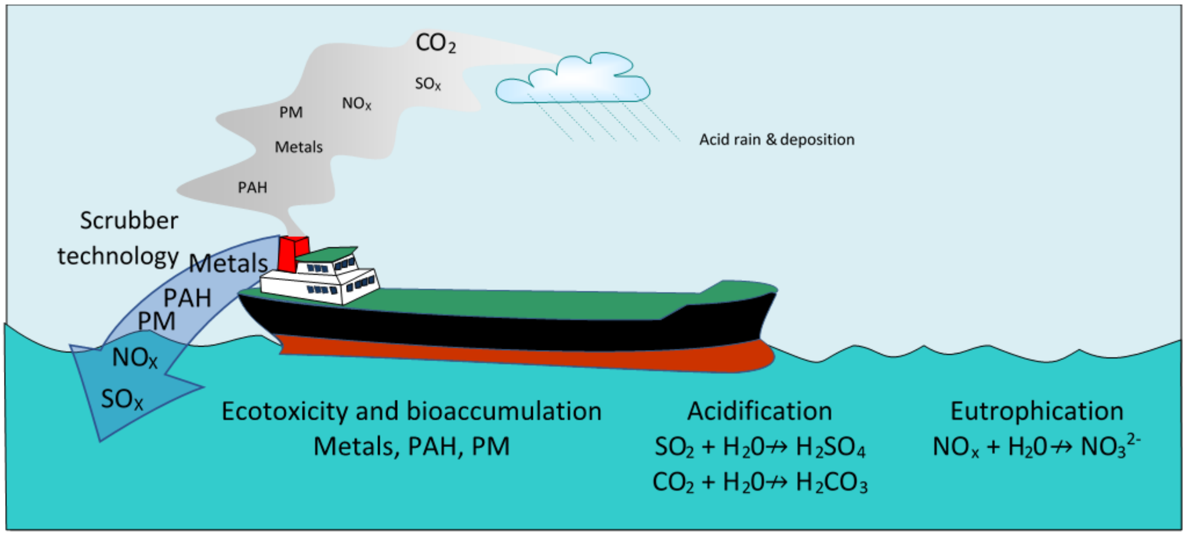

Figure 1. Redistribution of pollutants in air emissions to the sea and the potential impacts in the marine environment by use of scrubber technology: ecotoxicity, bioaccumulation, acidification and eutrophication.

An increasing number of ships have opted to install scrubbers due to the price difference between heavy fuel oil and low sulphur fuels (Abadie *et al.* 2017); (Figure 2). There is also an incentive for the oil industry to continue to use the shipping industry as market for the heavy fuel oil and there are concerns regarding potential disposal of chemical waste into fuel blends (Human Environment and Transport Inspectorate 2018). Broad use of scrubbers is of concern because of the potential effects of scrubber discharge water on marine life and oceanic biogeochemical processes. Early discussions within the IMO regarding the use of scrubbers (MEPC 1998, United States 2003) stressed the importance of ensuring that air pollution is not just transferred to the marine environment. Yet, scrubber discharge water is poorly regulated, and the IMO Marine Environment Protection Committee (MEPC) Guidelines for Exhaust Gas Cleaning Systems (hereafter 'EGCS Guidelines') adopted in 2008 and revised in 2009 and 2015, do not adequately address the potential impacts of scrubber discharge water on the marine environment (Bosch *et al.* 2009, US EPA 2011, Linders *et al.* 2019).

[1] Edited following comments from RGSCRUB

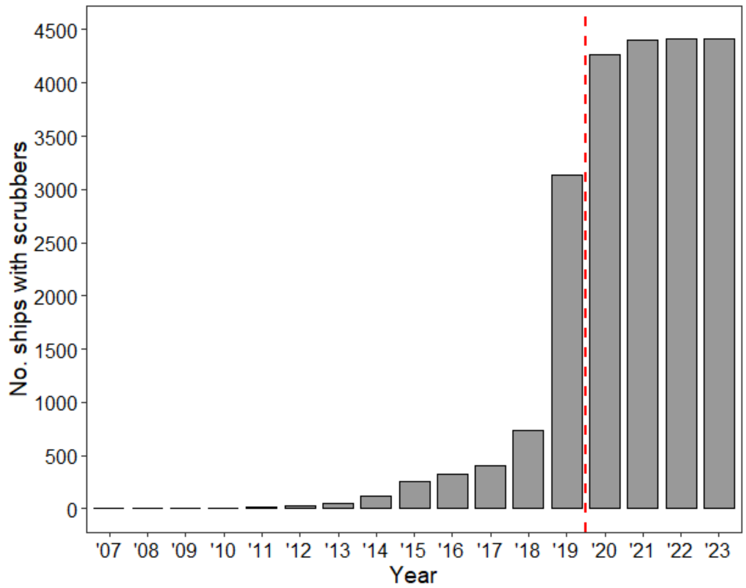

Figure 2. The number of ships with scrubbers (in operation and on order) worldwide increased following reduced IMO limits on sulphur emissions (1 January 2020; red line). Source: DNV- GL Alternative Fuels Insight. 6 July 2020. https://afi.dnvgl.com/

In coastal areas with heavy traffic, especially estuaries and semi-enclosed basins, broad use of scrubbers implies an additional pressure to the aquatic environment. Additional pressures hamper efforts to achieve good environmental status in accordance with marine environmental management, such as the concept of "no deterioration" of the EU Water Framework Directive (EU WFD); (EC 2000) and the Environmental Quality Standards (EQS) and environmental targets of the EU Marine Strategy Framework Directive (EU MSFD); (EC 2008, Borja *et al.* 2017, EC 2017). Belgium is the only EC member state that has enforced a nationwide ban of scrubber water discharge. In 2016, the EC acknowledged the "*increasing evidence from recent studies and analyses of wash-water samples of existing scrubbers that the wash-water contains poly-aromatic hydrocarbons (PAH) and heavy metals (e.g. vanadium, zinc, cadmium, lead and nickel) in potentially larger quantities than initially thought*", yet concluded that more time is needed to gather enough data for consensus.

Other targets set in international agreements are also challenged; e.g. regulation 4 of the International Convention for the Prevention of Pollution from Ships (MARPOL) Annex VI (IMO 2008) and the Agenda 2030 and its Sustainable Development Goals (SDGs), particularly SDG 14 – Life below water (UN General Assembly 2015). To address the alarming state of the ocean and to encourage organizational, scientific and technical actions to enable better chances of achieving the SDGs, the United Nations has proclaimed the Decade of Ocean Science for Sustainable Development (2021–2030). One key societal outcome of the UN Ocean Decade is "a clean ocean, where sources of pollution are identified and removed" (IOC 2019). Pollution risk from ships using scrubbers is also high for marine protected areas, a primary management tool for conserving marine biodiversity, as vessels traveled through all but 5 of over 10 000 marine protected areas in 2019 (Figure 3).

Broad use of scrubbers will cause regular and repeated discharge of highly polluted water into the marine environment. Concern around the potential impacts of this added pollution have already become evident, even though the introduction of scrubbers on ships is relatively recent. An increasing number of ports, regions and states have restricted the use of scrubbers in their territorial waters. Here we present a scientific review of the state of knowledge on the potential impacts of scrubbers on the marine environment, including both biogeochemical processes and contaminants such as polycyclic aromatic hydrocarbons (PAHs), metals, and their mixtures.

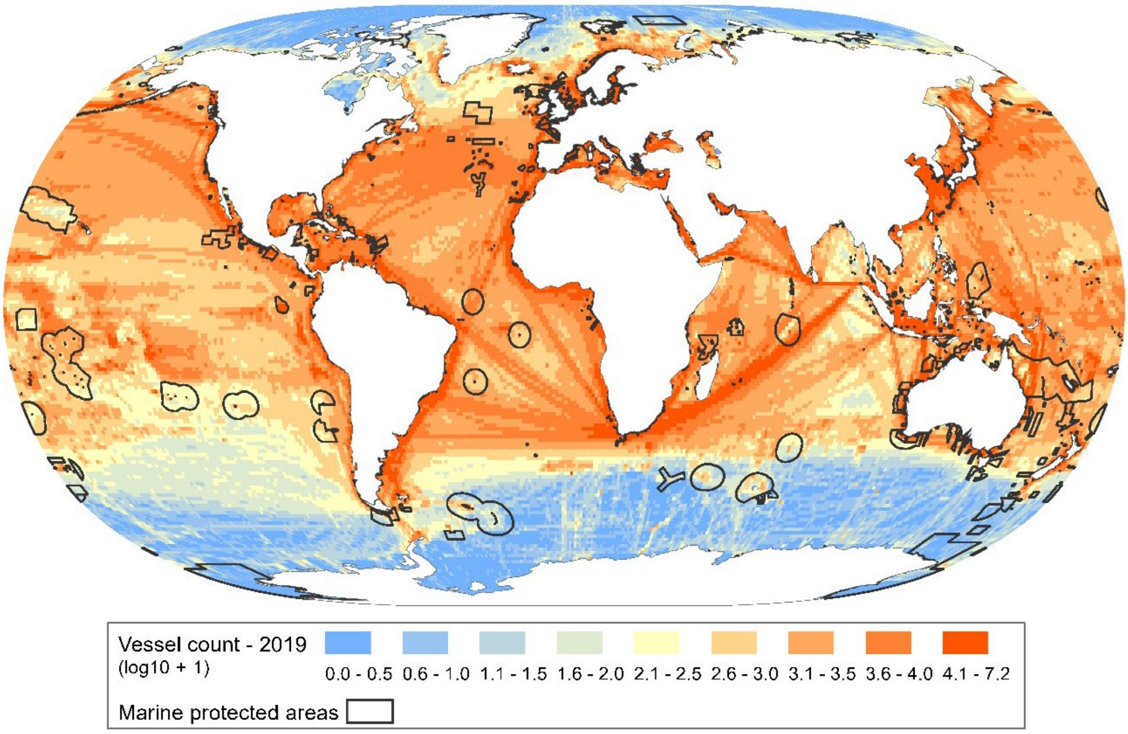

Figure 3. Map showing the overlap of vessel traffic and marine protected areas in 2019. Vessel counts are unique vessels within a 1° grid tracked using Automatic Identification Systems (AIS). Data sources: Canadian Space Agency and World Database on Protected Areas (https://www.protectedplanet.net/).

1.1 Scrubber operation and production rates of discharge water volumes

Scrubbers are classified as open loop (OL), closed loop (CL) or hybrid systems (can use OL and CL modes); (Figure 4). OL scrubbers dominate the current global market (81%), whereas hybrid systems are present in 17% of ships equipped with scrubbers, and CL systems are relatively rare (2%). The type of system or the mode of operation affect the discharge volumes and pollutant concentrations in scrubber discharge water because of the different water processing approaches and methods (as explained below).

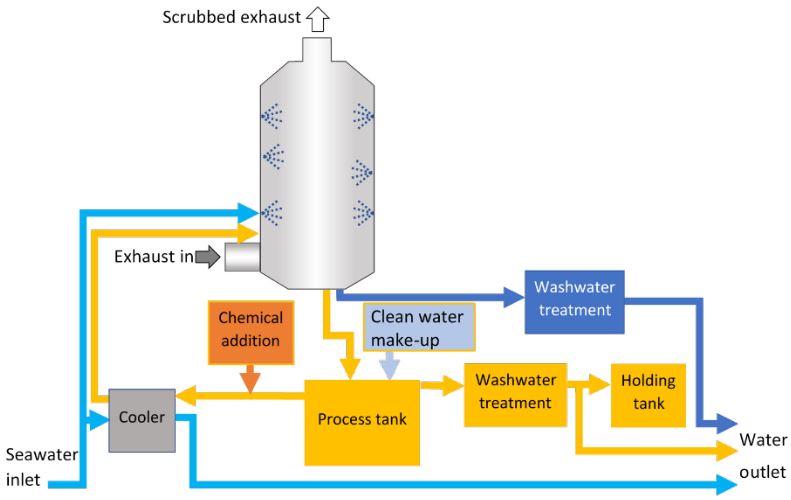

Figure 4. Simplified diagram of a hybrid scrubber system. Light and dark blue lines represent the open loop mode, and the yellow lines show the closed loop mode. Modified from EGCSA (2012), https://www.egcsa.com/resources/technical_gallery/.

Open loop systems, also called seawater systems, require large volumes of seawater (on the order of 10s of m^3 water/MWh of engine power output) and rely on its natural alkalinity for removal of sulphur oxides in the scrubbing process. The used water is directly discharged back to the sea, rarely with treatment for removal of solids or dilution with seawater to reduce acidity (see Figure 4, light and dark blue lines). The average water flow rate in OL systems is 45 m^3*MWh^{-1} (US EPA 2011, EGCSA 2012, Lloyd's Register 2012) and was considered by the EGCS Guidelines as the basis to develop the discharge criteria (Annex 16 of MEPC 2008a). This implies that a medium size vessel (with 12 MW engine power) with a scrubber installed would have a discharge volume of 540 m^3*h^{-1} (~143 000 gallons*h^{-1}). This is notably higher than typical bilge water discharges, which range from 0.01–13 m^3/d (CE Delft and CHEW 2017). The required flow rate, however, varies greatly as a function of the physical-chemical properties of the water (temperature, alkalinity and salinity), the desired SO_x removal efficiency (Karle and Turner 2007), and the effectiveness of the water-gas contact, depending on the system design (EGCSA 2012). For instance, Teuchies *et al.* (2020) reported an average flowrate of 87 ± 50 m^3*MWh^{-1}, Buhaug *et al.* (2006) indicated flow rates in the range of 40–100 m^3*MWh^{-1} while Schmolke *et al.* (2020) recorded flow rates of 75–140 m^3*MWh^{-1} for effective reduction of SO_x under stable conditions.

Closed loop systems, also called freshwater systems, employ freshwater treated with an alkaline substance to adjust the pH level to enable effective SO_x removal. After the washing process in the scrubbing tower, the water is processed, recirculated and a small portion (bleed-off) is removed from the system and released to the sea (see Figure 4, yellow lines). Bleed-off discharge takes place after solids removal and ranges from 0.1–0.3 m^3*MWh^{-1} (MEPC 2008a). Teuchies *et al.* (2020) reported an average flowrate of 0.47 ± 0.25 m^3*MWh^{-1}. The removal of solids implies partial reduction of contaminants. Alternatively, bleed-off water is stored in a holding tank for later discharge into the sea (where allowed) or disposal ashore in port reception facilities. Residuals removed during water treatment (also known as sludgefoof[2]) must be properly disposed of ashore according to the EGCS Guidelines, local regulations, and the recent EU Directive (2019/883) on port reception facilities for the delivery of waste from ships (EC 2019).

In both OL and CL systems, other substances in addition to SO_x are transferred from the exhaust gas to the wash water and are entrained in the scrubber discharge water (Figure 1). This includes

[2] Edited following comments from RGSCRUB

contaminants, such as heavy metals, oil residues, polycyclic aromatic hydrocarbons (PAHs), and nitrogen oxides (Endres *et al.* 2018).

1.2 Chemical composition of scrubber discharge water

Several studies have characterized the chemical composition and concentrations of contaminants in scrubber discharge water from OL (Table 1) and CL systems (Table 2). The chemical composition depends on several factors, including scrubber design and contaminant removal efficiency, fuel and lube oil composition, and ship operation conditions (such as engine load, ship age and quality of combustion, water treatment installed, etc.). For example, corrosion of the scrubber system may contribute to the presence of metals in the discharge water (Den Boer and Hoen 2015). In CL systems, the water residence time strongly affects the resulting water quality (Kjølholt *et al.* 2012). Although CL discharge volumes are smaller compared to typical OL discharge volumes, the concentrations of contaminants are typically higher. For instance, Teuchies *et al.* (2020) reported concentrations of metals (40 times on average) and PAHs (1.3 times on average) higher in CL than in OL discharges, and concluded that due to the bleed-off treatment in CL systems, the amount of contaminants discharged to the marine environment are less than from OL systems (6 times for metals and 183 times for PAHs).

1.2.1 Metals

Eleven metals have been recorded in scrubber discharge water; the highest reported concentrations are of vanadium, nickel, copper and zinc (Tables 1 and 2). Heavy metals are mainly found in their dissolved state in scrubber discharge water (Carnival Corporation & PLC and DNV-GL 2019, Schmolke *et al.* 2020). Vanadium and nickel originate from, and strongly correlate to the sulphur content of, the fuel (Teuchies *et al.* 2020), but the high concentrations of copper and zinc do not correspond to the fuel composition (Turner *et al.* 2017, Ushakov *et al.* 2020). These instead may be related to materials used within the vessel equipment, such as the sampling tube, antifouling system and corrosion protection anodes. Elevated levels of copper and zinc have also been found in OL inlet water samples (Schmolke *et al.* 2020).

1.2.2 Organic substances

Organic substances contained in scrubber discharge water are hydrocarbon oil residues (OL: 0.1-0.4 mg*L^{-1}; CL: 2-21 mg*L^{-1}) (Kjølholt *et al.* 2012, Magnusson *et al.* 2018, Schmolke *et al.* 2020, Ushakov *et al.* 2020) and PAHs (Tables 1 and 2). Oil residues are partially combusted components from fuel and lubricating oil. PAHs may originate from the fuel (petrogenic) and from the fuel combustion process (pyrogenic).

Table 1. Concentration of contaminants in scrubber discharge water from open loop (OL) systems as reported by several studies (adapted from Linders *et al.* (2019)).

	Studies [number of sampled ships]						
	A [20]	B [5]	C [Lab]	D [1]	E [1]	F [1]	G [1]
Metals	**Mean value (μg*L^{-1}) (minimum-maximum)**						
Arsenic	0.0 (0-0)	3.3 (1-6.9)	1.0	1.4	0.2	<0.1	1.7
Cadmium	0.0 (0-0)	0.03 (0.01-0.07)	0.035	BD	<0.2	0.05	<0.01
Chromium	27.3 (2-60)	-	22.8	1.9	4.8	<1.0	1.9
Copper	45.9 (6-140)	6.4 (1.6-15.7)	8.12	21.0	188	41.6	2.3
Iron	-	-	997	-	-	-	-
Lead	72.3 (20-120)	0.08 (0.04-2.1)	1.7	0.61	17.0	5.0	0.64
Mercury	8.0 (8-8)	-	-	-	0.086	<0.1	-
Molybdenum	-	-	-	-	-	-	11.1
Nickel	63.0 (20-240)	15.7 (4-67)	17.9	41.0	42.0	32.8	29.7
Vanadium	213.3 (20-860)	78.4 (11-290)	58.0	162.0	164.3	35.0	111.1
Zinc	236.4 (20-2,000)	4.7 (2-133)	48.3	6.7	325.0	6.0	10.9
PAHs	**Mean value (μg*L^{-1}) (minimum-maximum)**						
Acenaphthene	0.34 (0.01-1.6)	-	-	-	-	-	1.92
Acenaphthylene	0.16 (0.02-0.58)	-	-	-	-	-	0.027
Anthracene	0.12 (0.02-1.2)	-	-	-	-	-	0.12
Benzo(a)anthracene	0.23 (0.02-1.2)	0.02 (<0.006-0.04)	0.006	-	-	-	0.34
Benzo(a)pyrene	0.11 (0.01-0.55)	0.04 (<0.012-0.1)	0.014	-	-	-	1.09
Benzo(b)fluoranthene	0.10 (0.01-0.37)	-	0.012	-	-	-	<0.01
Benzo(g,h,i)perylene	0.08 (0.01-0.36)	-	0.014	-	-	-	0.095
Benzo(k)fluoranthene	0.04 (0.01-0.09)	-	-	-	-	-	0.074
Chrysene	0.26 (0.02-1.6)	-	-	-	-	-	0.016
Dibenzo(a,h)anthracene	0.03 (0.01-0.08)	-	0.006	-	-	-	0.012
Fluoranthene	0.17 (0.01-0.76)	-	-	-	-	-	0.021
Fluorene	0.63 (0.04-1.8)	-	-	-	-	-	<0.01
Indeno(1,2,3-c,d)pyrene	0.04 (0.01-0.14)	-	-	-	-	-	<0.01
Naphthalene	3.65 (0.02-14)	3.02 (0.57-9.47)	0.006	-	0.48	-	<0.01
Phenanthrene	1.88 (0.08-6.1)	1.61 (0.67-2.89)	0.006	-	-	-	0.012
Pyrene	0.42 (0.01-2.6)	-	0.007	-	-	-	<0.01
PAH_{EPA16}	8.25 (0.31-33.0)	4.69 (1.24-12.5)	0.071	-	0.48	-	3.70

A, EGCSA and Euroshore (2018); B, Germany (2018); C, Japan (2019); D, Koski *et al.* (2017); E, Kjølholt *et al.* (2012); F, Buhaug *et al.* (2006); G, Ushakov *et al.* (2020); BD, below detection limit.

Table 2. Concentration range[3] of contaminants in scrubber discharge water from closed loop (CL) systems as reported by several studies (as prepared within the ongoing project ImpEx, funded by the German Environment Agency (UBA), Marin-Enriquez *et al.* (2020)).

	Studies [number of sampled ships]			
	A [3]	**B [4]**	**C [1]**	**D [2]**
Metals	**Value (µg*L-1) minimum-maximum**			
Arsenic	9-25	<10-30	8.8-9-8	10-20
Cadmium	0.05-0.4	0.96-<20	<0.05-0.09	>0.2-<0.5
Chromium	-	<10-14,000	-	9-22
Copper	10-58	<10-200	390-860	32-150
Iron	304-709	-	-	-
Lead	1-3	<5-<10	1.6-3.8	0.16-<6
Mercury	-	<0.200	<0.050	0.001-0.005
Molybdenum	-	-	-	-
Nickel	478-6,289	220-6,600	1,300-3,100	830-4,400
Vanadium	3,542-10,637	2,800-25,000	6,100-14,000	9,800-13,000
Zinc	76-240	40-2,400	160-420	<70
PAHs	**Value (µg*L-1) minimum-maximum**			
Acenaphthene	<0.005-1.035	0.03-0.49	-	2.10
Acenaphthylene	<0.002-0.20	0.01-0.07	-	0.36
Anthracene	2.16-15.0	<0.01-0.11	-	<0.13-0.40
Benzo(a)anthracene	0.51-1.96	<0.01-0.09	-	0.21
Benzo(a)pyrene	0.06-0.37	<0.01	<0.01	0.014-<0.10
Benzo(b)fluoranthene	0.19-1.11 i)	<0.01-0.06	0.10 i)	0.10-0.11
Benzo(g,h,i)perylene	0.07-0.65	<0.01-0.011	<0.02 ii)	0.03-<0.10
Benzo(k)fluoranthene	0.19-1.11 i)	<0.01	0.10 i)	0.02-0.07
Chrysene	0.55-3.41	<0.01-0.16	-	0.33
Dibenzo(a,h)anthra-cene	0.04-0.14	<0.01	-	<0.10
Fluoranthene	0.66-3.88	0.04-0.44	-	0.22-1.49
Fluorene	0.33-2.89	0.09-1.9	-	3.2
Indeno(1,2,3-c,d)py-rene	0.03-0.31	<0.01	<0.02 ii)	<0.10
Naphthalene	0.12-3.85	0.06-5.7	0.32-0.49	4.4-4.8
Phenanthrene	2.35-20.1	0.49-4.5	-	10.0
Pyrene	0.94-5.90	0.04-0.5	-	0.54
PAH_{EPA16}	11.8-54.4	0.8-12.6	3.8-24	16.0-21.9

A, Schmolke *et al.* (2020); B, EGCSA and Euroshore (2018); C, Kjølholt *et al.* (2012); D, Magnusson *et al.* (2018).

[3] Added following comments from RGSCRUB

i) Sum of benzo(b)fluoranthene and benzo(k)fluoranthene; ii) Sum of benzo(g,h,i)perylene and indeno(1,2,3-c,d)pyrene

1.2.3 pH and alkalinity

A pH decrease in the water used for SO_X-scrubbing is a result of the absorption of SO_2 and its transformation to sulphate species, which produces hydrogen ions that increase acidity. Studies have reported an acidic pH in OL discharge water samples (2.8–5.8), whereas the pH in CL discharge tends to be higher (4.9–7.6) (Table 3); (Kjølholt *et al.* 2012, Koski *et al.* 2017, Magnusson *et al.* 2018, Schmolke *et al.* 2020, Ushakov *et al.* 2020). However, the pH range for OL systems includes some samples taken after dilution, which is used in some systems to increase the pH of the discharge water before release to prevent acute environmental effects. Onboard dilution also reduces the corrosive properties of acidic scrubber discharge water in the piping.

Alkalinity is a crucial parameter in the wash water to ensure efficient SO_X removal (Karle and Turner 2007). In OL systems, bicarbonate ions in seawater react with hydrogen ions neutralizing the acidity and raising the pH again (Den Boer and Hoen 2015); thus, enhancing further absorption of SO_2. This implies that the natural alkalinity of seawater is consumed by the scrubbing process. The alkalinity measurements by Schmolke *et al.* (2020) showed a significant drop of alkalinity in OL systems with inlet values in the range of 1.6–2.6 mmol*L^{-1} and outlet values in the range of 0.0–1.4 mmol*L^{-1}. As aforementioned, in CL systems, alkaline substances are added to the fresh water to adjust the pH level. Schmolke *et al.* (2020) reported zero (0 mmol*L^{-1}) alkalinity in all discharge water samples from CL systems. Both pH decrease and alkalinity consumption raise concerns about the effects of scrubber discharges on ocean acidification (see section 2.2 Acidification).

Table 3. Average values (± 95% CI) of pH and sulphur concentrations for open and closed loop discharge water and open loop inlet water across published studies[4]. N = number of samples included. The average and confidence interval of pH is calculated from the 10^{-pH} values, i.e. the [H^+]. As prepared for the ongoing EU H2020 EMERGE project report by Ytreberg *et al.* (2020).

	Open loop scrubber discharge		Open loop inlet water		Closed loop scrubber discharge	
Parameter	**$\bar{X}$ ± 95% CI**	**N**	**$\bar{X}$ ± 95% CI**	**N**	**$\bar{X}$ ± 95% CI**	**N**
pH	3.85 ± 0.33	36	7.72 ± 0.14	29	4.54 ± 0.51	11
Sulphur (mg*L^{-1})	2,200 ± 446	18	2,376 ± 480	13	12,280 ± 10,104	9

1.2.4 Nutrients

Nitrate in scrubber discharge water is highly dependent on the environmental concentrations in the water taken for scrubbing, as well as on the NO_X removed from the exhausts (EGCSA and Euroshore 2018). The NO_X removal rate in conventional scrubbers is generally assumed to be limited (<10%) (Den Boer and Hoen 2015) due to poor solubility of nitrogen monoxide in water, which is present in higher amounts in the exhaust than the more soluble nitrogen dioxide (Lloyd's Register 2012). Scrubber discharge water samples showed nitrate concentrations in the range of <0.03-22.3 mg*L^{-1} in OL and <4.4-290 mg*L^{-1} in CL systems (EGCSA and Euroshore 2018, Magnusson *et al.* 2018, Schmolke *et al.* 2020, Ushakov *et al.* 2020). However, there is substantial variation in the reported data (Table 4) and in close to 30% of the measurements that included

[4] Edited following comments from RGSCRUB

analyses of both inlet and scrubber discharge water, the reported nitrate values are lower in the scrubber discharge water than the inlet concentrations. In an ongoing project financed by the Swedish Transport Agency, potential chemical interferences in spectrophotometric analyses of nitrate (potentially resulting in false low nitrate values) in scrubber discharge water will be investigated.

Table 4. Concentrations of nutrients, nitrogen species and iron (average ± 95% CI) measured in scrubber discharge water from open and closed loop systems, inlet water associated with open loop systems. N = number of samples included. As prepared for the ongoing EU H2020 EMERGE project report by Ytreberg *et al.* (2020).

	Open loop scrubber discharge		Open loop inlet water		Closed loop scrubber discharge	
Nitrogen species (mg*L^{-1})	**$\bar{X}$ ± 95% CI**	**N**	**$\bar{X}$ ± 95% CI**	**N**	**$\bar{X}$ ± 95% CI**	**N**
Nitrate (NO_3^{2-})	2.83 ± 2.06	31	3.21 ± 2.23	30	110.98 ± 135.73	4
Nitrite (NO_2^-)	0.76 ± 0.68	28	0.97 ± 1.28	26	55.76 ± 130.71	4
Ammonium (NH_4^+)	0.73 ± 0.03	17	0.07 ± 0.04	14	-	-
Iron	0.24 ± 0.37	4	0.032 ± 0.08	3	-	-

1.3 Estimates of scrubber contaminant loads to the environment[5]

Contaminant loading to the environment from the use of scrubbers is significant when compared to other sources of contaminants. Teuchies *et al.* (2020) modeled contaminant fluxes in the Harbour docks in Port of Antwerp with a "HIGH" scenario with 20% of the ship emissions treated by open loop scrubbers. For several contaminants, the input from scrubbers exceeded the sum of all other known sources: naphthalene (57 kg*yr^{-1} for scrubbers compared to 19 kg*yr^{-1} for all other sources), phenanthrene (30 kg*yr^{-1} for scrubbers compared to 11 kg*yr^{-1} for all other sources), fluorene (10 kg*yr^{-1} for scrubbers compared to 6 kg*yr^{-1} for all other sources), and nickel (994 kg*yr^{-1} for scrubbers compared to 60 kg*yr^{-1} for all other sources). The Baltic Sea, a semi-enclosed brackish sea with intense maritime traffic, and the North Sea were the first designated Sulphur Emission Control Areas (enforced 2005 and 2006, respectively). Following regulations, extensive measurements have been made in these seas to estimate contaminant loads from scrubbers (e.g. Jalkanen and Johansson 2019, Schmolke *et al.* 2020, Ytreberg *et al.* 2020). In other regions, estimates of scrubber discharge water volumes and contaminant loads are scarce. However, Georgeff *et al.* (2019) estimated 47 million tonnes of scrubber discharge will be released in Pacific Canada during 2020.

For the Baltic and North Seas, Schmolke *et al.* (2020) used an emission model based on ship traffic (determined using Automatic Identification System [AIS] signals) to estimate the total input of scrubber discharge water and pollutants. Yearly discharge volumes were modeled under different scenarios taking into account the uncertainty about the number of ships fitted with scrubbers and the range of discharge water flowrates from values recorded during a sampling campaign (from 60 m^3*MWh^{-1} up to 140 m^3*MWh^{-1} in the case of open loop). Total pollutant loads were calculated based on estimated water emissions and concentrations obtained from analysis of discharge water samples (minimum and maximum concentrations). The total yearly scrubber water discharges in the Baltic and North Seas ranged from 210 to 4500 million tonnes. Vanadium and

[5] Edited following comments from RGSCRUB

nickel emission loads from scrubber discharge water were estimated in the range of 3–1407 tonnes and 1–331 tonnes per year, respectively. Similarly, the total yearly emission loads for oil and PAH_{EPA16} ranged from 11–1226 tonnes and 0.3–63 tonnes, respectively[6].

The annual reports by the Finnish Meteorological Institute to HELCOM Maritime on emissions and discharge by shipping in the Baltic Sea are based on AIS data linked to the produced volumes of different waste streams from ships using the Ship Traffic Emission Assessment Model (STEAM, Jalkanen and Johansson (2019)). From this, Jalkanen and Johansson (2019) estimated the discharge of scrubber water (assuming OL: 45 m^3*MWh^{-1} and CL: 0.25 m^3*MWh^{-1}) in the Baltic Sea at 77 million m^3 during 2018. The total number of individual ships operating in the Baltic Sea in 2018 was approximately 8000 (with roughly 2000 ships estimated to be in operation at any given time); of these, 99 ships were equipped with a scrubber (14 OL, 10 CL and 75 hybrid). In combination with concentrations of contaminants (trace elements and PAHs) compiled by Ytreberg *et al.* (2020), a scoping calculation can be made to compare the load of contaminants from waste streams on board ships in the Baltic Sea during 2018 (Figure 5). Even though almost all 2000 ships were discharging bilge, black and grey water, the load of metals and PAHs from the 99 ships equipped with scrubbers was higher by 10–100-fold, with the load from the open loop systems dominating.

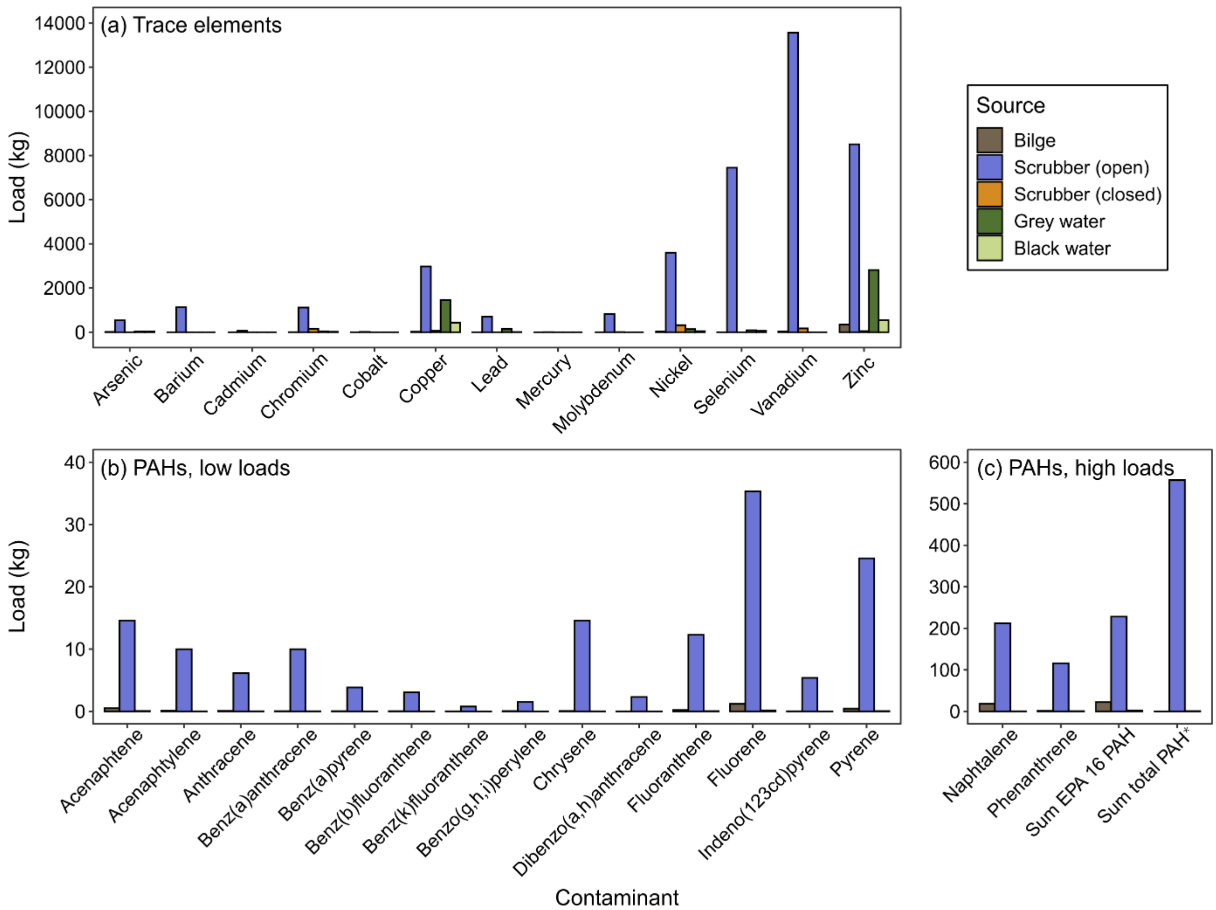

Figure 5. Comparison of trace elements (a), low level PAH (b) and high level PAH (c) contaminant loads from shipping related waste streams in the Baltic Sea in 2018. PAH (b) and (c) only compare loads from open and closed loop scrubber discharge water with bilge water, as grey and black water are not expected to contain PAH. *Sum total PAHs was not reported for bilge water,

[6] Removed comparison of total air emissions in the entire OSPAR area following discussions at ADGSCRUB

but is included to highlight that only analyzing Sum EPA 16 PAH, which excludes e.g. alkylated PAHs, leads to an underestimation of the total PAH emissions in scrubber discharge water. Data from Jalkanen *et al*. 2019, and Ytreberg *et al*. 2020.

2 Consequences and impacts of scrubber discharge water

The combination of contaminants, acidifying, and eutrophying substances in scrubber discharge water can be expected to impact the marine environment. However, the extent of the impact is challenging to assess as it includes potential interacting effects (Rudén 2019) and depends on ship related factors such as the number of ships equipped with scrubbers, type of operation and fuel composition, as well as environmental factors, like hydrographic conditions, physical and chemical properties of the water and types of organisms (Linders *et al.* 2019).

2.1 Contamination

There is now a growing body of laboratory evidence characterizing the toxicological threat posed by scrubber discharges on a range of marine biota (Koski *et al.* 2017, Endres *et al.* 2018, Magnusson *et al.* 2018). While information on direct field-related impacts is limited, it has already been highlighted that the increase in the use of scrubbers and their associated discharges are likely to pose a long-term environmental threat in ecologically sensitive areas (Lange *et al.* 2015). Consideration should also be given to heavily impacted receiving environments, such as ports and estuaries where scrubber dischargers are likely to further contribute to a complex mix of metals, PAHs, organohalogens, and other industrial pollutants. The combined threat from these sources of pollution need to be included when undertaking studies to establish the environmental risk associated with scrubber discharges (Kjølholt *et al.* 2012, Endres *et al.* 2018). Faber *et al.* (2019) suggested that the use of scrubbers in many open harbours is highly unlikely to breach chemical exceedance limits in water and sediment[7]. However, the assumptions made in that study considered equal dilution across the sea area, rather than examining shipping behaviours more closely.

Simulations in port environments estimate high increases in contaminant levels as a result of scrubber discharge. Simulations of the Port of Antwerp showed pronounced increases in the surface water for naphthalene, with an increased concentration of 39% under “scenario LOW” and 189% under “scenario HIGH” and vanadium, which increased 9% under “scenario LOW” and 46% under “scenario HIGH” (Teuchies *et al.* 2020). The modeling results from the Scheldt estuary for naphthalene showed increased concentration of 5.0% with “scenario LOW” and 25% with “scenario HIGH”. In both the Port of Antwerp and the Scheldt estuary, the EQS for surface water according to EU WFD are already exceeded with respect to fluoranthene, further surpassed by scrubber discharge. Nickel, zinc and vanadium are all close to the EQS in the Port of Antwerp, and for nickel and zinc the scrubber discharge contribution is expected to cause exceedance. In the Scheldt estuary, the modeled concentration of pyrene in surface water also exceeds the EQS according to EU WFD and vanadium is close to the EQS.

2.1.1 Scrubber discharge water is toxic to marine biota

Scrubber discharge water has been shown to have lethal and sub-lethal effects on the marine zooplankton community, depending on exposure time and dilution in laboratory experiments. Effects on copepods (crustaceans commonly found in coastal waters) include reduced survival and feeding rates and delayed development and molting. Instant mortality occurred at 80–100%

[7] Edited following comments from RGSCRUB

treatments of scrubber discharge water within minutes of exposure, and diverse chronic sublethal effects occurred at 1% treatment within days or weeks of exposure (Koski *et al.* 2017, Magnusson *et al.* 2018). While difficult to test in a laboratory setting, the accumulated effects of long-term exposure to scrubber discharge water can be expected to be severe and have the potential to influence zooplankton community structure and associated secondary production, depending on the residence time of the water in an enclosed port or harbour.

These strong negative responses to scrubber discharge water occur at concentrations of scrubber discharge water with heavy metals and PAH concentrations that are many-fold lower than the concentrations that induce effects in marine zooplankton in single-compound exposures (Koski *et al.* 2017, Magnusson *et al.* 2018). For instance, the nickel concentration in OL scrubber discharge water is ≤ 60 µg*L^{-1} (Table 1), whereas the LD_{50} (Lethal Dose, 50%) of marine zooplankton exposed to nickel is at much higher concentrations of 0.25–2.6 mg*L^{-1} (Verriopoulos and Dimas 1988, Mohammed *et al.* 2010, Tlili *et al.* 2016, Zhou *et al.* 2016). Similarly, the LD_{50} of the copepod *Oithona davisae* exposed to naphthalene was 7.2 mg*L^{-1} (Barata *et al.* 2005) and the LD_{50} of the copepod *Pseudodiaptomus pelagicus* exposed to phenanthrene was 161 µg*L^{-1} (Kennedy *et al.* 2019), both of which are many-fold higher than the concentrations measured in scrubber discharge water (Table 1). Therefore, heavy metals and PAH compounds in scrubber discharge water are likely acting synergistically, an effect that may be enhanced by the acidity, especially for the metals (Parmentier *et al.* 2019 and references therein). Alternatively, or additionally, the observed effects on copepods may be caused by unknown compounds present in the discharge water.

There are strong indications that other compounds in scrubber discharge water than analyzed so far provoke toxic effects. In-vitro biotests on bleed-off discharge water showed stronger effects of the response of the aryl-hydrocarbon receptor than could be explained by PAH concentrations alone (Kathmann *et al.* in prep.). This receptor mediates important biological effects including mutagenicity of compounds such as PAHs, dioxins and dioxin-like PCBs in vertebrates. Multiple studies have shown that assessment of toxic effects based only on priority PAHs may underestimate the presence of aryl-hydrocarbon receptor agonists and mutagenic compounds (Vondráĉek *et al.* 2007, Sun *et al.* 2014, Lam *et al.* 2018). The full characterization of toxic PAHs in scrubber discharge waters, including those compounds in particulate form, and many alkyl-homologues and strong mutagens and/or carcinogens, such as $C_{24}H_{14}$ PAHs, is not routinely done at present (Allen *et al.* 1998, Durant *et al.* 1998, Linders *et al.* 2019).

2.1.2 Bioaccumulation of contaminants from scrubber discharge water

Beyond the acute toxic effects of the scrubber discharge water, there is potential for bioaccumulation of contaminants in the food web. Scrubbers discharge large amounts of metals and PAHs in dissolved, readily bioavailable form. These contaminants at ultra-trace levels will be concentrated in marine plankton, filtering organisms, fish and marine mammals, to levels which may impair their vital functions and their biological performance (e.g. Echeveste *et al.* 2011 and 2012, Tiano *et al.* 2014, Battuello *et al.* 2016, Calbet *et al.* 2016, Chouvelon *et al.* 2019, Ytreberg *et al.* 2019). In fact, the concentrations of contaminants may be hundreds to million times higher in plankton than in the water column (e.g. Berglund *et al.* 2000, Gobas *et al.* 2009, Hallanger *et al.* 2011, Frouin *et al.* 2013, Strady *et al.* 2015, Chouvelon *et al.* 2019 and references therein).

Bioaccumulation of contaminants in marine food webs is influenced by many factors, such as contaminant properties (e.g. Fisher *et al.* 2000), organismal ecophysiology (Xu and Wang 2001, Wang 2002), and physical and chemical environmental conditions (Breitburg *et al.* 1999, Wang *et al.* 2001). However, plankton play a key role in the fate of many persistent organic contaminants on a global scale (Dachs *et al.* 1999, Galban-Malagon *et al.* 2013b a and b, Parmentier *et al.* 2019).

The body burden of contaminants discharged from scrubbers in the marine plankton and higher trophic-level biota, together with the water-exposure pathways, should be considered in assessments of the potential impacts of scrubber discharge water in the marine ecosystem. Although zooplankton can also have high efflux and detoxification rates (Wang 2002), the presence of heavy metals in fish, mussels and marine mammals confirms that these substances can bioaccumulate in the food web, and that increased concentrations in the water and sediment should be of concern.

Contaminants also accumulate in sediments and they can remain there or can move to the water column depending on the redox conditions and the diagenetic processes that take place; in particular, metals show higher mobility in water with lower pH. The activity of benthic communities such as burrowing and bioturbation, as well as human activities including dredging in harbours, may enhance the remobilization of contaminants from sediments to the water column. Hence, sediments can act as a sink or source of contaminants and the understanding of the functioning of the natural sediment-water system and of its interaction with biota is necessary for the assessment and management of water bodies. Even though monitoring the ecotoxicological risk of sediments is not incorporated in the EU WFD (EC 2000, Borja *et al.* 2004), several other international guidelines focusing on dredged material emphasize the importance of ecotoxicological testing of sediments in addition to chemical, physical and biological characterization (DelValls *et al.* 2004). By analyzing contaminants in both water and sediment to determine the status of water quality, resources could be better targeted at waterbodies where levels of pollution have a greater impact on fish and other marine biota.

2.1.3 Effects of PAHs and heavy metals on fish and mammals

Although no studies exist on the direct effects of scrubber discharge water on fish or marine mammals, PAHs and heavy metals are known to induce detrimental effects on these organisms. Observed effects of PAHs on adult fish include narcosis, mortality, decrease in growth, lower condition factor, edema, cardiac dysfunction, a variety of deformities, lesions and tumors of the skin and liver, cataracts, damage to immune systems and compromised immunity, estrogenic effects, bioaccumulation, bioconcentration, trophic transfer and biochemical changes (Logan 2007). Similarly, chronic exposure of early life stages of sensitive fish species to some PAHs can lead to adverse developmental effects, including cardiac dysfunction (reviewed in Billiard *et al.* 2008). However, the responses to PAHs are variable and mediated by the life history and ecology of the fish species and mechanisms causing adverse effects (Logan 2007), as well as by the interaction between exposure period and concentration (Santana *et al.* 2018). The effects of PAHs are often non-additive as most environmental exposures are from complex mixtures of PAHs and multiple mechanisms are involved in developmental effects (Incardona *et al.* 2004, Billiard *et al.* 2008). The minimum concentrations needed for adverse effects of PAHs on fish are therefore hard to predict.

Early developmental stages of fish are particularly sensitive to water pollution from heavy metals (Jezierska *et al.* 2009) and negative correlations between fish size and concentrations of cadmium, chromium, copper, iron, lead and zinc have been demonstrated (Canli and Atli 2003). Marine fish tend to have relatively high levels of arsenic, cadmium and lead in their tissues compared to other human food items (Bosch *et al.* 2016). However, the effects of scrubber discharge water-relevant metals (nickel and vanadium) on fish, and their bioaccumulation up the food chain, are not well studied.

Marine mammals are long-lived, apex predators that can accumulate relatively high levels of PAHs and metals in their tissues. Studies of pollutant concentrations, particularly metals, have been conducted for a broad variety of marine mammal species around the world and it has been suggested that marine mammals are important biomonitoring organisms for metal concentrations (e.g. Monteiro *et al.* 2016, Machovsky-Capuska *et al.* 2020, Monteiro *et al.* 2020). While many studies have measured pollutant concentrations, an animal's life history (e.g., age, breeding, diet, body condition, fasting periods, food availability, habitat use, migration, etc.) and associated changes in physiology can influence tissue contaminant concentrations and toxicological risk. However, studies are beginning to demonstrate that high concentrations of PAHs and metals can have negative effects on marine mammals. For example, De Guise *et al.* (1996) show that the metals present in Canada's St. Lawrence Estuary could lead to the inability of individual beluga whales (*Delphinapterus leucas*) to mount an adequate immune response and may explain the prevalence of severe diseases in that population. Lavery *et al.* (2009) found that South Australian adult bottlenose dolphins (*Tursiops aduncus*) with evidence of renal damage had significantly higher concentrations of cadmium, copper and zinc in their liver and that two dolphins showed signs of possible severe and prolonged metal toxicity. Thompson *et al.* (2007) reviewed metal and PAH concentrations in Pacific harbor seals (*Phoca vitulina richardii*) in the San Francisco Estuary and found that the concentrations can have adverse effects on individual health. Finally, Desforges *et al.* (2016) reviewed the effects of environmental pollutants on the immune system of marine mammals. They found systemic suppression of immune function in marine mammals exposed to environmental pollutants and suggested that exposure to immunotoxic pollutants may be a contributing factor to infectious disease outbreaks.

2.2 Acidification

Ocean acidification (pH and alkalinity decline) is one of the major human-related stressors currently affecting marine ecosystems (e.g. Doney *et al.* 2009, Turley and Gattuso 2012). In particular, maritime traffic emissions (CO_2, SO_X and NO_X) from the burning of fossil fuel oils superimpose on global climate change to acidify ocean waters (Hunter *et al.* 2011). CO_2 related acidification is acting on global scale, as a result of the gas exchange at the air-sea interface, where an increased CO_2 concentration in the atmosphere drives an increased uptake of CO_2 in the ocean, thereby shifting the carbonate system towards release of protons (H^+) (Equation 1).

Equation 1. $$CO_2 + H_2O \leftrightarrow HCO_3^- + H^+ \leftrightarrow CO_3^{2-} + 2H^+ \text{ (carbonate species equilibria)}$$

In contrast to CO_2, SO_{X}- and NO_X-related acidification is acting at local or regional scales following deposition of atmospheric emissions (Hunter *et al.* 2011). In the atmosphere, SO_X and NO_X will react with water and rapidly be converted to strong acid species (H_2SO_4 and HNO_3). Hunter *et al.* (2011) give a detailed description of the differences between CO_2 versus SO_X and NO_X related acidification and conclude that there are two main differences. First, CO_2-related acidification does not alter the alkalinity while each mole of the strong diprotic acid H_2SO_4 will consume 2 equivalents of alkalinity. Analogously, the monoprotic HNO_3 will cause a decrease in alkalinity of 1 equivalent. Secondly, on a longer time scale (few months to a year) the acidification by strong acids (H_2SO_4 and HNO_3) will increase the partial pressure of CO_2 in the water (shifting Equation 1 to the left), resulting in a CO_2 flux from the ocean to the atmosphere. For each ton of SO_2 discharged by scrubbers, the ocean uptake of atmospheric CO_2 is reduced by half a ton (Stips *et al.* 2016), reducing the ability of the ocean to absorb CO_2 (sink role of the ocean) and further contributing to global climate change (Hunter *et al.* 2011).

2.2.1 Modeled pH decrease from scrubbers

In 2014, global shipping CO_2 emissions represented 2.6% of total CO_2 emissions (Smith *et al.* 2014). Eyring *et al.* (2005) and Corbett *et al.* (2007) estimated that shipping was responsible for 15% of the world's airborne NO_X emissions and 5–8% of SO_X emissions. NO_X is primarily formed from nitrogen in the air during combustion at high temperatures, whereas SO_X is directly linked to the sulphur content of the oil type. For instance, if 35% of the fleet (in gross tonnage) in the North Sea Sulphur Emission Control Area was equipped with OL scrubbers, the total amount of SO_X discharged at sea would be 13 times higher than if the whole fleet used low-sulphur fuel oil instead (Dulière *et al.* 2020).

Mathematical modelling approaches provide estimates of the contribution to ocean acidification resulting from the use of scrubbers and from climate change over regional to global areas (e.g. Artioli *et al.* 2012, Hassellöv *et al.* 2013, Stips *et al.* 2016, Turner *et al.* 2018, Dulière *et al.* 2020). Model estimations of shipping-related ocean acidification mostly rely on: (1) available information on CO_2, SO_X and NO_X marine input from maritime traffic and (2) the ability of models to simulate the physical and biogeochemical processes of the marine environment. Studies often base their estimations on extrapolations from scrubber discharge water measurements or on estimates reconstructed from traffic emission models that use information on fuel, ship characteristics and positions (e.g. STEAM3 and DREAM models). Studies that present results averaged over large domains often estimate smaller pH differences (due to the smoothing effect of the average) than those over smaller domains; localized studies can provide a more realistic estimation in potentially problematic areas with intense maritime traffic (Table 5).

On a global scale, NO_X and SO_X-related acidification resulting from human activities is only a few percent of CO_2-induced acidification (Doney *et al.* 2007). Nevertheless, in areas of intense maritime traffic where scrubber water discharges are permitted, scrubber-related ocean acidification could become equivalent to several years or decades-worth of CO_2-induced acidification (Dulière *et al.* 2020). This tendency intensifies for semi-enclosed and enclosed seas (Stips *et al.* 2016).

Table 5. Overview of the annual pH decrease (acidification) in response to ship-borne SO_2 and CO_2 emissions, adapted from Stips *et al.* (2016).

Study	Area	$\Delta pH*yr^{-1}$ (SO_2)	$\Delta pH*yr^{-1}$ (CO_2)
Doney 2007	Global	<0.0004	~0.0010
Hunter 2011	North Sea	0.0014	0.0016
Hunter 2011	Baltic Sea	0.0005	0.0018
Hunter 2011	South China Sea	0.0008	0.0015
Hassellöv 2013	North Sea	0.0024	-
Hassellöv 2013	Global	0.0004	-
Beare 2013	North Sea	-	0.0
COWI 2013	Sound	0.0010	-
Hagens 2014	North Sea	0.0005	-
Hagens 2014	Baltic Sea	0.0001	-
Bates 2014	Global	-	0.0018
Omstedt 2015	Baltic Sea	0.0001	-
Stips 2016	North Sea (0-20m)	0.00024	0.0010
Stips 2016	North Sea	0.00011	0.0008
Stips 2016	Rotterdam	0.0025	0.0010
Moldanová 2018	Baltic Sea	0.0001	-
Bindoff 2019	Global	-	0.0017-0.0027
Dulière 2020	Southern North Sea	0.0040	-
Dulière 2020	Dutch & Belgian coastal areas	Up to 0.031	-
Teuchies 2020	Port of Antwerp	Up to 0.015	-

2.2.2 Potential effects on redox conditions and port sediment[8]

The lower pH and warmer temperature of scrubber discharge water relative to ambient water may cause indirect effects through alteration of redox conditions. In particular, contaminants in sediments may be released if there is a change in conditions, such as the local environment becoming more acidic (Borch *et al.* 2010, Grundl *et al.* 2011). UK port stakeholders are concerned about how the release of acidified, warm scrubber discharge water in ports and harbours may affect availability of contaminants (especially inorganic species) in sediments (British Ports Association 2019). Sediment contaminant concentrations may increase through direct inputs, as described in section 2.1.2, or indirectly as a result of increased mobility following increasing acidity. This may affect the EU WFD environmental status due to increased contaminant concentrations in waters, as well as affecting dredge sediment assessment (according to national regulatory frameworks, e.g. UK Cefas guidelines) such that sediment previously acceptable for disposal at sea is no longer allowed and different methods of disposal (likely to be

[8] Edited following comments from RGSCRUB

more expensive) may be required. There is also uncertainty over potentially higher risk harbours that have low flushing.

2.3 Eutrophication

Introduction of excess nutrients to the marine environment, e.g. from agricultural run-off, sewage, and atmospheric deposition of NO_X, can cause oxygen depletion of coastal waters, increased risk of harmful algal blooms (Sellner *et al.* 2003), and reductions in biodiversity (Smith and Schindler 2009). Shallow sea areas with limited water exchange and substantial nutrient input, e.g. the Baltic Sea, are prone to eutrophication (e.g. Diaz and Rosenberg 2008). The shipping-related nutrient input is dominated (>99%) by atmospheric deposition of nitrogen originating from the formation of NO_X during combustion of fuel. In the EGCS Guidelines, there is a limit set for maximum allowed removal of 12% NO_X in the exhausts by a scrubber, corresponding to a nitrate concentration of 60 mg*L^{-1} (or 968 µmol*L^{-1}) in the discharge water. This results in a more localized transfer of NO_X from ship exhausts to the marine environment, compared to deposition of atmospheric emissions. Koski *et al.* (2017) and Ytreberg *et al.* (2019) showed that NO_X uptake well below the set limits stimulated microbial plankton growth, indicating that scrubber discharge water can contribute to eutrophication.

Today there is consensus among the Baltic Sea States that nutrient loads need to be reduced to improve the environmental status of the Baltic Sea (HELCOM 2018). The total nutrient input from shipping in the Baltic Sea was estimated to account for 6% of the total nitrogen input from all sources in 2014 (Bartnicki and Benedictow 2017). Raudsepp *et al.* (2019) reported a somewhat lower estimate of 1.3–3.3% from all shipping-related nitrogen sources, but also stated that this input could locally impact different biogeochemical variables up to 10%. As land-based emissions of NO_X decrease, the relative share of shipping related emissions increase. For that reason, MARPOL Annex VI (first adopted in 1997 and revised thereafter) includes the regulation of NO_X emissions from ships, and the IMO has designated the Baltic and North Seas as NO_X Emission Control Areas (NECAs) as of 1 January 2021.

3 Available mitigation measures and their environmental consequences

The introduction and global use of a new technology that has known adverse effects and currently unpredictable consequences for the marine environment calls for application of the precautionary principle to avoid another example of *Late lessons from early warnings* (European Environment Agency 2001). The discussions on potential negative impacts from broad use of scrubbers have been ongoing for the past 20 years (MEPC 1998). The relevance and complexity of the subject has led to many commissioned reports submitted to the IMO both by member States and representatives from the maritime industry (Linders *et al.* 2019 and references therein). Most studies, with few exceptions (Kjølholt *et al.* 2012, Faber *et al.* 2019, Japan 2019), conclude that more research on the environmental impact from scrubbers is needed due to lack of consistent data on scrubber discharge water composition, as well as lack of understanding of cumulative risk in the marine environment (Heywood and Kasseris 2019). However, there is no doubt that discharge of scrubber water applies an additional pressure on biogeochemical processes and pollution in the marine environment, and that the persistent organic pollutants (POPs), such as DL-PCBs, PCDDs and PCDFs, that were found at low traces in scrubber discharge water (Linders *et al.* 2019) should be further investigated. Repeated releases of POPs, banned under the Stockholm Convention because of their widespread, long-term impacts in humans and wildlife, can contribute to species-level declines and cause ecosystem-wide impacts; some specific congeners, such as the highly toxic TCDD, are known to have reproductive and developmental impacts on fish (King-Heiden *et al.* 2012). Mitigation measures to reduce the negative impacts of scrubber discharge water fall into three categories: i) avoidance of scrubber discharge; ii) technological advances; and iii) improved regulations, monitoring and enforcement.

3.1 Avoidance of scrubber water discharge

A complete avoidance of the impacts from scrubber water discharge requires use of compliant fuels. Distilled fuels, like Marine Gas Oil (MGO), Liquefied Natural Gas (LNG) or biofuels, have been reported not contain the same toxic combinations as residual fuel oils (Sippula *et al.* 2014, Corbin *et al.* 2020, Lehtoranta *et al.* 2019, Su *et al.* 2019) and are compliant regarding emissions to air, without increasing the impact on the marine environment. In addition to the discharge of scrubber water during normal operation, the use of scrubbers allows the continued carriage of heavy fuel oil on ships, which in the event of accidental fuel spills, are likely to have significant economic and ecological consequences compared to spills of cleaner fuels (Deere-Jones 2016). However, there remains a need for caution regarding the new low sulphur fuel blends, often referred to as hybrid fuels that are compliant with the IMO sulphur regulations. These fuels may contain higher concentrations of contaminants compared to distilled fuels (Takasaki *et al.* 2018, Finland and Germany 2020). Initial tests have also shown that these oils may be non-compatible with available oil spill clean-up equipment (Hellstrøm 2017), which is continuously being investigated in the EU funded project IMAROS (2020).

According to the United Nations Convention on the Law of the Sea (UNCLOS) (United Nations 1982) *Article 195 on Duty not to transfer damage or hazards or transform one type of pollution into another* it is stated that: "In taking measures to prevent, reduce and control pollution of the marine environment, States shall act so as not to transfer, directly or indirectly, damage or hazards from one area to another or transform one type of pollution into another". Further, in accordance with Article 211 (3) UNCLOS, port States have full sovereignty over their ports; i.e. they are free to adopt their own, more stringent regulations or even ban scrubber water discharge (Endres *et al.* 2018). In response to wide-spread concerns around scrubber discharge water, the use of scrubber systems or their discharge has been banned by 28 ports or regions in countries around the world (Nepia 2020).

3.2 Investment in technological advances

In order to reduce the impacts of scrubber discharge water, significant technological advances would be required. A zero discharge CL scrubber system, where all residues are left in port reception facilities would be a hypothetical alternative, yet there are many obstacles to overcome to consider this a realistic option. First, removal of contaminants at the scale of scrubber discharge water production rates is challenging and requires investment in additional equipment and new expertise regarding maintenance. There are a few examples of zero discharge scrubber setups where the CL residues are left ashore, which is possible in some trade routes, for instance, ferries operating on short distances and returning to the same port facilities. If all ships equipped with scrubbers deposited scrubber-related waste in port, it would require a large expansion of port reception facilities and lead to substantially higher costs for the operation of scrubbers. Secondly, additional treatment of scrubber discharge water implies increased costs of chemicals and increased energy consumption (Lindstad and Eskeland 2016). Finally, the risk of accidental heavy fuel oil spills still remains with potentially severe consequences (Deere-Jones 2016). Adequately evaluating the option of new technological advances would require a thorough Life Cycle Assessment, which is beyond the scope of this review.

3.3 Regulations, monitoring and enforcement

During the previous century, the rational for the disposal of waste and hazardous substances in the aquatic environment was that "the solution to pollution is dilution". However, this rational was widely disproved with the advent of modern industrial activities and their use and discharge of toxic chemicals which are largely not biodegradable. Now this outdated concept is being offered in response to the scrubber discharge water concerns. Many of these pollutants are persistent, have the potential to bioaccumulate, and exert toxic potential at very small dosages. These pollutants became legacies that the world oceans and coastal systems already bear, on top of which are the continuous inputs of contaminants and discharges from various sources.

Avoidance of scrubber water discharge is a precautionary, protective measure which reduces the need for extensive monitoring, both on-board and *in situ*, to ensure that the use of scrubbers does not impair the environmental status in areas of intensive shipping. Broad introduction of scrubbers on ships presents a potential exceedance of environmental quality standards, especially for areas with high shipping density (Figure 3). The additional inputs of persistent, bioaccumulating and toxic contaminants from scrubber discharge waters may result in the failure to achieve good environmental status at local and sub-regional scales and to meet the objectives of international agreements and regulations such as Regional Sea Conventions and European Directives (OSPAR, HELCOM, Barcelona Convention, EU MSFD, EU WFD). Updates in environmental research and monitoring programs are required to include assessment and mitigation of ecosystem impacts from the introduction of scrubbers worldwide. In particular, the significance of the contaminant inputs and their impacts should be addressed through cumulative impact assessment methods that consider all other contributing contaminant sources and additional human pressures in a specific area. The few existing reports that claim broad use of scrubbers to be of no concern for the marine environment all omit background concentrations and environmental impacts of other sources in their calculations (e.g. Kjølholt 2012, Japan 2019, MEPC 74/INF.24, Faber *et al.* 2019). Currently available modelling efforts of potential risks of scrubber water discharge in ports e.g. Faber *et al.* (2019) can be improved by modelling high risk ports where there is significant dredging of sediment and use by large cruise ships and container ships, and by further evaluating different sediment type and contaminant load scenarios.

3.3.1 Enforcement of scrubber water discharge limits[9]

Under the International Convention for the Control and Management of Ships' Ballast Water and Sediments (BWMC) (IMO 2004), mandatory discharge standards for the concentration of viable organisms have been developed and enforcement is in place. In contrast to the BWMC, where the discharge standard is included in the core of the convention, in MARPOL Annex VI Regulation 14 (IMO 2008), it is the sulphur content of marine fuels that is regulated through set limit values. A scrubber is defined as an "equivalent method" to be compliant with sulphur content limits in marine fuels, and the EGCS guidelines are focused on the approval of those systems and monitoring systems thereafter; subsequently, enforcement is currently comprised only of assessing whether the system has been approved and is working as indicated for a limited time. Effective mitigation of scrubber impacts needs stringent requirements and standards, monitoring protocols and widespread, effective enforcement, which will also imply increased costs

3.3.2 Revised discharge limits

The increasing use of scrubbers requires updated discharge limits for a number of contaminants present in large quantities in scrubber discharge water. MARPOL Annex VI introduced discharge norms for new waste categories, in particular the residues from scrubbers, including sludge, discharge and bleed-off waters. However, harmonized and generally approved protocols and procedures for the assessment and control of contaminant discharges from these new scrubber waste categories are not yet fully established and agreed upon. There is a complete lack of discharge limits for a number of potentially harmful substances and elements in scrubber discharge water, including large quantities of metals and persistent organic pollutants (POPs) and there is a need to update and revise existing discharge limits for some substances, such as PAHs.

3.3.2.1 Metal pollutants not included in the EGCS guidelines

There are currently no limits in place for metal content in scrubber discharge water despite reported high concentrations of vanadium, nickel, copper and zinc. The high concentrations of heavy metals found in discharge water (see section 1.2) demonstrated that the limit value on turbidity, proposed as an indicator of metal content in the EGCS Guidelines (MEPC 2008b, 2009 and 2015), is not sufficient to protect the environment. There is an urgent need for further and continuous improvement of methodological protocols, revisions of existing limits and establishing new limits for metal content (Bosch *et al.* 2009, MEPC 2015, Linders *et al.* 2019).

3.3.2.2 PAH discharge concentration limit in the EGCS guidelines

The EGCS Guidelines established a discharge criteria defined as PAH phenanthrene equivalent (PAH_{phe}) for PAH concentrations in scrubber discharge water, and as a surrogate for oil residues. The limit value is dependent on the specific discharge water flow rate (t/MWh). The method for PAH_{phe} determination was defined as an optical measurement with ultraviolet light or fluorescence detection by means of an online-sensor installed onboard, allowing continuous monitoring of the dissolved PAH discharge. However, the measurement of PAHs by optical methods has drawbacks. The optical measurement is subject to strong interferences (quenching, scattering of emitted light, etc.), which may be related, for instance, to changing suspended particulate matter

---

[9] Edited following comments from RGSCRUB

and organic matter concentrations. Additionally, optical measurement overlooks PAHs present in particulate form, which could only be measured with frequent sampling and filtering, followed by laboratory coupled gas chromatography-mass spectrometry (GC-MS) analysis.

Furthermore, the PAH_{phe} concept, created and applied exclusively under the EGCS Guidelines of the IMO, is not well defined, which could introduce many flaws and misunderstandings. In practice, almost all PAH_{phe}-equivalency were not summed concentrations of PAH determined by GC-MS analysis (Linders *et al.* 2019). Further, considering the underestimation of PAH in scrubber discharge water reported when using Sum 16 EPA analyses (Figure 5) there is an urgent need to include alkylated PAHs in the analyses of scrubber discharge water.

Finally, the 50 $\mu g*L^{-1}$ PAH discharge limit for scrubbers may not be protective for the marine environment. A rough estimate by Linders *et al.* (2019) showed that if all ships were equipped with OL scrubbers and complied with the PAH discharge limit, their total emissions would be about 10 times higher than worldwide PAH emissions from all sources (all biomass and fossil fuel combustion; Shen *et al.* (2013), Gonzalez-Gaya *et al.* (2016)). Though <10% of the global fleet has installed scrubbers to-date, this calculation indicates that under broad scale use, the current PAH discharge limit does not provide any practical restriction. Therefore, revision is required for the discharge criteria for PAHs and oil discharges in the EGCS Guidelines.

3.3.2.3 Re-evaluate NO_X limits

The removal of NO_X from scrubber discharge water is generally assumed to be <10% (Den Boer and Hoen, 2015), below the current limit set for maximum allowed removal of 12% NO_X in the exhausts by a scrubber. At the IMO MEPC Sub-Committee on Pollution Prevention and Response (PPR) 7th meeting (PPR 7) in February 2020, concerns regarding the difficulty to achieve adequate measurements of NO_X removal, together with reported low values of nitrates in scrubber discharge water, led to suggested exclusion of NO_X limits; however, this did not gain support. It is advisable to continue the evaluation of the NO_X limits, particularly considering that NO_X uptake well below set limits stimulated growth of a microbial plankton community in the Baltic Sea (Koski *et al.* 2017, Ytreberg *et al.* 2019).

3.3.2.4 pH and comparison with ambient water

Although pH is generally considered a standard parameter, it is also important to understand that pH measurements in seawater, especially in areas with salinity gradients, is not a trivial task (Kuliński *et al.* 2017). Schmolke *et al.* (2020) observed deviations on pH measurements carried out on-board with calibrated equipment and the ship online-monitoring data. Although for most of the samples the deviations were below 25%, it is noted that little differences of pH values mean significant changes as pH is based on a logarithmic scale.

Beside the analytical challenges to make accurate pH measurements, there is also an exception criteria in the current EGCS guidelines that may be prone to bias. According to the guidelines, scrubber discharge water should have a pH of no less than 6.5 measured at the ship's overboard discharge. However, there is an exception that during maneuvering and transit, a maximum difference of 2 pH units is allowed between measurements at the ship's inlet and overboard discharge. If many ships are operating scrubbers in a confined area, the inlet pH may already be lower than the natural ambient pH. Thus, using comparative inlet and outlet values, rather than a minimum standard, may give a false impression that it is acceptable to discharge water of even lower pH.

3.3.3 Need for transparent, well-defined sampling and reporting protocols

Enforcement of regulations and limits requires effective and efficient sampling and reporting protocols. Studies are needed to better understand scrubber effectiveness (mainly SO_X removal) and the transfer of contaminants from scrubber discharge water to the marine environment. Improved evaluation and full chemical characterization of contaminants, eutrophying and acidifying substances discharged by scrubbers is essential in this context and urgently needed. Existing sampling protocols are incomplete and may introduce considerable bias in the quantification of the contaminant discharge. For instance, a number of reports that evaluate contaminant discharge by OL scrubbers subtract contaminant concentrations in the inlet seawater from concentrations in the outlet water before discharge. Inlet seawater concentrations have been incorrectly assumed to be the natural background concentrations for the area where the ship operates. However, as for pH mentioned previously, the contaminant concentrations in the inlet samples are influenced by other discharges to the environment, including from scrubbers of all ships operating in the area. Moreover, inlet samples are often collected after passage through the onboard pumps and may be contaminated by the ship's lubricants and metallic pipes (i.e., copper-containing antifouling paints in the sea chests and cathodic pipe protection systems). This portion of contaminants, though not directly related to the scrubber process, would not be discharged into the marine environment if scrubbers were not used; therefore, they should not be regarded as background contaminants from the surrounding environment. The mass balance approach, with mandatory sampling and reporting of chemical characterization of inlet water, scrubber discharge water, fuel and lubricants, along with data on water flows, and engine load, for better quantification of contaminant discharge, should be further developed and applied (Linders *et al.* 2019).

4 Conclusion

Transferring contaminants from air emissions to the ocean does not mitigate their impact and instead, the use of scrubber systems is creating an emerging global problem. The growing use of scrubbers by ships to meet the reduced sulphur emission limits will yield significant amounts of acidic and contaminated scrubber discharge water. Scrubber discharge water is documented to comprise a cocktail of heavy metals, PAHs and other organic compounds which have not yet been identified. This mixture has demonstrated the potential for substantial toxic effects in laboratory studies, causing immediate mortality in plankton and exhibiting negative synergistic effects. The substances found in scrubber discharge water are likely to have further impacts through bioaccumulation, acidification and eutrophication in the marine environment. While a single ship with an installed scrubber may pose limited, local risk to marine ecosystem health, a global shipping community employing scrubbers to meet air emission limits is of serious concern. The impacts of scrubber discharge water can be completely avoided through the use of alternative fuels, such as distilled low sulphur fuels. Distilled fuels have the added benefit that they remove the threat of heavy fuel oil spills from shipping activities. If the use of distilled fuels is not adopted, then there is urgent need for:

1) significant investment in technological advances and port reception facilities to allow zero discharge closed loop scrubber systems;
2) improved protocols and standards for measuring, monitoring and reporting on scrubber discharge water acidity and pollutants;
3) evidence-based regulations on scrubber water discharge limits that consider the full suite of contaminants.

References

Abadie, L. M., N. Goicoechea and I. Galarraga (2017). Adapting the shipping sector to stricter emissions regulations: Fuel switching or installing a scrubber? *Transportation Research Part D: Transport and Environment* 57: 237-250.

Allen, J. O., J. L. Durant, N. M. Dookeran, K. Taghizadeh, E. F. Plummer, A. L. Lafleur, A. F. Sarofim and K. A. Smith (1998). Measurement of $C_{24}H_{14}$ polycyclic aromatic hydrocarbons associated with a size-segregated urban aerosol. *Environmental Science & Technology* 32(13): 1928-1932.

Artioli, Y., J. C. Blackford, M. Butenschon, J. T. Holt, S. L. Wakelin, H. Thomas, A. V. Borges and J. I. Allen (2012). The carbonate system in the North Sea: Sensitivity and model validation. *Journal of Marine Systems* 102: 1-13.

Barata, C., A. Calbet, E. Saiz, L. Ortiz and J. M. Bayona (2005). Predicting single and mixture toxicity of petrogenic polycyclic aromatic hydrocarbons to the copepod Oithona davisae. *Environmental Toxicology and Chemistry* 24(11): 2992-2999.

Bartnicki, J. and A. Benedictow (2017). Contributions of emissions from different countries and sectors to atmospheric nitrogen input to the Baltic Sea basin and its sub-basins. EMEP/MSC-W report for HELCOM. EMEP/MSC-W TECHNICAL REPORT 2/2017. Oslo. (ISSN 0332-9879). pp: 1-88.

Bates, N. R., Y. M. Astor, M. J. Church, K. Currie, J. E. Dore, M. Gonzalez-Davila, L. Lorenzoni, F. Muller-Karger, J. Olafsson and J. M. Santana-Casiano (2014). A Time-Series View of Changing Surface Ocean Chemistry Due to Ocean Uptake of Anthropogenic CO2 and Ocean Acidification. *Oceanography* 27(1): 126-141.

Battuello, M., P. Brizio, R. M. Sartor, N. Nurra, D. Pessani, M. C. Abete and S. Squadrone (2016). Zooplankton from a North Western Mediterranean area as a model of metal transfer in a marine environment. *Ecological Indicators* 66: 440-451.

Beare, D., A. McQuatters-Gollop, T. van der Hammen, M. Machiels, S. J. Teoh and J. M. Hall-Spencer (2013). Long-Term Trends in Calcifying Plankton and pH in the North Sea. *Plos One* 8(5): 10.

Berglund, O., P. Larsson, G. Ewald and L. Okla (2000). Bioaccumulation and differential partitioning of polychlorinated biphenyls in freshwater, planktonic food webs. *Canadian Journal of Fisheries and Aquatic Sciences* 57(6): 1160-1168.

Billiard, S. M., J. N. Meyer, D. M. Wassenberg, P. V. Hodson and R. T. Di Giulio (2008). Nonadditive effects of PAHs on early vertebrate development: Mechanisms and implications for risk assessment. *Toxicological Sciences* 105(1): 5-23.

Bindoff, N., W. Cheung, J. Kairo, J. Arístegui, V. Guinder, R. Hallberg, N. Hilmi, N. Jiao, M. Karim, L. Levin, S. O'Donoghue, S. Purca Cuicapusa, B. Rinkevich, T. Suga, A. Tagliabue and P. Williamson (2019). Changing Ocean, Marine Ecosystems, and Dependent Communities. In: IPCC Special Report on the Ocean and Cryosphere in a Changing Climate. pp: 447-588.

Borch, T., R. Kretzschmar, A. Kappler, P. Van Cappellen, M. Ginder-Vogel, A. Voegelin and K. Campbell (2010). Biogeochemical Redox Processes and their Impact on Contaminant Dynamics. *Environmental Science & Technology* 44(1): 15-23.

Borja, A., M. Elliott, M. C. Uyarra, J. Carstensen and M. Mea (2017). Editorial: Bridging the Gap between Policy and Science in Assessing the Health Status of Marine Ecosystems. *Frontiers in Marine Science* 4: 3.

Borja, A., V. Valencia, J. Franco, I. Muxika, J. Bald, M. J. Belzunce and O. Solaun (2004). The water framework directive: water alone, or in association with sediment and biota, in determining quality standards? *Marine Pollution Bulletin* 49(1-2): 8-11.

Bosch, A. C., B. O'Neill, G. O. Sigge, S. E. Kerwath and L. C. Hoffman (2016). Heavy metals in marine fish meat and consumer health: a review. *Journal of the Science of Food and Agriculture* 96(1): 32-48.

Bosch, P., P. Coenen, E. Fridell, S. Åström, T. Palmer and M. Holland (2009). Cost Benefit Analysis to Support the Impact Assessment accompanying the revision of Directive 1999/32/EC on the Sulphur Content of certain Liquid Fuels. Final report ED45756 by AEA to the European Commission. pp: 1-169.

Breitburg, D. L., J. G. Sanders, C. C. Gilmour, C. A. Hatfield, R. W. Osman, G. F. Riedel, S. B. Seitzinger and K. G. Sellner (1999). Variability in responses to nutrients and trace elements, and transmission of stressor effects through an estuarine food web. *Limnology and Oceanography* 44(3): 837-863.

British Ports Association. (2019). "Ports' Open-Loop Scrubber Concerns Must Be Addressed." Retrieved 30 July, 2020, from https://www.britishports.org.uk/news/bpa-ports-open-loop-scrubber-concerns-must-be-addressed.

Buhaug, Ø., H. Fløgstad and T. Bakke (2006). MARULS WP3: Washwater Criteria for seawater exhaust gas-SOx scrubbers. Submitted by United States to MEPC 56 as document MEPC 56/INF.5. International Maritime Organization. pp: 1-67.

Calbet, A., C. Schmoker, F. Russo, A. Trottet, M. S. Mahjoub, O. Larsen, H. Y. Tong and G. Drillet (2016). Non-proportional bioaccumulation of trace metals and metalloids in the planktonic food web of two Singapore coastal marine inlets with contrasting water residence times. *Science of the Total Environment* 560: 284-294.

Canli, M. and G. Atli (2003). The relationships between heavy metal (Cd, Cr, Cu, Fe, Pb, Zn) levels and the size of six Mediterranean fish species. *Environmental Pollution* 121(1): 129-136.

Carnival Corporation & PLC and DNV-GL (2019). Compilation and Assessment of Lab Samples from EGCS Washwater Discharge on Carnival Ships.

CE Delft and CHEW (2017). The Management of Ship-Generated Waste On-board Ships. Publication code 16.7I85.130. Project EMSA/OP/02/2016. pp: 1-90.

Chouvelon, T., E. Strady, M. Harmelin-Vivien, O. Radakovitch, C. Brach-Papa, S. Crochet, J. Knoery, E. Rozuel, B. Thomas, J. Tronczynski and J.-F. Chiffoleau (2019). Patterns of trace metal bioaccumulation and trophic transfer in a phytoplankton-zooplankton-small pelagic fish marine food web. *Marine Pollution Bulletin* 146: 1013-1030.

Corbett, J. J., C. Wang, J. J. Winebrake and E. Green (2007). Allocation and forecasting of global ship emissions. Prepared for the Clean Air Task Force and Friends of the Earth International: Boston, MA, USA. pp: 1-27.

Corbin, J. C., W. H. Peng, J. C. Yang, D. E. Sommer, U. Trivanovic, P. Kirchen, J. W. Miller, S. Rogak, D. R. Cocker, G. J. Smallwood, P. Lobo and S. Gagne (2020). Characterization of particulate matter emitted by a marine engine operated with liquefied natural gas and diesel fuels. *Atmospheric Environment* 220: 11.

COWI (2013). Assessment of possible impacts of scrubber water discharges on the marine environment - supplementary note. Danish Environmental Protection Agency. pp: 1-5.

Dachs, J., S. J. Eisenreich, J. E. Baker, F. C. Ko and J. D. Jeremiason (1999). Coupling of phytoplankton uptake and air-water exchange of persistent organic pollutants. *Environmental Science & Technology* 33(20): 3653-3660.

De Guise, S., J. Bernier, D. Martineau, P. Beland and M. Fournier (1996). Effects of in vitro exposure of beluga whale splenocytes and thymocytes to heavy metals. *Environmental Toxicology and Chemistry* 15(8): 1357-1364.

Deere-Jones, T. (2016). Ecological, economic, and social cost of marine/coastal spills of fuel oils (refinery residuals). Report for the European Climate Foundation. pp: 1-44.

DelValls, T. A., A. Andres, M. J. Belzunce, J. Buceta, M. C. Casado-Martinez, R. Castro, I. Riba, J. R. Viguri and J. Blasco (2004). Chemical and ecotoxicological guidelines for managing disposal of dredged material. *Trac-Trends in Analytical Chemistry* 23(10-11): 819-828.

Den Boer, E. and M. t. Hoen (2015). Scrubbers – An economic and ecological assessment. Delft, CE Delft. pp: 1-45.

Desforges, J. P. W., C. Sonne, M. Levin, U. Siebert, S. De Guise and R. Dietz (2016). Immunotoxic effects of environmental pollutants in marine mammals. *Environment International* 86: 126-139.

Diaz, R. J. and R. Rosenberg (2008). Spreading dead zones and consequences for marine ecosystems. *Science* 321(5891): 926-929.

Doney, S. C., V. J. Fabry, R. A. Feely and J. A. Kleypas (2009). Ocean Acidification: The Other CO_2 Problem. *Annual Review of Marine Science* 1: 169-192.

Doney, S. C., N. Mahowald, I. Lima, R. A. Feely, F. T. Mackenzie, J. F. Lamarque and P. J. Rasch (2007). Impact of anthropogenic atmospheric nitrogen and sulfur deposition on ocean acidification and the inorganic carbon system. *Proceedings of the National Academy of Sciences of the United States of America* 104(37): 14580-14585.

Dulière, V., K. Baetens and G. Lacroix (2020). Potential impact of wash water effluents from scrubbers on water acidification in the southern North Sea. pp: 1-31.

Durant, J. L., A. L. Lafleur, E. F. Plummer, K. Taghizadeh, W. F. Busby and W. G. Thilly (1998). Human lymphoblast mutagens in urban airborne particles. *Environmental Science & Technology* 32(13): 1894-1906.

EC (2000). The EU Water Framework Directive - integrated river basin management for Europe. 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. Off. J. Eur. Union L 327. pp: 1-73.

EC (2008). The EU Marine Strategy Framework Directive. Directive 2008/56/EC of the European Parliament and of the Council establishing a framework for community action in the field of marine environmental policy, Off. J. Eur. Union L164. pp: 19-40.

EC (2016). Note to the attention of the members of the European Sustainable Shipping Forum. Commission's views on the discharge of scrubber wash water and the updated table summarising the position of Member States on the acceptability of discharges of scrubber wash water - Agenda item 6.C ESSF of 26/1/2016. Directorate-General Environment. Directorate C - Quality of Life, Water & Air. Unit C.1 - Water and Unit C.3 - Air. Ref. Ares(2016)254855 - 18/01/2016. pp: 1-12.

EC (2017). Commission Decision (EU) 2017/848 of 17 May 2017 laying down criteria and methodological standards on good environmental status of marine waters and specifications and standardised methods for monitoring and assessment, and repealing Decision 2010/477/EU, Off. J. Eur. Communities L125. pp: 43-74.

EC (2019). Directive (EU) 2019/883 of the European Parliament and of the Council of 17 April 2019 on port reception facilities for the delivery of waste from ships, amending Directive 2010/65/EU and repealing Directive 2000/59/EC. Off. J. Eur. Union L 151. pp: 1-116.

Echeveste, P., S. Agusti and J. Dachs (2011). Cell size dependence of additive versus synergetic effects of UV radiation and PAHs on oceanic phytoplankton. *Environmental Pollution* 159(5): 1307-1316.

Echeveste, P., S. Agusti and A. Tovar-Sanchez (2012). Toxic thresholds of cadmium and lead to oceanic phytoplankton: Cell size and ocean basin-dependent effects. *Environmental Toxicology and Chemistry* 31(8): 1887-1894.

EGCSA (2012). A practical guide to exhaust gas cleaning systems for the maritime industry. EGCSA Handbook 2012. pp: 1-190.

EGCSA and Euroshore (2018). Report on Analyses of Water Samples from Exhaust Gas Cleaning Systems. Submitted by CESA to MEPC 73/INF.5. . IMO. pp: 39 pp.

EMERGE (2020). Evaluation, control and Mitigation of the EnviRonmental impacts of shippinG Emissions, EU Horizon 2020 research and innovation programme under grant agreement No 874990.

Endres, S., F. Maes, F. Hopkins, K. Houghton, E. M. Martensson, J. Oeffner, B. Quack, P. Singh and D. Turner (2018). A New Perspective at the Ship-Air-Sea-Interface: The Environmental Impacts of Exhaust Gas Scrubber Discharge. *Frontiers in Marine Science* 5(139).

European Environment Agency (2001). Late lessons from early warnings: the precautionary principle 1896–2000. Environmental issue report No 22. Luxembourg: Office for Official Publications of the European Communities. (ISBN 92-9167-323-4). pp: 1-210.

Eyring, V., H. W. Köhler, J. van Aardenne and A. Lauer (2005). Emissions from international shipping: 1. The last 50 years. *Journal of Geophysical Research-Atmospheres* 110(D17): 12.

Faber, J., D. Nelissen, T. Huigen, H. Shanti, B. van Hattum and F. Kleissen (2019). The impacts of EGCS washwater discharges on port water and sediment. Submitted to PPR 7 by Cruise Lines International Association (CLIA) Europe as document PPR 7 INF.18. Delft, CE Delft. pp: 1-62.

Finland and Germany (2020). Initial results of a Black Carbon measurement campaign with emphasis on the impact of the fuel oil quality on Black Carbon emissions. Submitted by Finland and Germany to PPR 7 as document PPR 7/8. International Maritime Organization. pp: 1-9.

Fisher, N. S., I. Stupakoff, S. Sanudo-Wilhelmy, W. X. Wang, J. L. Teyssie, S. W. Fowler and J. Crusius (2000). Trace metals in marine copepods: a field test of a bioaccumulation model coupled to laboratory uptake kinetics data. *Marine Ecology Progress Series* 194: 211-218.

Frouin, H., N. Dangerfield, R. W. Macdonald, M. Galbraith, N. Crewe, P. Shaw, D. Mackas and P. S. Ross (2013). Partitioning and bioaccumulation of PCBs and PBDEs in marine plankton from the Strait of Georgia, British Columbia, Canada. *Progress in Oceanography* 115: 65-75.

Galban-Malagon, C. J., N. Berrojalbiz, R. Gioia and J. Dachs (2013a). The "Degradative" and "Biological" Pumps Controls on the Atmospheric Deposition and Sequestration of Hexachlorocyclohexanes and Hexachlorobenzene in the North Atlantic and Arctic Oceans. *Environmental Science & Technology* 47(13): 7195-7203.

Galban-Malagon, C. J., S. Del Vento, N. Berrojalbiz, M. J. Ojeda and J. Dachs (2013b). Polychlorinated Biphenyls, Hexachlorocyclohexanes and Hexachlorobenzene in Seawater and Phytoplankton from the

Southern Ocean (Weddell, South Scotia, and Bellingshausen Seas). *Environmental Science & Technology* 47(11): 5578-5587.

Georgeff, E., X. Mao and B. Comer (2019). A whale of a problem? Heavy fuel oil exhaust gas cleaning systems and British Columbias resident killer whales. Submitted to the PPR 7 meeting by FOEI, WWF and Pacific Environment as document PPR 7/INF.22. International Council on Clean Transportation.

Germany (2018). Results from a German project on washwater from exhaust gas cleaning systems. Submitted to PPR 6 as document PPR 6/INF.20. IMO. pp: 1-16.

Gobas, F. A., W. de Wolf, L. P. Burkhard, E. Verbruggen and K. Plotzke (2009). Revisiting Bioaccumulation Criteria for POPs and PBT Assessments. *Integrated Environmental Assessment and Management* 5(4): 624-637.

Gonzalez-Gaya, B., M. C. Fernandez-Pinos, L. Morales, L. Mejanelle, E. Abad, B. Pina, C. M. Duarte, B. Jimenez and J. Dachs (2016). High atmosphere-ocean exchange of semivolatile aromatic hydrocarbons. *Nature Geoscience* 9(6): 438-444.

Grundl, T. J., S. Haderlein, J. T. Nurmi and P. G. Tratnyek (2011). Introduction to Aquatic Redox Chemistry. *Aquatic Redox Chemistry*. P. G. Tratnyek, T. J. Grundl and S. B. Haderlein. Washington, Amer Chemical Soc. 1071**:** 1-14.

Hagens, M., K. A. Hunter, P. S. Liss and J. J. Middelburg (2014). Biogeochemical context impacts seawater pH changes resulting from atmospheric sulfur and nitrogen deposition. *Geophysical Research Letters* 41(3): 935-941.

Hallanger, I. G., N. A. Warner, A. Ruus, A. Evenset, G. Christensen, D. Herzke, G. W. Gabrielsen and K. Borga (2011). Seasonality in contaminant accumulation in Arctic marine pelagic food webs using trophic magnification factor as a measure of bioaccumulation. *Environmental Toxicology and Chemistry* 30(5): 1026-1035.

Hassellöv, I.-M., D. R. Turner, A. Lauer and J. J. Corbett (2013). Shipping contributes to ocean acidification. *Geophysical Research Letters* 40(11): 2731-2736.

HELCOM (2018). State of the Baltic Sea – Second HELCOM holistic assessment 2011-2016. Baltic Sea Environment Proceedings 155. (ISSN 0357-2994). pp: 1-155.

Hellstrøm, K. C. (2017). Weathering Properties and Toxicity of Marine Fuel Oils. Prepared for Kystverket (Norweigian Coastal Administration). SINTEF. pp: 1-82.

Heywood, J. B. and E. Kasseris (2019). MEPC 74/INF.10 EGCS Environmental Impact Literature Review. Submitted by Panama. IMO. pp: 21 pp.

Human Environment and Transport Inspectorate, T. N. (2018). Heavy fuel oil for seagoing vessels. On-road fuels for West Africa. Blended in the Netherlands. The Hague. pp: 1-48.

Hunter, K. A., P. S. Liss, V. Surapipith, F. Dentener, R. Duce, M. Kanakidou, N. Kubilay, N. Mahowald, G. Okin, M. Sarin, M. Uematsu and T. Zhu (2011). Impacts of anthropogenic SOx, NOx and NH3 on acidification of coastal waters and shipping lanes. *Geophysical Research Letters* 38: 6.

IMAROS (2020). EU UCPM Improving response capacities and understanding the environmental impacts of new generation low sulphur MARine fuel Oil Spills. Project ID 874387, DG/Agency: ECHO.

IMO (2004). International Convention for the Control and Management of Ships' Ballast Water and Sediments (BWMC). Londonpp: 1-43.

IMO (2008). MARPOL 2008 Amendments to the Annex of Protocol of 1997 to amend the International Convention for the Prevention of Pollution from Ships, 1973, as modified by the protocol of 1978. (Revised MARPOL Annex VI) (Resolution MEPC.176(58)).

Incardona, J. P., T. K. Collier and N. L. Scholz (2004). Defects in cardiac function precede morphological abnormalities in fish embryos exposed to polycyclic aromatic hydrocarbons. *Toxicology and Applied Pharmacology* 196(2): 191-205.

IOC (2019). The Science we Need for the Ocean We Want: The United Nations Decade of Ocean Science for Sustainable Development (2021-2030). Paris. pp: 1-24.

Jalkanen, J.-P. and L. Johansson (2019). Discharges to the sea from Baltic Sea shipping in 2006-2018. Submitted by Finland to the Baltic Marine Environment Protection Commission HELCOM MARITIME 19-2019, as document INF 13-4. pp: 1-18.

Japan (2019). Proposal on the refinement of the title for a new output and the development of the guidelines for evaluation and harmonization of developing local rules on discharge of liquid effluents from EGCS into sensitive waters. Submitted by Japan to PPR 7 as document PPR 7/12/3. London. International Maritime Organization. pp: 1-5.

Jezierska, B., K. Lugowska and M. Witeska (2009). The effects of heavy metals on embryonic development of fish (a review). *Fish Physiology and Biochemistry* 35(4): 625-640.

Karle, I.-M. and D. R. Turner (2007). Seawater Scrubbing - reduction of SOx emissions from ship exhausts. Alliance of Global Sustainability. Gothenburg, Sweden. (ISBN: 978-91-976534-1-1). pp: 1-28.

Kathmann *et al.* (in prep.).

Kennedy, A. J., T. W. Biber, L. R. May, G. R. Lotufo, J. D. Farrar and A. J. Bednar (2019). Sensitivity of the Marine Calanoid Copepod Pseudodiaptomus pelagicus to Copper, Phenanthrene, and Ammonia. *Environmental Toxicology and Chemistry* 38(6): 1221-1230.

King-Heiden, T. C., V. Mehta, K. M. Xiong, K. A. Lanham, D. S. Antkiewicz, A. Ganser, W. Heideman and R. E. Peterson (2012). Reproductive and developmental toxicity of dioxin in fish. *Molecular and Cellular Endocrinology* 354(1-2): 121-138.

Kjølholt, J. S., S. Aakre, C. Jürgensen and J. Lauridsen (2012). Assessment of possible impacts of scrubber water discharges on the marine environment. Prepared for Danish Ministry of the Environment. Environmental Protection Agency. Environmental project 1431. pp: 1-93.

Koski, M., C. Stedmon and S. Trapp (2017). Ecological effects of scrubber water discharge on coastal plankton: Potential synergistic effects of contaminants reduce survival and feeding of the copepod Acartia tonsa. *Marine Environmental Research* 129: 374-385.

Kuliński, K., B. Schneider, B. Szymczycha and M. Stokowski (2017). Structure and functioning of the acid-base system in the Baltic Sea. *Earth System Dynamics* 8(4): 1107-1120.

Lam, M. M., R. Bulow, M. Engwall, J. P. Giesy and M. Larsson (2018). Methylated PACs Are More Potent Than Their Parent Compounds: A Study of Aryl Hydrocarbon Receptor-Mediated Activity, Degradability, and Mixture Interactions in the H4IIE-luc Assay. *Environmental Toxicology and Chemistry* 37(5): 1409-1419.

Lange, B., T. Markus and L. P. Helfst (2015). Impacts of scrubbers on the environmental situation in ports and coastal waters. TEXTE 65/2015. Dessau-Roßlau. (UBA-FB) 002015/E). pp: 1-88.

Lavery, T. J., C. M. Kemper, K. Sanderson, C. G. Schultz, P. Coyle, J. G. Mitchell and L. Seuront (2009). Heavy metal toxicity of kidney and bone tissues in South Australian adult bottlenose dolphins (Tursiops aduncus). *Marine Environmental Research* 67(1): 1-7.

Lehtoranta, K., P. Aakko-Saksa, T. Murtonen, H. Vesala, L. Ntziachristos, T. Ronkko, P. Karjalainen, N. Kuittinen and H. Timonen (2019). Particulate Mass and Nonvolatile Particle Number Emissions from Marine Engines Using Low-Sulfur Fuels, Natural Gas, or Scrubbers. *Environmental Science & Technology* 53(6): 3315-3322.

Linders, J., E. Adams, B. Behrends, A. Dock, S. Hanayama, R. Luit, C. Rouleau and J. Tronczynski (2019). Exhaust Gas Cleaning Systems – A roadmap to risk assessment. Report of the GESAMP Task Team on exhaust gas cleaning systems. Submitted to PPR 7 as document PPR 7/INF.23. London. IMO. pp: 1-121.

Lloyd's Register (2012). Understanding exhaust gas treatment systems – Guidance for shipowners and operators. pp: 1-56.

Logan, D. T. (2007). Perspective on ecotoxicology of PAHs to fish. *Human and Ecological Risk Assessment* 13(2): 302-316.

Machovsky-Capuska, G. E., G. von Haeften, M. A. Romero, D. H. Rodriguez and M. S. Gerpe (2020). Linking cadmium and mercury accumulation to nutritional intake in common dolphins (Delphinus delphis) from Patagonia, Argentina. *Environmental Pollution* 263: 8.

Magnusson, K., P. Thor and M. Granberg (2018). Risk Assessment of marine exhaust gas EGCS water, Task 2, Activity 3, EGCSs closing the loop. IVL Swedish Environmental Research Institute. pp: 1-44.

Marin-Enriquez, O., K. Ewert and A. Krutwa (2020). Environmental Impacts of Exhaust Gas Cleaning Systems for Reduction of SO_x on Ships – Analysis of status quo. Interim report compiled within the framework of the project ImpEx (WP 1). Not published.

MEPC (1998). MEPC 41/WP.5 Agenda item 8. Prevention of air pollution from ships. Report by the Drafting Group. London. International Maritime Organization. pp: 1-8.

MEPC (2008a). MEPC 58/23. Report of the MEPC on its 58th session. Agenda Item 23. pp: 1-274.

MEPC (2008b). Resolution MEPC.170(57) Annex 4. 2008 Guidelines For Exhaust Gas Cleaning Systems. pp: 1-23.

MEPC (2009). Resolution MEPC.184(59) Annex 9. 2009 Guidelines For Exhaust Gas Cleaning Systems. pp: 1-24.

MEPC (2015). Resolution MEPC.259(68) Annex 1. 2015 Guidelines for Exhaust Gas Cleaning Systems. pp: 1-23.

Mohammed, E. H., G. Z. Wang and J. L. Jiang (2010). The effects of nickel on the reproductive ability of three different marine copepods. *Ecotoxicology* 19(5): 911-916.

Monteiro, S. S., M. Bozzetti, J. Torres, A. S. Tavares, M. Ferreira, A. T. Pereira, S. Sa, H. Araujo, J. Bastos-Santos, I. Oliveira, J. V. Vingada and C. Eira (2020). Striped dolphins as trace element biomonitoring tools in oceanic waters: Accounting for health-related variables. *Science of the Total Environment* 699: 9.

Monteiro, S. S., A. T. Pereira, E. Costa, J. Torres, I. Oliveira, J. Bastos-Santos, H. Araujo, M. Ferreira, J. Vingada and C. Eira (2016). Bioaccumulation of trace element concentrations in common dolphins (Delphinus delphis) from Portugal. *Marine Pollution Bulletin* 113(1-2): 400-407.

Nepia. (2020). "North of England P&I Association Limited. No Scrubs: More Ports Declare Ban on EGCS Discharges *Update*." Retrieved June 8th 2020, from https://www.nepia.com/industry-news/no-scrubs-more-ports-declare-ban-on-egcs-discharges-update/.

Omstedt, A., M. Edman, B. Claremar and A. Rutgersson (2015). Modelling the contributions to marine acidification from deposited SOx, NOx, and NHx in the Baltic Sea: Past and present situations. *Continental Shelf Research* 111: 234-249.

OSPAR (2018). Discharges, Spills and Emissions from Offshore Oil and Gas Installations in 2016. pp: 1-52.

Parmentier, K. F. V., Y. Verhaegen, B. P. De Witte, S. Hoffman, D. H. R. Delbare, P. M. Roose, K. D. E. Hylland, T. Burgeot, G. J. Smagghe and K. Cooreman (2019). Tributyltin: A Bottom–Up Regulator of the Crangon crangon Population? *Frontiers in Marine Science* 6(633).

Raudsepp, U., I. Maljutenko, M. Kouts, L. Granhag, M. Wilewska-Bien, I.-M. Hassellöv, K. M. Eriksson, L. Johansson, J.-P. Jalkanen, M. Karl, V. Matthias and J. Moldanova (2019). Shipborne nutrient dynamics and impact on the eutrophication in the Baltic Sea. *Science of the Total Environment* 671: 189-207.

Rudén, C. (2019). Future chemical risk management. Accounting for combination effects and assessing chemicals in groups. pp: 1-258.

Santana, M. S., L. Sandrini-Neto, F. F. Neto, C. A. O. Ribeiro, M. Di Domenico and M. M. Prodocimo (2018). Biomarker responses in fish exposed to polycyclic aromatic hydrocarbons (PAHs): Systematic review and meta-analysis. *Environmental Pollution* 242: 449-461.

Schmolke, S., K. Ewert, M. Kaste, T. Schöngaßner, T. Kirchgeorg and O. Marin-Enriquez (2020). Environmental Protection in Maritime Traffic – Scrubber Wash Water Survey. Final Report. UBA Texte. Dessau-Roßlau. Texte 162/2020. German Environment Agency. ISSN 1862-4804. pp:1-97.

Sellner, K. G., G. J. Doucette and G. J. Kirkpatrick (2003). Harmful algal blooms: causes, impacts and detection. *Journal of Industrial Microbiology & Biotechnology* 30(7): 383-406.

Shen, H. Z., Y. Huang, R. Wang, D. Zhu, W. Li, G. F. Shen, B. Wang, Y. Y. Zhang, Y. C. Chen, Y. Lu, H. Chen, T. C. Li, K. Sun, B. G. Li, W. X. Liu, J. F. Liu and S. Tao (2013). Global Atmospheric Emissions of Polycyclic Aromatic Hydrocarbons from 1960 to 2008 and Future Predictions. *Environmental Science & Technology* 47(12): 6415-6424.

Sippula, O., B. Stengel, M. Sklorz, T. Streibel, R. Rabe, J. Orasche, J. Lintelmann, B. Michalke, G. Abbaszade, C. Radischat, T. Gröger, J. Schnelle-Kreis, H. Harndorf and R. Zimmermann (2014). Particle Emissions from a Marine Engine: Chemical Composition and Aromatic Emission Profiles under Various Operating Conditions. *Environmental Science & Technology* 48(19): 11721-11729.

Smith, T. W. P., J. P. Jalkanen, B. A. Anderson, J. J. Corbett, J. Faber, S. Hanayama, E. O'Keeffe, S. Parker, L. Johansson, L. Aldous, C. Raucci, M. Traut, S. Ettinger, D. Nelissen, D. S. Lee, S. Ng, A. Agrawal, J. J. Winebrake, M. Hoen, S. Chesworth and A. Pandey (2014). Third IMO GHG Study 2014. International Maritime Organization (IMO) London, UK. pp: 1-327.

Smith, V. H. and D. W. Schindler (2009). Eutrophication science: where do we go from here? *Trends in Ecology & Evolution* 24(4): 201-207.

Stips, A., K. Bolding, D. Macías, J. Bruggeman and C. Eayrs (2016). Scoping report on the potential impact of on-board desulphurisation on the water quality in SOx Emission Control. pp: 1-61.

Strady, E., M. Harmelin-Vivien, J. F. Chiffoleau, A. Veron, J. Tronczynski and O. Radakovitch (2015). Po-210 and Pb-210 trophic transfer within the phytoplankton-zooplankton -anchovy/sardine food web: a case study from the Gulf of Lion (NW Mediterranean Sea). *Journal of Environmental Radioactivity* 143: 141-151.

Su, P. H., P. Geng, L. J. Wei, C. Y. Hou, F. Yin, G. T. Tomy, Y. F. Li and D. L. Feng (2019). PM and PAHs emissions of ship auxiliary engine fuelled with waste cooking oil biodiesel and marine gas oil. *Iet Intelligent Transport Systems* 13(1): 218-227.

Sun, Y., C. A. Miller, T. E. Wiese and D. A. Blake (2014). Methylated phenanthrenes are more potent than phenanthrene in a bioassay of human aryl hydrocarbon receptor (AhR) signaling. *Environmental Toxicology and Chemistry* 33(10): 2363-2367.

Takasaki, K., D. Tsuru, C. Takahashi and T. Takaishi (2018). *Combustion Quality of low-sulphur Marine Fuels after 2020 – will be better or worse?* , Rostock.

Teuchies, J., T. J. S. Cox, K. Van Itterbeeck, F. J. R. Meysman and R. Blust (2020). The impact of scrubber discharge on the water quality in estuaries and ports. *Environmental Sciences Europe* 32(1): 103.

Thompson, B., T. Adelsbach, C. Brown, J. Hunt, J. Kuwabara, J. Neale, H. Ohlendorf, S. Schwarzbach, R. Spies and K. Taberski (2007). Biological effects of anthropogenic contaminants in the San Francisco Estuary. *Environmental Research* 105(1): 156-174.

Tiano, M., J. Tronczynski, M. Harmelin-Vivien, C. Tixier and F. Carlotti (2014). PCB concentrations in plankton size classes, a temporal study in Marseille Bay, Western Mediterranean Sea. *Marine Pollution Bulletin* 89(1-2): 331-339.

Tlili, S., J. Ovaert, A. Souissi, B. Ouddane and S. Souissi (2016). Acute toxicity, uptake and accumulation kinetics of nickel in an invasive copepod species: Pseudodiaptomus marinus. *Chemosphere* 144: 1729-1737.

Turley, C. and J. P. Gattuso (2012). Future biological and ecosystem impacts of ocean acidification and their socioeconomic-policy implications. *Current Opinion in Environmental Sustainability* 4(3): 278-286.

Turner, D. R., M. Edman, J. A. Gallego-Urrea, B. Claremar, I.-M. Hassellöv, A. Omstedt and A. Rutgersson (2018). The potential future contribution of shipping to acidification of the Baltic Sea. *Ambio* 47(3): 368-378.

Turner, D. R., I. M. Hassellöv, E. Ytreberg and A. Rutgersson (2017). Shipping and the environment: Smokestack emissions, scrubbers and unregulated oceanic consequences. *Elementa-Science of the Anthropocene* 5.

UN General Assembly (2015). A/RES/70/1 Transforming our world: the 2030 Agenda for Sustainable Development. Resolution adopted by the General Assembly on 25 September 2015. pp: 1-35.

United Nations (1982). United Nations Convention on the Law of the Sea (UNCLOS). pp: 1-202.

United States (2003). Guidelines on on-board exhaust gas cleaning systems. Submitted by the United States to the Sub-committee on ship design and equipment 47th session, Agenda item 20, as document DE 47/20. London. International Maritime Organization. pp: 1-7.

US EPA (2011). Exhaust Gas Scrubber Washwater Effluent. United States Environmental Protection Agency. Office of Wastewater Management Washington, DC. pp: 1-46.

Ushakov, S., D. Stenersen, P. M. Einang and T. Ø. Ask (2020). Meeting future emission regulation at sea by combining low-pressure EGR and seawater scrubbing. *Journal of Marine Science and Technology* 25(2): 482-497.

Wang, W. X. (2002). Interactions of trace metals and different marine food chains. *Marine Ecology Progress Series* 243: 295-309.

Wang, W. X., R. C. H. Dei and Y. Xu (2001). Cadmium uptake and trophic transfer in coastal plankton under contrasting nitrogen regimes. *Marine Ecology Progress Series* 211: 293-298.

Verriopoulos, G. and S. Dimas (1988). Combined toxicity of copper, cadmium, zinc, lead, nickel, and chrome to the copepod *Tisbe holothuriae*. *Bulletin of Environmental Contamination and Toxicology* 41(3): 378-384.

Vondráĉek, J., L. Svihalkova-Sindlerova, K. Pencikova, S. Marvanova, P. Krcmar, M. Ciganek, J. Neca, J. E. Trosko, B. Upham, A. Kozubik and M. Machala (2007). Concentrations of methylated naphthalenes, anthracenes, and phenanthrenes occurring in Czech river sediments and their effects on toxic events associated with carcinogenesis in rat liver cell lines. *Environmental Toxicology and Chemistry* 26(11): 2308-2316.

Xu, Y. and W. X. Wang (2001). Individual responses of trace-element assimilation and physiological turnover by the marine copepod Calanus sinicus to changes in food quantity. *Marine Ecology Progress Series* 218: 227-238.

Ytreberg, E., I.-M. Hassellöv, A. T. Nylund, M. Hedblom, A. Y. Al-Handal and A. Wulff (2019). Effects of scrubber washwater discharge on microplankton in the Baltic Sea. *Marine Pollution Bulletin* 145: 316-324.

Ytreberg, E., A. Lunde Hermansson and I.-M. Hassellöv (2020). Deliverable 2.1 - Database and analysis on waste stream pollutant concentrations, and emission factors. EMERGE: Evaluation, control and Mitigation of the EnviRonmental impacts of shippinG Emissions, funded by European Union’s Horizon 2020 research and innovation programme under grant agreement No 874990. pp: 1-37.

Zhou, C., V. Vitiello, E. Casals, V. F. Puntes, F. Iamunno, D. Pellegrini, W. Changwen, G. Benvenuto and I. Buttino (2016). Toxicity of nickel in the marine calanoid copepod Acartia tonsa: Nickel chloride versus nanoparticles. *Aquatic Toxicology* 170: 1-12.

Annex 1: Technical minutes from the Scrubbers Review Group

- RGSCRUB
- By correspondence August 2020
- Participants: Sonja Endres (Chair) and Johannes Teuchies
- Working Group: WGSHIP

1. ICES viewpoint on scrubbers – Review by Sonja Endres

This ICES viewpoint represents a major effort to summarize and evaluate the potential impacts of seawater scrubbing in shipping on the marine environment. In addition, the authors compare alternative mitigation measures including existing and future technological and operational alternatives.

This report is an excellent summary of the current state of research and comprehensively classifies the potential risks of scrubber use. It has been carefully researched and written in an easily understandable manner. The objectives of this viewpoint are of high relevance and interest for both, policy-makers and industry.

Therefore, I have only some minor comments and I would like to add some further ideas and remarks from to be considered shortly in the text:

Chapter 1 Global use of scrubbers

- Installation of scrubbers on ships: There were also concerns by shipping companies on the availability of low sulphur fuel on the market. Limited amounts would come at increasing costs for them. The installation of EGCS is also costly but assuming declining costs for HFO made it attractive as an intermediate solution while looking for long-term alternatives such as LNG or even methanol → see outlook zero emission shipping

Chapter 2 Consequences

- An ongoing transdisciplinary project called ShipTRASE financed by the Belmont forum will analyse the environmental, economic and legal aspects of ship emission reduction mechanisms such as introducing ECAs. It will assess the efficiency of current emission control regulation (or lack thereof) on different levels and evaluate governance instruments. Regarding biochemical cycling, it will study the impact of scrubber discharge on organic compounds in the seawater and the concentration of climate-actives gases in the surface ocean. For further information, please contact Christa Marandino, GEOMAR or Anna Rutgersson from Uppsala University.

Chapter 3.3 Regulations, monitoring and enforcement

- Legal regulation on different levels, i.e. IMO, EU, individual states, can address vessel-based air pollution by different means of standard-setting. Revised or new legal standards can concern types of fuels, use of mitigation technology during vessel operation, port-reception facilities for waste disposal, and ship-building. A close collaboration between science, industry and decision-makers is necessary to achieve acceptable and sustainable solutions that will be implemented and can be monitored. Since technology develops fast, legal regulation need to be flexible to be adapted to upcoming technical options.

Chapter 4 Conclusion

- The MEPC agreed to define a Baltic Sea Special Area prohibiting liquid discharge from new (since June 2019) and existing (from June 2021) passenger ships to the sea. So far, scrubber discharge is not included (correct me if I am wrong) but should be considered as "sewage" and consequently prohibited in the Baltic Sea in the future.
- Outlook future zero emission shipping (IMO GHG Strategy 2050): In April 2018, MEPC adopted a strategy on the reduction of greenhouse gas emissions from ships. The aim is to reduce the total annual GHG emissions in global shipping by at least 50% by 2050 compared to 2008. Therefore, in the next 25 years, the majority of shipping companies are expecting to replace present fuel oils by cleaner alternatives, such as liquified natural gas (LNG) or methanol and in certain cases also electric propulsion. The consequences of alternative fuel use need to be investigated in advance to avoid side effects, such as those potentially introduced by the use of scrubbers.

Minor comments:

Use µg*L-1 instead of µg/L, same for mg/L

2. ICES viewpoint on scrubbers – Review by Johannes Teuchies

I believe this review is very valuable and gives a good overview of the existing knowledge on scrubbers and their potential impact on aquatic ecosystems. The number of vessels installing a scrubber is increasing rapidly, but legislation on washwater discharge is limited and not consistent between countries or areas. To my knowledge, the information provided by industry or shipping companies (no or very little effects from scrubber washwater) is not entirely in line with recent scientific results (effects of scrubber washwater are expected, mainly in certain areas).

Some thoughts, suggestions:

- For me, an important issue concerning the impact of scrubbers is the difference in total contaminant fluxes resulting from HFO+scrubber and MGO. When reading this opinion paper and not familiar with the topic, the main impact of scrubbers might be interpreted as 'trading reductions in air pollution for increased water pollution'. However, of large importance to me is that the total contaminant flux to the environment (water + air) is much higher for HFO+scrubber compared to MGO. Moreover, it has been reported that, in addition to the generated washwater fluxes also the emissions to the air are higher for HFO+ scrubbers then for MGO for several contaminants (e.g. [1]). I believe that this broader view on impact is important to include in the risk assessment and maybe can have somewhat more attention in the paper (in general introduction, chapter on load or mitigation measures, ….)?
- As mentioned in the text (chapter 1.2.4) the impact of scrubbers on N flux and related eutrophication is very low. Trapping N in the washwater is limited and I would assume the overall outflux of NOx (air + water) is similar between HFO+scrubber and MGO (not for most other contaminants). I believe the impact of scrubbers on eutrophication is very limited and I would suggest to nuance.

- In chapter 1.2. 'Chemical composition' the differences between OL and CL are discussed. Differences between OL and CL also exist in load (chapter 1.3) and are clear from Figure 5. I suggest to shortly discuss this also in the text.
- Chapter 2 'Consequences' gives interesting information on possible effects of metals, PAH, acidification, synergistic effects, … However, it is rather general. Information to answer the question whether scrubber discharge will have a negative impact on aquatic ecosystems and under which circumstances is limited (in real environments, not lab based). I agree that this is a difficult question and information is limited. However, I suggest to try to improve the link between the given information on effects (toxicity, bioaccumulation, pH) and expected changes caused by scrubbers. Information on expected dilution factors and concentration changes might be included (e.g. calculated for a harbor in Teuchies et al., 2020).
- The effect of washwater on the redox conditions (chapter 2.2.2) is not very clear to me. I agree that acidification can have an impact the mobility of metals. But it is not clear to me how this relates to redox reactions (see comments in the text).
- I agree that the current limits set for contaminants in washwater are insufficient to protect most receiving aquatic ecosystems. However, I believe that defining new limits, organising sampling and having a control system will be extremely difficult for several reasons. (1) during sampling we found out that contaminant concentrations fluctuated already on one vessel using one type of fuel. A lot of factors will have an effect on the washwater concentrations, and it will be very difficult for ships to know under which conditions they will meet the criteria and be able to use the scrubber. (2) The impact of discharge will largely depend on conditions of the receiving water body. (3) In order to protect receiving aquatic ecosystems, I believe that the discharge limits (metals, PAHs, …) will end up to be lower than concentrations measured in most scrubber washwater. As long as vessels with scrubbers will use HFO, contaminant concentrations in washwater will be elevated.

  Hence, I'm not sure a revision of the discharge limits will be possible, a solution. The ban of scrubbers in certain areas (e.g. rivers, coasts, estuaries, harbours) might be a first step. I believe it should be made clear that the use of open loop scrubbers as an abatement technology will not contribute to mitigation of the impact of high sulfur emissions (which is the objective of the IMO sulphur guidelines).

Lehtoranta, K.; Aakko-Saksa, P.; Murtonen, T.; Vesala, H.; Ntziachristos, L.; Rönkkö, T.; Karjalainen, P.; Kuittinen, N.; Timonen, H., Particulate Mass and Nonvolatile Particle Number Emissions from Marine Engines Using Low-Sulfur Fuels, Natural Gas, or Scrubbers. *Environ. Sci. Technol.* **2019,** *53*, (6), 3315-3322.

3. Reviewers' comments and authors' responses

Reviewer # 1	**Authors' responses:**
ICES viewpoint on scrubbers – Review	
I believe this review is very valuable and gives a good overview of the existing knowledge on scrubbers and their potential impact on aquatic ecosystems. The number of vessels installing a scrubber is increasing rapidly, but legislation on washwater discharge is limited and not consistent between countries or areas. To my knowledge, the information provided by industry or shipping companies (no or very little effects from scrubber washwater) is not entirely in line with recent scientific results (effects of scrubber washwater are expected, mainly in certain areas).	We thank Reviewer #1 for the thorough and constructive review of the Background document to the Viewpoint on scrubbers. Responses to Reviewer# 1's questions, comments and suggestions, will be made below and in the document "ICES Viewpoint on scrubbers background document – revised Sept 12 2020.docx".
Some thoughts, suggestions	
- For me, an important issue concerning the impact of scrubbers is the difference in total contaminant fluxes resulting from HFO+scrubber and MGO. When reading this opinion paper and not familiar with the topic, the main impact of scrubbers might be interpreted as 'trading reductions in air pollution for increased water pollution'. However, of large importance to me is that the total contaminant flux to the environment (water + air) is much higher for HFO+scrubber compared to MGO. Moreover, it has been reported that, in addition to the generated washwater fluxes also the emissions to the air are higher for HFO+ scrubbers then for MGO for several contaminants (e.g. 1). I believe that this broader view on impact is important to include in the risk assessment and maybe can have somewhat more attention in the paper (in general introduction, chapter on load or mitigation measures, ….)?	We acknowledge the reasoning regarding HFO+scrubber vs MGO and will add the reference suggested. However, many ships will not use MGO but may use other compliant fuels such as the new generation of fuel blends, often referred to as hybrid fuels. We believe that this issue is already addressed in the background document under 3.1. "Distilled fuels, like Marine Gas Oil (MGO), Liquefied Natural Gas (LNG) or biofuels, have been reported not contain the same toxic combinations as residual fuel oils (Sippula et al. 2014, Corbin et al. 2020, Su et al. 2019) and are compliant regarding emissions to air, without increasing the impact on the marine environment."
- As mentioned in the text (chapter 1.2.4) the impact of scrubbers on N flux and related eutrophication is very low. Trapping N in the washwater is limited and I would assume the overall outflux of NOx (air + water) is similar between HFO+scrubber and MGO (not for most other contaminants). I believe the impact of scrubbers on eutrophication is very limited and I would suggest to nuance.	We agree that the impact of scrubbers on the N-flux is probably the least explored effect, yet e.g. Ytreberg et al 2018 found that scrubber discharge water stimulated growth of natural microbial communities. At Chalmers University of Technology we are also currently looking into the possibility that spectrophotometric analysis of NO2+NO3 in scrubber discharge water may be biased due to interference of something in the scrubber water, since we have observed stimulation of growth that cannot be explained by the nutrient nor trace element content of the scrubber water compared to a control. This is however unpublished results that cannot be included as a basis for the Viewpoint. At the same time, scrubber discharge water is not a

	standard water to analyse in analytical lab, and as long as the NO2+NO3 concentrations is not extreme there is no reason to question them; you will only note it if you run experiments with primary producers. Therefore, we chose to still mention the Eutrophication as a case, without picturing it as the biggest threat from scrubbers. In e.g. the Baltic Sea however, the potential contribution to eutrophication from scrubbers could be of more concern than in non-eutrophied environments.
- In chapter 1.2. 'Chemical composition' the differences between OL and CL are discussed. Differences between OL and CL also exist in load (chapter 1.3) and are clear from Figure 5. I suggest to shortly discuss this also in the text.	Added text in *italics*: Despite almost all 2000 ships discharging bilge, black and grey water, the load of metals and PAHs from the 99 ships equipped with scrubbers was higher by 10-100-fold, *completely dominated by open loop*.
- Chapter 2 'Consequences' gives interesting information on possible effects of metals, PAH, acidification, synergistic effects, … However, it is rather general. Information to answer the question whether scrubber discharge will have a negative impact on aquatic ecosystems and under which circumstances is limited (in real environments, not lab based). I agree that this is a difficult question and information is limited. However, I suggest to try to improve the link between the given information on effects (toxicity, bioaccumulation, pH) and expected changes caused by scrubbers. Information on expected dilution factors and concentration changes might be included (e.g. calculated for a harbor in Teuchies et al., 2020).	*Added the following paragraph in italics:* *Simulations in port environments estimate high increases in contaminant levels as a result of scrubber discharge. Simulations of the Port of Antwerp showed pronounced increases in the surface water for naphthalene, with an increased concentration of 39% under "scenario LOW" and 189% under "scenario HIGH" and vanadium, which increased 9% under "scenario LOW" and 46% under "scenario HIGH" (Teuchies et al. 2020). The modeling results from the Scheldt estuary for naphthalene showed increased concentration of 5.0% with "scenario LOW" and 25% with "scenario HIGH". In both the Port of Antwerp and the Scheldt estuary, the EQS for surface water according to EU WFD are already exceeded with respect to fluoranthene, further surpassed by scrubber discharge. Nickel, zinc and vanadium are all close to the EQS in the Port of Antwerp, and for nickel and zinc the scrubber discharge contribution is expected to cause exceedance. In the Scheldt estuary, the modeled concentration of pyrene in surface water also exceeds the EQS according to EU WFD and vanadium is close to the EQS.*
- The effect of washwater on the redox conditions (chapter 2.2.2) is not very clear to me. I agree that acidification can have an impact the mobility of metals. But it is not clear to me how this relates to redox reactions (see comments in the text).	Redox conditions are in general defined at standard conditions and will change with altered physico-(geo)chemical conditions e.g. with altered pH and temperature (e.g. Grundl et al 2011). To fully explain the redox chemistry is beyond the scope of this

	document, but we have added new references to facilitate further reading for an interested reader. Re: mobility of metals. Added text in italics: as a result of *increased mobility following* increasing acidity.
- I agree that the current limits set for contaminants in washwater are insufficient to protect most receiving aquatic ecosystems. However, I believe that defining new limits, organising sampling and having a control system will be extremely difficult for several reasons. (1) during sampling we found out that contaminant concentrations fluctuated already on one vessel using one type of fuel. A lot of factors will have an effect on the washwater concentrations, and it will be very difficult for ships to know under which conditions they will meet the criteria and be able to use the scrubber. (2) The impact of discharge will largely depend on conditions of the receiving water body. (3) In order to protect receiving aquatic ecosystems, I believe that the discharge limits (metals, PAHs, …) will end up to be lower than concentrations measured in most scrubber washwater. As long as vessels with scrubbers will use HFO, contaminant concentrations in washwater will be elevated. Hence, I'm not sure a revision of the discharge limits will be possible, a solution. The ban of scrubbers in certain areas (e.g. rivers, coasts, estuaries, harbours) might be a first step. I believe it should be made clear that the use of open loop scrubbers as an abatement technology will not contribute to mitigation of the impact of high sulfur emissions (which is the objective of the IMO sulphur guidelines).	Agree regarding increased costs for increased control and monitoring, added text in italics: Effective mitigation of scrubber impacts needs stringent requirements and standards, monitoring protocols and widespread, effective enforcement, *which will also imply increased costs* No, and it is not proposed that the revision of discharge limits will solve the issue, but if scrubbers are to be continued to be used, then there is an urgent need for alternative measures as summarized in the Conclusions section.
Lehtoranta, K.; Aakko-Saksa, P.; Murtonen, T.; Vesala, H.; Ntziachristos, L.; Rönkkö, T.; Karjalainen, P.; Kuittinen, N.; Timonen, H., Particulate Mass and Nonvolatile Particle Number Emissions from Marine Engines Using Low-Sulfur Fuels, Natural Gas, or Scrubbers. Environ. Sci. Technol. 2019, 53, (6), 3315-3322.	Added

Reviewer # 2	Authors' responses:
ICES viewpoint on scrubbers – Review	
This ICES viewpoint represents a major effort to summarize and evaluate the potential impacts of seawater scrubbing in shipping on the marine environment. In addition, the authors compare alternative mitigation measures including existing and future technological and operational alternatives. This report is an excellent summary of the current state of research and comprehensively classifies the potential risks of scrubber use. It has been carefully researched and written in an easily understandable manner. The objectives of this viewpoint are of high relevance and interest for both, policy-makers and industry. Therefore, I have only some minor comments and I would like to add some further ideas and remarks from to be considered shortly in the text:	Thank you!
Chapter 1 local use of scrubbers	
- Installation of scrubbers on ships: There were also concerns by shipping companies on the availability of low sulphur fuel on the market. Limited amounts would come at increasing costs for them. The installation of EGCS is also costly but assuming declining costs for HFO made it attractive as an intermediate solution while looking for long-term alternatives such as LNG or even methanol ◎ see outlook zero emission shipping	Agree, but there was an Imo investigation prior to 2018 to analyse the availability of fuels prior to the stricter regulations that were proposed to enter into force 2020. If the investigation had shown that there wasn't enough fuel availability, IMO had a possibility to postpone the date of entry into force until 2025.
Chapter 2 Consequences	
- An ongoing transdisciplinary project called ShipTRASE financed by the Belmont forum will analyse the environmental, economic and legal aspects of ship emission reduction mechanisms such as introducing ECAs. It will assess the efficiency of current emission control regulation (or lack thereof) on different levels and evaluate governance instruments. Regarding biochemical cycling, it will study the impact of scrubber discharge on organic compounds in the seawater and the concentration of climate-actives gases in the surface ocean. For further information, please contact Christa Marandino, GEOMAR or Anna Rutgersson from Uppsala University.	Thank you for the information. As discussed at the ADG it is however impossible to include this information as it is not yet published. But we look forward to follow the development of this exciting research!
Chapter 3.3 Regulations, monitoring and enforcement	
- Legal regulation on different levels, i.e. IMO, EU, individual states, can address vessel-based air pollution by different means of standard-setting. Revised or new legal standards can concern types of fuels, use of mitigation technology during vessel operation, port-reception facilities for waste disposal, and ship-building. A close collaboration between science, industry and deci-	Agree.

sion-makers is necessary to achieve acceptable and sustainable solutions that will be implemented and can be monitored. Since technology develops fast, legal regulation need to be flexible to be adapted to upcoming technical options.	
Chapter 4 Conclusion	
- The MEPC agreed to define a Baltic Sea Special Area prohibiting liquid discharge from new (since June 2019) and existing (from June 2021) passenger ships to the sea. So far, scrubber discharge is not included (correct me if I am wrong) but should be considered as "sewage" and consequently prohibited in the Baltic Sea in the future.	This was a constructive thought, but maybe difficult to propose as sewage is already defined rather well and the scrubber issue is already handled under Annex VI.
- Outlook future zero emission shipping (IMO GHG Strategy 2050): In April 2018, MEPC adopted a strategy on the reduction of greenhouse gas emissions from ships. The aim is to reduce the total annual GHG emissions in global shipping by at least 50% by 2050 compared to 2008. Therefore, in the next 25 years, the majority of shipping companies are expecting to replace present fuel oils by cleaner alternatives, such as liquified natural gas (LNG) or methanol and in certain cases also electric propulsion. The consequences of alternative fuel use need to be investigated in advance to avoid side effects, such as those potentially introduced by the use of scrubbers.	Agree.
Minor comments:	
Use µg*L-1 instead of µg/L, same for mg/L	Changed.