Basic principles for development and implementation of plastic clean-up technologies

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Basic principles for development and implementation of plastic clean-up technologies: What can we learn from fisheries management?

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HIGHLIGHTS

- Plastic clean-up technologies generally lack environmental cost-benefit analyses.
- We suggest key principles from fisheries management for such evaluations.
- Principles of catch efficiency and bycatch reduction are relevant for plastic clean-ups.
- Scarce data on plastic distribution and overlap with ecosystem components limits evaluation.
- A lack of cost-benefits analyses risks damage to ecosystems and inefficient mitigation actions.

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ABSTRACT

Plastic pollution compromises ocean health, with large amounts of plastics continuing to enter marine and coastal environments. Various mitigative engineering solutions are being developed and implemented in response to this threat. While recognising the positive impacts of clean-ups, we highlight two perspectives given little attention to date, which are vital to evaluating the cost-benefit ratio of clean-ups: firstly, clean-up efficiency where density and accessibility of litter are key, and secondly, potential negative externalities that implementation of clean-up technologies may have. These principles, catch per unit effort and the impact on non-target species, are well known from fisheries management. We argue they should also be applied in evaluating marine litter removal schemes.

1. Introduction

Plastic pollution is one of the present day's great challenges compromising the health of the world's oceans. An estimated 5–13 million tons of plastic entered our oceans in 2010 from land-based sources. This amount is predicted to fourfold by 2050 (Jambeck et al., 2015). In addition, there are considerable marine litter inputs from sea-based sources (Deshpande et al., 2020; Macfayden et al., 2009; Ryan et al., 2019). An estimated five trillion pieces of plastic weighing over 250,000 tons afloat in the world's oceans (Eriksen et al., 2014); an estimate which does not consider litter that has sunk to the seafloor or been beached. A decrease in the influx of litter is critical to combating the problem, yet efforts to reduce litter already present in the marine environment are also desirable (Rochman, 2016). Beach clean-ups are a common way to achieve the latter, but there is also considerable ongoing global technology development to remove ocean plastics.

The Ocean Clean-up is the best-known such technology (TheOceanClean-up, 2020). Its goal is to corral and capture floating plastics in the North Pacific Gyre. Numerous other schemes to remove floating plastics are also under development or already in use (Table 1). For example, the Seabin collects floating debris from ports and marinas (SeabinProject, 2020), Petroleum Geo-Services ASA (PGS) is combining an air bubble curtain to concentrate plastics at the surface with a towed collection boom (Falk-Andersson et al., 2018), and Mr. Trash Wheel funnels trash from harbours and rivers, preventing loss to the sea (MrTrashWeel, 2020). The research field of marine litter is relatively new, with scientific publications rising exponentially the past decade (Ryan, 2015a). Consequently, interest

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Table 1
Brief overview of surface (sea and rivers) clean-up technologies already available and under development (the latter are indicated by launch year N/A).

Technology	Type	Size*	Target area	Launch year	Units in use	Tonnage cleaned	CPUE** estimate	EIA***	Source
The Ocean Cleanup System 001	Passive	Large	Open sea and/or coast	2018§	1	No data	2014	2018	(TheOceanClean-up, 2019)
PGS bubble curtain tow	Active	Medium/large	Open sea and/or coast	–	–	–	2018	2018	(Falk-Andersson et al., 201)
Sea Cleaners "the Manta"	Active	Medium/large	Open sea and/or coast	–	–	–	–	–	(TheSeaCleaners, 2020)
Cleaner Ocean Foundation "SeaVax"	Active	Medium	Coastal waters	–	–	–	–	–	(BlueGrowth, 2019)
Ranmarine Technology WasteShark	Active	Small	Marinas/estuaries	2018	No data	No data	–	–	(RanMarine, 2020)
Seabin	Passive	Small	Marinas/estuaries	2016	860	869	–	–	(SeabinProject, 2020)
Mr. Trash Wheel	Passive	Medium	Rivers/estuaries	2014	No data	1396	–	–	(MrTrashWeel, 2020)
Storm Water Systems: Bandalong Litter Trap	Passive	Medium	Rivers	2008	No data	No data	–	–	(StormWaterSystems, 2020)
The Ocean Cleanup Interceptor	Passive	Medium	Rivers	2019	2	No data§§	–	2019	(TheOceanClean-up, 2020)
River Cleaning	Passive	Small/medium	Rivers	–	–	–	–	–	(RiverCleaning, 2019)
Urban Rivers Trash Robot	Active	Small	Rivers	2018	No data	No data	–	–	(UrbanRivers, 2019)
Storm Water Systems: StormX Trash Trap	Passive	Small	Storm drains	1995	No data	No data	–	–	(StormWaterSystems, 2020)

* Relative size of system/infrastructure, compared to other systems/infrastructures.
** CPUE or efficiency evaluation of technology during technology development (excluding reports of litter caught using the technology).
*** Environmental impact assessment or other type of evaluation of environmental impact.
§ Feasibility study in 2014.
§§ Not publicly available and therefore cannot be externally viewed and assessed.

in remediation technology development is also quite recent. Increased media attention the past few years has undoubtedly acted as a trigger, and the flux of technology development projects likely reflects a phase of inflated expectations in the innovation hype cycle (van Lente et al., 2013). Removal of plastic pollution is obviously desirable, but in many cases the discussion focuses exclusively on remediation without a detailed consideration of the effectiveness, costs and benefits of such solutions, and without acknowledging a basic lack of data.

To ensure the sustainability and net positive environmental benefit of marine plastic pollution remediation schemes, we propose that innovation within the field should look to key principles from fisheries management. We illustrate the analogy between catch-per-unit-effort (CPUE) from fisheries science and the efficiency of removing marine litter, and point to the large knowledge gaps regarding the density of marine plastics in time and space. Finally, while there are benefits of clean-ups through reducing the stock of litter, we demonstrate that there may also be negative ecological impacts of litter removal through by-catch and habitat destruction. Accounting for these factors will limit the negative environmental externalities of clean-up technologies and improve the socio-economic benefit of downstream measures to combat marine plastics. We limit our discussion to floating macro-litter as an example as this has been the primary focus of technology development, but the discussion applies to the clean-up of all marine litter.

2. Framework for discussing the cost-benefit of marine litter clean-up

In developing clean-up technologies of marine litter, the focus has been on removing litter from the environment (point 1 and 3, Fig. 1). From fisheries management we know that catch is a function of stock size and effort, combined with the accessibility of the fished resource. The most important factors determining the economics of a fishery are stock size, growth and catch-efficiency, with the cost of fishing being lower for dense, rapidly growing populations that are easily accessible (Flaaten, 2011). Similarly, the cost-effectiveness of marine litter clean-ups is determined by the density and accessibility of litter, which again could be affected by the clean-up actions themselves through reduction in the stock of plastics (point 4, Fig. 1). In addition, unintentional interactions between harvest technologies and non-targeted species and habitats may result in significant negative environmental impacts of harvesting (point 2, Fig. 1) (Armstrong and Falk-Petersen, 2008; Pikitch et al., 2004; Wells and Rooker, 2004).

Naturally there are some key differences between fisheries management and plastics removal, primarily with respects to mechanisms of population growth and the management objectives (sustainable yield *vs.* extinction); furthermore, the market, including the price, of marine plastics is not well established. Nevertheless, the concepts of stock size, effort and yield remain highly transferable and applicable to marine plastic pollution mitigation; particularly within an ecosystem-based management approach to consider direct and indirect ecological impacts of harvesting plastics. In the next sections we illustrate how the framework adapted from key fisheries management principles in Fig. 1 can be applied in the context of plastic clean-up strategies.

3. Catch-per-unit-effort

In any fishery, the resource is preferably pursued where its density, and therefore the CPUE, is the greatest. The concentrations of macroplastics floating at the sea surface are highly variable, both in space and time (Table 2). Any clean-up technology should therefore target specific areas and times where and when densities are high. Factors driving this variability include entry points, settlement rates, litter characteristics (*e.g.*, buoyancy, degradation rate), oceanographic factors (*e.g.*, currents, wind), and seasonal changes in weather patterns (Critchell et al., 2015; Galgani et al., 2015; Jambeck et al., 2015; Pedrotti et al., 2016; Van Sebille et al., 2012). Processes influencing spatial variation in the concentration of surface plastics have been described and modelled, yet there are relatively few empirical recordings of floating plastics in general, and macroplastics in particular (Galgani et al., 2015). As pointed out by other authors, the microplastic fraction dominates plastic pollution studies in marine and freshwater environments (Blettler et al., 2018; Eriksen et al., 2014; Lebreton et al., 2017; Schmidt et al., 2017). The majority of studies on the distribution of floating plastics have also been done on microplastics. Lusher (2015) found 24 studies on the abundance floating microplastic in the Pacific Ocean and 21 from the Atlantic Ocean, while this review only identified 11 studies on macroplastics in the Pacific and 5 in the Atlantic (Table 2). A review of research articles published on floating plastic pollution in the past two years 2018–2019 (search terms on Science Direct: "marine

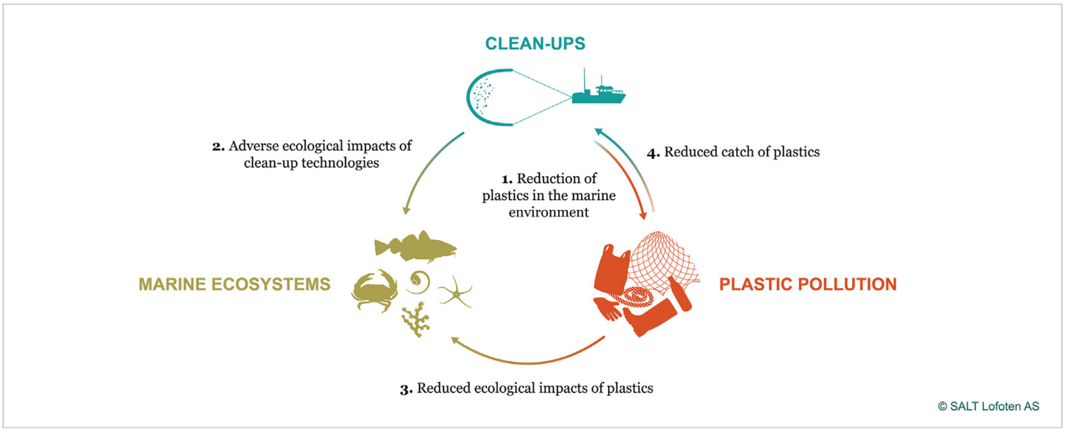

Fig. 1. A simple visual representation of a fisheries management-based framework to identify factors to consider when evaluating marine plastic removal schemes; modified from (Armstrong and Falk-Petersen, 2008).

litter" OR "marine debris" OR plastic) AND (surface OR ocean OR pelagic OR floating OR gyre) yielded 26 relevant articles, 65% of which concerned microplastics only, 15% concerned macroplastics, and 19% concerned both. Clean-up technologies, on the other hand, typically focus on larger fractions. Thus, there are relatively few data to determine the density of plastic available for surface clean-up even in the areas where densities are believed to be the highest.

Given the highly variable densities of floating plastics reported in the literature, including over small spatial scales and in time, CPUE is challenging to estimate. Nevertheless, we attempt to do so based on available data (Table 2). Each estimate is made assuming a 500 m aperture of the collection array (*e.g.*, a trawl or boom), a 3 m vertical profile, and a tow speed of 1.5 knots, as per the suggested design of the PGS collection array (Falk-Andersson et al., 2018), and assuming 100% capture efficiency of encountered plastics. Note that most data on floating plastics are surface observations only and allow no real consideration of the vertical dimensions of the clean-up technology. However, this may have little bearing on CPUE as he majority of plastics are believed to be within the top half meter of the water column (Reisser et al., 2015). The estimates show large variations in the expected mean CPUE, both in terms of the number of items (<1–5400 items per hour towed) and weight (<1–350 kg per hour towed), among studies (Table 2). Note also that all density estimates are scaled up based on sampling of smaller areas, often several within a km^2, further highlighting the patchiness of floating plastics. Consequently, density estimates are subject to considerable error. This is further complicated by great variations in sampling methodology, making comparisons among studies challenging.

Globally, the highest accumulations of floating plastic appears to be in the subtropical gyres, particularly in the northern hemisphere, and in the waters off south-east Asia (Eriksen et al., 2014; Law et al., 2010; Lebreton et al., 2012; Van Sebille et al., 2012). The most comprehensive empirical study on floating plastics to date was conducted in the North Pacific Subtropical Gyre (Lebreton et al., 2018). The mean concentration of meso- and macroplastic pieces was 67 kg km^{-2}, which equates to an estimated CPUE of 120 kg hr^{-1} (Table 2). However, the minimum and maximum values recorded varied considerably, ranging from 800 g to 500 kg km^{-2} (Lebreton et al., 2018) and suggesting variable CPUE from 1 kg to 1 t hr^{-1} (Table 2). Consequently, litter densities are highly patchy even within key accumulation zones. Thus, even here CPUE could be very low and we have a limited understanding of where and when there may be high-density zones within these areas where clean-up technologies can be more efficient.

While litter density may be high at any given point in time in accumulation zones, not all litter will reach them and interception closer to sources will have a greater cumulative impact. A modelling study of floating microplastics found that the cumulative removal is expected to be higher in nearshore areas where litter input to the ocean is high (*e.g.*, south-east Asia) (Sherman and van Sebille, 2016). The residence time of plastics at the sea surface, and thus their potential for transport to accumulation zones, is affected by buoyancy and susceptibility to fragmentation and biofouling (Ryan, 2015b). Less buoyant items are more common in rivers and nearshore than in the high seas (Crosti et al., 2018; Ryan, 2015b). Soft plastics are rarely observed >800 km offshore and likely sink or fragment shortly after leaving the coast (Marcus Eriksen, 5 gyres Institute, pers. com. 2018) and their presence indicates relatively recent discards (Arcangeli et al., 2018). There is also ample evidence to suggest sustained high litter densities close to source points. Several empirical studies have reported negative correlations between litter density and distance from shore (Díaz-Torres et al., 2017; Pedrotti et al., 2016; Rudduck et al., 2017; Ryan, 2013, 2014; Thiel et al., 2013). Both model simulations and empirical data suggest plastic from river outflows accumulates on nearby beaches (Critchell et al., 2015; Rech et al., 2014), implying relatively short transport pathways. Surface plastic concentrations are positively correlated with coastal population density (Pedrotti et al., 2016), as well as the density of fisheries- and shipping traffic (Grøsvik et al., 2018).

High densities of litter are also found in many rivers as these are transport routes for litter from populations living in their catchment areas, particularly in regions with poor waste management systems (Lebreton et al., 2017). Litter densities vary widely both among and within rivers, just as there is considerable spatial variation in litter densities at the sea surface (*e.g.*, Gasperi et al., 2014; Lebreton et al., 2017; van Emmerik et al., 2018). However, the mean concentration of plastic debris in rivers has been estimated to be 40–50 times higher than the maximum concentration observed floating in the open ocean (Schmidt et al., 2017). Modelling studies have identified the rivers estimated to carry the most litter globally, although these estimates are backed by limited empirical data (Lebreton et al., 2017; Schmidt et al., 2017). Empirical studies in rivers are on the rise, but many focus exclusively on microplastics and sampling design is often biased towards them (Blettler et al., 2018; Schmidt et al., 2017). Nevertheless, recent observations from certain Asian rivers do confirm that macroplastic densities can be high with daily discharge rates of 200 kg to 1.5 t (van Emmerik et al., 2018, 2019) (Table 3). These estimates suggest sustained CPUE of <1 to 200 kg per hour for stationary clean-up technology cross-sectioning the entire river with 100% capture efficiency of encountered plastic (Table 3).

In addition to varying spatially, litter density also varies over time, both at sea and in rivers (Díaz-Torres et al., 2017). Acute pollution events may arise, such as through ship wrecking, container losses and storm events. The 2011 tsunami in Japan, for example, swept an estimated 5 million tons of litter into the ocean as it retreated; an estimated 70% of which sank close to the shore (Bagulayan et al., 2012; Murray

Table 2
A summary of available data on floating anthropogenic litter, excluding microplastics (*i.e.*, mesoplastics and larger). Note that all density estimates (# or mass per km^2) are scaled up based on sampling of smaller areas, often several within a km^2. Range and mean values are given when available; if a range is given in mean values, this represents spatiotemporal variation. CPUE (catch-per-unit-effort) estimates are based on a 500 m aperture of the collection array, a tow speed of 1.5 knots, and the assumption of 100% capture efficiency of encountered litter.

Ocean basin	Litter density ($kg\ km^{-2}$)	Litter density ($\#\ km^{-2}$)	Source(s)	CPUE ($kg\ h^{-1}$)	CPUE ($\#\ h^{-1}$)
North Atlantic		0–112; $\bar{x} = 11–37$	(Dufault and Whitehead, 1994)		0–207 $\bar{x} = 20–68$
		1–12	(Sá et al., 2016)		1.8–22
		0.2–123; $\bar{x} = 0.8$	(Chambault et al., 2018)		0.4–228; $\bar{x} = 1.5$
North Sea		0–3	(Dixon and Dixon, 1983)		0–6
		$\bar{x} = 32$	(Thiel et al., 2011)		$\bar{x} = 60$
		0–272; $\bar{x} = 31$	(Gutow et al., 2018)		0–503; $\bar{x} = 57$
Mediterranean		$\bar{x} = 2000$	(Morris, 1980)		$\bar{x} = 3700$
		$\bar{x} = 0,1$	(McCoy, 1988)		$\bar{x} = 0.2$
		$\bar{x} = 1.2–5$	(Aliani et al., 2003)		$\bar{x} = 2–9$
	$\bar{x} = 0.17$	$\bar{x} = 5$	(Eriksen et al., 2014)[a]	$\bar{x} = 0.24$	$\bar{x} = 7$
		0–162; $\bar{x} = 25$	(Suaria and Aliani, 2014)		0–300; $\bar{x} = 46$
		$\bar{x} = 4–38$	(Di-Méglio and Campana, 2017)		$\bar{x} = 7–70$
		$\bar{x} = 175$	(Fossi et al., 2017)		$\bar{x} = 324$
		$\bar{x} = 32–115$	(Carlson et al., 2017)		$\bar{x} = 59–213$
		$\bar{x} = 2–5$	(Arcangeli et al., 2018)		$\bar{x} = 4–9$
		$\bar{x} = 0.5–4$	(Campana et al., 2018)		$\bar{x} = 0.7–5.6$
		0–4500	(Zeri et al., 2018)		0–8300
	$\bar{x} = 0.7$	$\bar{x} = 2897$	(Ruiz-Orejón et al., 2018)	$\bar{x} = 1.3$	$\bar{x} = 5400$
	$\bar{x} = 80–190$		(Compa et al., 2019)	$\bar{x} = 148–351$	
		18–1600; $\bar{x} = 232$	(Constantino et al., 2019)		18–1600 $\bar{x} = 430$
		$\bar{x} = 0–3.2$	(Garcia-Garin et al., 2019)		$\bar{x} = 0–4.6$
		0–690; $\bar{x} = 175$	(Palatinus et al., 2019)		0–1300 $\bar{x} = 324$
Black Sea		max = 136	(Suaria et al., 2015)		max = 252
Gulf of Mexico		0.6–2.4	(Lecke-Mitchell and Mullin, 1997)		1–4.5
South Atlantic	$\bar{x} = 0.2$	$\bar{x} = 1.8$	(Eriksen et al., 2014)[a]	$\bar{x} = 0.3$	$\bar{x} = 2.5$
		$\bar{x} = 0.6–67$	(Ryan et al., 2014)		$\bar{x} = 1.1–124$
North Pacific		$\bar{x} = 4.2$	(Venrick et al., 1973)		$\bar{x} = 8$
		0.2–1.8	(Day and Shaw, 1987)		0.4–3
		$\bar{x} = 0.4$; max = 3.3	(Shiomoto and Kameda, 2005)		$\bar{x} = 0.6$; max = 6
		0–2.5	(Pichel et al., 2007)		0–5
		0–6300	(Titmus and David Hyrenbach, 2011)		0–11,700
	$\bar{x} = 17$	$\bar{x} = 53$	(Eriksen et al., 2014)[a]	$\bar{x} = 23$	$\bar{x} = 74$
		$\bar{x} = 40–400$	(Díaz-Torres et al., 2017)		$\bar{x} = 47–740$
	0.8–500; $\bar{x} = 67$	40–2400; $\bar{x} = 690$	(Lebreton et al., 2018)[b]	1.4–926; $\bar{x} = 124$	56–3300 $\bar{x} = 960$
South China Sea		0–375,000[c]	(Uneputty and Evans, 1997)		0–694,500
	$\bar{x} = 8–20$	$\bar{x} = 2–370$	(Zhou et al., 2016)	$\bar{x} = 15–37$	$\bar{x} = 4–685$
South Pacific		0–35	(Thiel et al., 2003)		0–65
		0–240	(Hinojosa and Thiel, 2009)		0–445
		0–100	(Hinojosa et al., 2011)		0–185
	$\bar{x} = 1.9$	$\bar{x} = 17$	(Eriksen et al., 2014)[a]	$\bar{x} = 2.6$	$\bar{x} = 24$
Indian Ocean		$\bar{x} = 3.0–9.6$	(Ryan, 2013)		$\bar{x} = 6–18$
		$\bar{x} = 386$	(Ryan, 2013)		$\bar{x} = 715$
	$\bar{x} = 2.8$	$\bar{x} = 19$	(Eriksen et al., 2014)[a]	$\bar{x} = 3.9$	$\bar{x} = 26$
Arctic Ocean		0–0.2; $\bar{x} = 0.001$	(Bergmann et al., 2016)		0–0.4; $\bar{x} = 0.002$
Southern Ocean		0–1	(Barnes et al., 2010)		0–1.8
		$\bar{x} = 0.03$	(Ryan et al., 2014)		$\bar{x} = 0.06$

[a] Items >20 cm.
[b] Within the North Pacific Gyre.
[c] Note that these values were recorded from within a small embayment and scaled up.

et al., 2015). Seasonality in human behaviour (Ariza et al., 2008; Crosti et al., 2018; Gabrielides et al., 1991) and daily and seasonal changes to weather patterns also alters litter dispersal and accumulation patterns. Rainy seasons and high flow increase riverine litter input to the oceans (Crosti et al., 2018; Lebreton et al., 2017; Lima et al., 2014; Moore et al., 2011; van Emmerik et al., 2018, 2019). Strong winds may reduce local surface concentrations of plastics at sea, presumably due to increased vertical mixing and more rapid transport of particles (Collignon et al., 2012). High riverine discharge events, waves, currents and large tides may disperse debris (UNEP and GRID-Arendal, 2016). Stratification processes, wind and tidal currents are expected to affect the residence time and transport of plastics in estuaries (Kukulka et al., 2012; Sadri and Thompson, 2014). Consequently, litter which leaves a river or shore may not be available close to the coast for a very long time, nor present in sustained high concentrations over time. Removing litter close to its source before it disperses is most effective, yet a better understanding of the temporal dynamics of litter input is needed to implement clean-up technologies when CPUE is expected to be high. Acute pollution events, such as the 2011 tsunami in Japan, would require that clean-up technologies be available on short notice and the technology must be robust enough to handle the physical forces often associated with storm events and large litter items.

Table 3
A summary of available data on riverine plastics, excluding microplastics (*i.e.*, mesoplastics and larger). Densities reported are the estimated daily flux of litter passing downstream; studies which did not estimate daily discharge rates are excluded from the review. Range and mean values are given when available; if a range is given in mean values, this represents spatiotemporal variation. CPUE (catch-per-unit-effort) estimates are the maximum possible values based on intercepting litter in a cross-section of the river with 100% capture efficiency.

River(s)	Litter density (kg d^{-1})	Litter density (# d^{-1})	Litter fraction	Source(s)	CPUE (kg h^{-1})	CPUE (# h^{-1})
Jakarta, Indonesia (5 rivers)	200–1500		Plastics only.	(van Emmerik et al., 2019)	8–63	
Sungai Batu, Malaysia	690–4700 $\bar{x} = 2000$		All debris, anthropogenic and organic (8%)	(Malik and Manaf, 2018)	29–194 $\bar{x} = 87$	
Saigon River, Vietnam	$\bar{x} = 200–300$		Plastic (8%) and organic (91%)	(van Emmerik et al., 2019)	$\bar{x} = 8–13$	
Rhone River, France		0–226 $\bar{x} = 37$	Plastics only.	(Castro-Jiménez et al., 2019)		0–9 $\bar{x} = 2$
Seine River, France	60–99 $\bar{x} = 74$[a]		Plastic (1–5%) and organic (92–99%)	(Gasperi et al., 2014)	3–4 $\bar{x} = 3$	
Tiber River, Italy		$\bar{x} = 2000$	All anthropogenic debris, incl. wood	(Crosti et al., 2018)		$\bar{x} = 85$
Ems River, Germany	$\bar{x} = 2–8$		All anthropogenic debris, incl. wood and feces	(Schöneich-Argent et al., 2020)	0.1–0.3	
Weser River, Germany	$\bar{x} = 4–33$		All anthropogenic debris, incl. wood and feces	(Schöneich-Argent et al., 2020)	0.2–1.4	
Elbe River, Germany	$\bar{x} = 40–2200$		All anthropogenic debris, incl. wood and feces	(Schöneich-Argent et al., 2020)	2–91	

[a] Note that these values are estimates based on total debris amounts and the proportion of plastics in samples.

4. Ecological impacts

As with fishing gear, any marine litter clean-up technology will interact not only with its target plastics, but also with various marine life and habitats. Floating plastic particles are assimilated with both the planktonic community and floating organic debris. By numbers, floating microplastics are considerably more abundant than macroplastics (Eriksen et al., 2014), yet in most cases even the microplastics encounter rate will be lower than that of zooplankton (*e.g.*, Collignon et al., 2012; Di Mauro et al., 2017; Figueiredo and Vianna, 2018; Lima et al., 2014; Pazos et al., 2018; Suaria and Aliani, 2014). Microplastics generally constitute <1% of plankton tow samples in numbers (Di Mauro et al., 2017; Figueiredo and Vianna, 2018; Fossi et al., 2012; Lima et al., 2014; Pazos et al., 2018). Even in the Pacific Central Gyre, the abundance of zooplankton is five times greater than that of plastics, although by mass the relationship was more even (Moore et al., 2001).

The potential for by-catch, habitat destruction and effects on nutrient flows must be considered during technology development and deployment. In this paper, by-catch includes any encounter with the clean-up technology that can contribute to mortality of a stock either through interaction with the technology or by being removed from the water. In fisheries management, regulations are employed to reduce by-catch, such as temporal and spatial closures, and gear regulations to increase selectivity. Many fisheries have caps or quotas that define acceptable by-catch levels of non-target species (Campell and Cornwell, 2008). Similarly, technology targeting plastics should be evaluated in terms of their potential impact on organisms they may interact with. This requires an understanding of the relative abundance and distribution in time and space of marine life in relation to the target plastic debris, their likely interactions with the technology, and the consequences of these interactions.

Applying a macroplastics surface trawl or boom array to skim surface plastics will presumably affect mainly the upper few meters of the water column, and thus have a lower encounter rate with marine life than most fishing gear operating at the surface; a typical large pelagic finfish trawl has a vertical mouth opening of approximately 20 m (Suuronen et al., 1997). Peak zooplankton abundance, including that of larval fishes and krill, often occur tens of meters below the sea surface (Apango-Figueroa et al., 2015; Broms et al., 2016; Díaz-Astudillo et al., 2017; Gray et al., 2019; Tarling and Thorpe, 2017), suggesting that shallow surface tows should miss a considerable proportion of zooplankton assemblages. Nevertheless, even the top few meters below the surface provides habitat for a variety of plankton, fishes and larger vertebrates. Numerous zooplankton, including fish larvae, various juvenile fishes and small pelagic fishes are present in the top 5 to 10 m below the surface (Dänhardt and Becker, 2011; Freon, 1996; Hardy et al., 1987; Krutzikowsky and Emmett, 2005), and these assemblages may be compositionally different from those in deeper layers (Gray et al., 2019). Air-breathing megafauna, such as marine mammals, turtles and seabirds, also use the sea surface for daily activities. The critically endangered northern right whale (*Eubalaena glacialis*), for example, spends over 70% of its time in the top 10 m of the water column and is highly susceptible to ship strikes and fishing gear entanglement (Baumgartner et al., 2017). Faster-swimming whales, such as fin (*Balaenoptera physalus*) and Bryde's whales (*B. edeni brydei*), are also at risk (Ebdon et al., 2020; Panigada et al., 2006).

Even if the proposed technology targets larger plastic items, there may be negative impacts on plankton. For example, the Ocean Cleanup's feasibility study predicted that the majority of zooplankton hitting the skirts made to guide the plastics into the collection device are likely to be killed despite not being removed from the water (Slat et al., 2014). Assuming the same operational parameters as used to estimate macroplastics CPUE (see Table 2), plankton tow data suggest potential encounter rates of 0.8–40 billion zooplankton hourly (Jacobsen et al., 2018; Yang et al., 2017). A year of continuous plastics removal (12-h day^{-1}, 365 days) could equate to 675 t of impacted zooplankton (based on Yang et al., 2017). The potential consequences of such mortality are difficult to predict, but may be negligible depending on the total effort of plastics removal and zooplankton stock size. For the Norwegian copepod (*Calanus finnmarchius*) fishery, a maximum annual harvest of 10% of the breeding stock is recommended, which is estimated to 1.65 million tonnes (Broms et al., 2016). However, given the generally low vertical resolution of zooplankton sampling in the upper 15 to 50 m of the water column, as well as substantial spatiotemporal variation in density (*e.g.*, Gislason and Astthorsson, 1995; Planque and Ibanez, 1997; Dorman et al., 2015; Li et al., 2016; Yang et al., 2017; Jacobsen et al., 2018), there is uncertainty associated with estimates of zooplankton encounter rates and further research will be necessary during technology development.

Potential encounter rates with fishes are also challenging to predict and will be subject to considerable spatiotemporal variation. Possible impacts may be direct through mortality of all life stages; indirect impacts through increased mortality of prey are also possible. Mortality of fish eggs and larvae in the plankton will likely represent only a small proportion of the total abundance, and population-level impacts may be negligible. Harvesting 2.5 million cod eggs and 2.3 million larvae in the Norwegian Sea is, based on mortality estimates of different life stages, only expected to result in a 0.7% decrease in year class size three years later (Broms et al., 2016). There are few available data on the fine-scale vertical distribution of pelagic fish. However, densities reported in the upper 3 m of the water column in two locations in the Wadden Sea (Dänhardt and Becker, 2011) suggest possible encounter rates of 65,000 to 400,000 fish per hour (assuming the same operational parameters as previously, see Table 2). If present in the path of a trawl, fish are not necessarily very successful at avoiding it. Herring (*Clupea harengus*), for example, display strong avoidance reactions to trawls only about 15% of the time (Suuronen et al., 1997), although the avoidance rate may be considerably higher for a trawl targeting plastics that has a much smaller vertical profile than a fisheries trawl. With respect to potential indirect impacts of fish stocks, modelling studies predict that an annual harvest of 3.4 million tonnes of *C. finnmarchius* will have negligible effects on the growth rates of mature herring (*Clupea harengus*), mackerel (*Scomber scombrus*) and blue whiting (*Micromesistius poutassou*) in the Norwegian Sea (Broms et al., 2016). However, only adult fishes were considered in the model; juveniles may be more vulnerable to increased mortality of zooplankton prey species. Recruitment of Atlantic cod (*Gadus morhua*) in the North Sea, for example, depends on the availability of energy-rich zooplankton prey, such as *C. finnmarchius* (Nicolas et al., 2014). Given the high mortality at earlier life stages, damage or by-catch to juvenile and adult fish, particularly schooling species, may be the greatest concern in implementation of clean-up technology.

There are several mechanisms by which the by-catch of non-target organisms may be reduced. Limiting the vertical profile of a clean-up device decreases likely interactions with non-target species. Assuming tow speeds of 1.5–2 knots (as per the preliminary parameters of PGS' proposed clean-up technology (Falk-Andersson et al., 2018)), active clean-up technologies may be towed somewhat slower than most pelagic finfish trawls (Krutzikowsky and Emmett, 2005; Kvalsvik et al., 2002; Suuronen et al., 1997) and may allow greater avoidance by fish. Slow tow speeds of active gear will also reduce the risk of whale strikes (Ebdon et al., 2020; Panigada et al., 2006; Wiley et al., 2011). However, the possibility that a slowly towed collector may act as a Fish Aggregating Device (FAD) should also be considered. FADs typically consist of a floating structure and a component extending down into the water column (*e.g.*, "curtain" nets), and drifting FADs may naturally move with currents at speeds upwards of 1 knot (Imzilen et al., 2019). Tow and current speed will also affect the efficacy of plastics collection, and optimising plastic CPUE may be in conflict with reducing by-catch. The skirts of a passively drifting boom array have been estimated to float horizontally and fail to guide plastics towards the collection unit at current speeds of only 0.5 knots (Slat et al., 2014). In addition to operational parameters, gear design itself may reduce by-catch. A variety of by-catch reduction devices (BRDs), such as sorting grids or devices utilising behavioural differences among species, are common in commercial trawl fisheries (Bayse and He, 2017; Hines et al., 1999; Kvalsvik et al., 2002).

The timing of operations may also be used to limit by-catch as the density of various marine organisms near the surface is extremely temporally variable (*e.g.*, Dänhardt and Becker, 2011; Gray et al., 2019; Luo et al., 2000). Zooplankton density tends to follow seasonal patterns, although the strength and nature of these vary (Gislason and Astthorsson, 1995; Li et al., 2016; Planque and Ibanez, 1997). Zooplankton by-catch could vary eight-fold between seasons (based on Luo et al., 2000). Negative impacts on pelagic eggs and larvae can be reduced by avoiding known spawning grounds, including areas upstream of these, during relevant times of the year (Broms et al., 2016). Pelagic fish are less successful at avoiding trawls at cold temperatures (<6–8 °C) and in the dark (Sajdlová et al., 2015; Suuronen et al., 1997; Williams et al., 2015) and by-catch may increase in winter and at night. Many zooplankton and fishes exhibit diel vertical migrations where they are present in the upper water column during the night and retreat to deeper water during the day (*e.g.*, Cardinale et al., 2003; Cisewski et al., 2010; Freon, 1996; Luo et al., 2000; Putzeys and Hernández-León, 2005), although these migrations may vary among species, life stages, spatially and seasonally (Ambriz-Arreola et al., 2017). Avoiding known feeding grounds of filter feeding whales and sharks would reduce the risk to these animals, but may not be desirable given that high plastic concentrations often coincide with feeding grounds, presumably because surface currents concentrate both plastics and zooplankton in similar areas (Fossi et al., 2016; Reisser et al., 2015). In conclusion, a good understanding of the spatial and temporal distribution of non-target species, is important to reduce by-catch rates. An alternative approach to reducing by-catch from clean-up technologies is to focus instead on removing litter caught as by-catch in traditional fisheries, such as in the Fishing For Litter scheme (Ronchi et al., 2019).

Not only by-catch, but also potential impacts on habitat must be considered. The indirect impact of certain fishing activities on habitats, including those important to the life cycle of commercially important species and biodiversity conservation in general, has resulted in the integration of habitat protection in marine management plans (Armstrong and Falk-Petersen, 2008). Floating mats of the brown macroalgae Sargassum, for example, is considered Essential Fish Habitat by the US Marine National Fisheries Service (Wells and Rooker, 2004). Bacteria and various sessile invertebrates colonise the algae, and several species of fish, shrimp and crabs are believed to depend on the shelter of these mats during early life stages (Rooker et al., 2006; Wells and Rooker, 2004). Similarly, organic matter ending up in rivers contributes significantly to many riverine food webs along the entire length of the river (Thorp and Delong, 1994; Wipfli et al., 2003). River discharge is also an important source of nutrients to coastal waters and ecosystems (Venkataramana et al., 2017). Thus, removal of organic debris as by-catch in litter clean-ups may have an impact on floating habitat in rivers as well as disrupting the flow of nutrients and associated ecosystems downstream (Vörösmarty et al., 2005; Yeakley et al., 2016). The surface microlayer of oceans, rivers and lakes also hosts a unique ecosystem, the neuston. These organisms can influence air-sea exchange processes, including processes affecting the global climate (Zaitsev and Liss, 2005). Anthropogenic litter may also itself be a habitat (Barnes and Milner, 2005; Goldstein et al., 2014; Kiessling et al., 2015). Humankind has long used artificial habitats to increase the harvest of plants and animals (Seaman and Sprague, 1991). Any disruption of habitat by clean-up technologies, be it organic or anthropogenic, merits careful consideration. A high encounter rate with organic matter may also affect the efficiency and viability of the technology itself, through for example clogging.

There are relatively few data with which to compare the relative densities and distributions of floating anthropogenic litter *versus* organic matter. In marine environments, in studies where all debris was assessed, the proportion of organic debris reported ranges from 12% to 30% in the Mediterranean (Arcangeli et al., 2018; Compa et al., 2019). Along the coast of Chile, floating macroalgae generally outweigh anthropogenic litter, although there is considerably spatiotemporal variation in both (Hinojosa et al., 2011). Of the 7 studies of floating macroplastic in rivers reviewed, only 3 reported the percentage of organic debris (Table 3). In studies where all debris was recorded, the proportion of total intercepted debris which was organic rather than anthropogenic ranged from <10% to >90% (Gasperi et al., 2014; Malik and Manaf, 2018; van Emmerik et al., 2018, 2019; Wan et al., 2018). A lack of reporting on the ratio of plastics to organic debris, makes it difficult to evaluate the potential ecological impact of clean-up technologies through removal of organic matter. Patchiness and seasonality could also impact this ratio. In the Saigon River, for example, the density of plastics is expected to be considerably higher, and the density of organic materials considerably lower, in the fall compared to the spring (van Emmerik et al., 2018). The removal of organic matter may therefore be limited by implementing clean-up schemes only during certain times of the year.

5. Maximising the net gain of clean-up technologies

To maximise the net gain of clean-up technologies, one must target geographic areas and times which both maximises CPUE and minimises negative environmental impacts. However, limited empirical data on spatiotemporal variation of floating macrolitter, combined with gaps in our understanding of their transport and fate, makes predicting CPUE challenging. The number of observations is also small given the size of the ocean and river basins. Of available empirical data, only about half of the ocean studies were conducted the past 5 years and can be regarded reasonably representative of the current situation (Table 2). Further compounding the issue is a lack of standardisation of sampling methods among empirical studies, which makes direct comparisons to elucidate spatiotemporal variation problematic. Additionally, most empirical studies report densities only in item counts, which gives no consideration of item size; few data enable CPUE estimates by weight (Table 2). Consequently, the data available for estimating CPUE of floating litter, both at the sea surface and in rivers, are insufficient to evaluate the potential catch efficiency of clean-up technologies.

All evidence suggests considerable spatiotemporal variation in litter density and environmental impacts, and that any clean-up technology will benefit from being adaptable to this variation. In general, however, the sustained CPUE is expected to be the highest close to point sources, (*i.e.*, areas or events with high discharge levels of plastics to the oceans, for example in rivers and close to shore in highly populated regions where litter is generated and mismanaged on land, and along shipping lanes/on key fishing grounds), particularly during high discharge events. Placing clean-up infrastructure in areas with constant influx, and having a response system in place for high discharge events, would likely be the most efficient (in terms of CPUE) over time and could significantly decrease the amount of plastic litter entering the open ocean. Reliably identifying these locations and times, however, requires extensive data, which are currently lacking. While various global point sources have been identified through modelling studies (Jambeck et al., 2015; Lebreton et al., 2017), there remain relatively few accompanying empirical data by which to estimate CPUE in their proximity. Rivers may also be considered point sources, particularly in countries with poor waste management infrastructure, but the relevant geographic regions are underrepresented in the few studies documenting macrolitter in rivers (Blettler et al., 2018).

In addition to litter density, litter composition may also affect CPUE. For example, some items washed out to sea by storm events may be too large for most clean-up technologies currently under development. Debris from the 2011 tsunami event in Japan, for example, included very large items such as fishing boats and docks (Bagulayan et al., 2012; Murray et al., 2015). The relative abundance of anthropogenic *vs.* organic debris, and of plastics within the anthropogenic litter, also varies in space and time (*e.g.*, Hinojosa et al., 2011; van Emmerik et al., 2018), with significant implications for the efficacy and impacts of clean-up technology. Variable reporting of densities of different debris fractions further complicate these considerations; not only is organic debris often not reported, but it is highly variable whether studies report plastic or general anthropogenic litter densities, or both. Despite these challenges, however, only 2 of the 12 clean-up technologies reviewed conducted a feasibility study to estimate expected CPUE or capture efficiency of encountered plastic during the development phase (Table 1).

Efforts to remove floating plastic pollution will almost certainly have a direct or indirect impact on biota. Clean-up technologies should therefore be constructed and deployed in a way that minimises this, and the developmental process must be accompanied by thorough Environmental Impact Assessments (EIAs), yet such assessments appear to be rare. Of the 12 technologies reviewed (Table 1), a feasibility study (Slat et al., 2014) and later a limited EIA was disclosed only for the Ocean Cleanup's floating array (CSA, 2018) (both widely criticised (Martini, 2014; Helm, 2019)) and a preliminary desktop EIA for the PGS bubble curtain tow (Falk-Andersson et al., 2018). An EIA was also conducted for the Ocean Cleanup Interceptor, but has not been made publicly available (pers. comm. Niels van Geenhuizen, Arcadis N.V. 2020). Granting access to EIAs for external evaluation is important in order to secure the credibility of such assessments. In general, larger and/or widely used technologies are expected to have greater potential for negative environmental impacts than smaller, less efficient infrastructure with limited geographical replication. More passive devises, such as those making use of the natural flow of rivers, could limit the negative impact on ecosystems and reduce the effort of collection. However, EIAs should be conducted also for small-scale and passive technologies, particularly if the objective is an extensive upscaling of their use.

Currently, there is insufficient information on the clean-up technologies developed or in development in order to do full environmental cost-benefit analyses. Given the strong focus in fisheries management on developing and operating gear technology in a manner that minimises harm on the environment (*e.g.* Beutel et al., 2008; Gullestad et al., 2015; Watson et al., 1999), it is worrying that such concerns are not similarly integrated in the development and application in plastic clean-up technologies at an early stage, particularly for large-scale (both in size and application) technologies. EIAs should include an evaluation of maximum acceptable by-catch and encounter rates for different taxa and key species during operation, as well as how to regularly assess whether these are exceeded. Additionally, such assessments should include an evaluation of the minimum acceptable CPUE of target litter that still results in a net positive environmental impact. Furthermore, the impact of clean-up technologies has to be seen in context of the multiple stressors affecting ecosystems today.

Lastly, during the development of clean-up technologies and solutions, one should consider in which marine compartment it is actually most cost-effective and least destructive to clean. While there is no doubt that considerable amounts of plastics are afloat in our oceans (Eriksen et al., 2014), there may be higher concentrations, more readily available litter, and/or easier to clean litter in other environments. Densities of up to 800,000 plastic items km^{-2} have been found on European seafloors (Enrichetti et al., 2020), for example, although removal with active gear may result in considerable by-catch and habitat damage as known from bottom trawling (Buhl-Mortensen and Buhl-Mortensen, 2018). The density of beached litter can also be high, and has been reported to exceed densities of nearby floating litter by an order of magnitude (Thiel et al., 2013). On Indonesian beaches, macroplastics have been reported in densities of 6,400,000 items km^{-2} (Sulistiawati et al., 2020), although beach litter density also shows considerable spatial variability, including over small scales (*e.g.*, Haarr et al., 2019; Thiel et al., 2013). Manually hand-picking beach litter largely allows the elimination of negative environmental impacts, especially if clean-ups are timed to avoid periods when resident organisms are vulnerable (*e.g.*, birds nesting), and limits damage to habitat (*e.g.*, avoids excessive digging out of litter integrated in vegetation). Given at times limited resources for technology development, one should also consider whether investments in one type of solution hinder the development and implementation of other, possibly more efficient, solutions (Stokstad, 2018).

6. Conclusions

We have pointed to key fisheries management principles which should be considered during development and initiation of surface clean-up technologies, in rivers and at sea, and which are equally relevant for clean-up technologies in other ecological compartments. Firstly, the density and availability of litter should be evaluated to determine the catch efficiency of technological devices in time and space. Secondly, the clean-up technology should be evaluated in terms of its potential negative environmental impacts through by-catch or habitat damage, and how these may be minimised. It is important that these concerns are integrated in development and evaluation of clean-up technologies from the beginning to secure efficiency and minimise harm. We have also identified important knowledge gaps with respect to evaluating where, if and under which conditions clean-up technologies should be applied, as well as on the spatiotemporal distribution of macroplastics and its overlap with organic debris and biota. Both environmental and socioeconomic impacts of clean-up technologies should be further studied.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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