A whale of a plastic tale: A plea for interdisciplinary studies to tackle micro- and nanoplastic pollution in the marine realm

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Review

A whale of a plastic tale: A plea for interdisciplinary studies to tackle micro- and nanoplastic pollution in the marine realm

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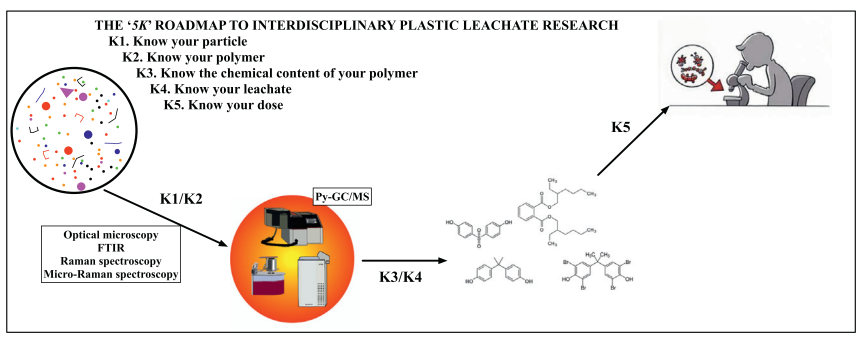
HIGHLIGHTS

- Plastics are releasing a wide variety of chemical compounds.
- Plastic pollution research should intrinsically be highly interdisciplinary.
- We show that this is still far from being the case in leachate studies.
- Plastic pollution research is essentially structured in monodisciplinary studies.
- We praise for a systematic chemical definition of what we biologically compare.

GRAPHICAL ABSTRACT



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ABSTRACT

Plastic is one of the most ubiquitous sources of both contamination and pollution of the Anthropocene, and accumulates virtually everywhere on the planet. As such, plastic threatens the environment, the economy and human well-being globally. The related potential threats have been identified as a major global conservation issue and a key research priority. As a consequence, plastic pollution has become one of the most prolific fields of research in research areas including chemistry, physics, oceanography, biology, ecology, ecotoxicology, molecular biology, sociology, economy, conservation, management, and even politics. In this context, one may legitimately expect plastic pollution research to be highly interdisciplinary. However, using the emerging topic of microplastic and nanoplastic leachate (i.e., the desorption of molecules that are adsorbed onto the surface of a polymer and/or absorbed into the polymer matrix in the absence of plastic ingestion) in the ocean as a case study, we argue that this is still far from being the case. Instead, we highlight that plastic pollution research rather seems to remain structured in mostly isolated monodisciplinary studies. A plethora of analytical methods are now available to qualify and quantify plastic monomers, polymers and the related additives. We nevertheless show though a survey of the literature that most studies addressing the effects of leachates on marine organisms essentially still lack of a quantitative assessment of the chemical nature and content of both plastic items and their leachates. In the context of the ever-increasing research effort devoted to assess the biological and ecological effects of plastic waste, we subsequently argue that the lack of a true interdisciplinary approach is likely to hamper the development of this research field. We finally introduce a roadmap for future research which has to evolve through the development of a sound and systematic ability to chemically define what we biologically compare.

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Contents

1. A foreword on plastic pollution

Plastic was initially considered a revolutionary material with nearly infinite applications that were anticipated to propel mankind into ‘a plastic era’ (Yarsley and Couzens, 1942). It quickly became the material supporting a wasteful and throw-away lifestyle, ‘the savior of the American housewife’ (Life Magazine, 1955), a description which, today, would be, at the very least, seen as a fundamentally politically incorrect marketing campaign. This prediction laid on the remarkable properties of this material, which is light-weight, highly durable and versatile, and relatively cheap to produce (Geyer et al., 2017). This plastic tale may, however, only be the beginning of an endless and not so happy ending. Plastic contamination and pollution have soon gained the other side of the same coin. Due to plastic abuse and the waste management systems worldwide struggling to cope the increasing influx of plastic waste (Jambeck et al., 2015; Issufu and Sumaila, 2020), this material has become one of the most ubiquitous sources of both contamination and pollution of the Anthropocene. Plastic wastes are accumulating on landfills and beaches worldwide as well as in the most remote terrestrial and oceanic locations (van Sebille et al., 2015; Napper et al., 2020), threatening both terrestrial and aquatic environments, the economy and human well-being on a global scale (Marks et al., 2020; Frias et al., 2021; Kumar et al., 2021).

The extent of this pollution is such that plastics are now considered as a key geological indicator of the Anthropocene as a distinctive stratal component (Corcoran et al., 2014; Zalasiewicz et al., 2016; Dibley, 2018). The term ‘Plasticene’ (i.e., sensu stricto the ‘Age of Plastics’), initially introduced in the context of the societal implications of plastic pollution (Stager, 2011; Reed, 2015), has since been increasingly used in the research and management literature (e.g., Mendoza et al., 2018; Gestoso et al., 2019; Tiller et al., 2019). The rapidly accelerating field of plastic research—and the related diversity of new terms used by researchers, resource managers, policy-makers, and the general public working on plastic-related issues—even recently seeded a ‘Plasticene Lexicon’ aiming at advancing standardization of language, and subsequently the research and concepts they describe (Haram et al., 2020).

Yearly plastic pollution exponentially increased from 1.5 million tons in 1950 to 368 million tons in 2019 (PlasticsEurope, 2020) and the staggering 9 billion tons produced so far are expected to increase sixfold by the mid-21st century (Zalasiewicz et al., 2019). Overall, 10 % of the produced plastic is expected to end up in the ocean (Halpern et al., 2008; Barnes et al., 2009), where ca. 80 % of the plastic litter is of terrestrial origin (Andrady, 2011), comprising 60 to 80 % of the overall anthropogenic litter (Derraik, 2002). In 2015, 15 to 51 trillion pieces of plastic (93,000 to 236,000 tons) were estimated in the oceans (van Sebille et al., 2015; Isobe et al., 2021), a number that has been predicted to rise by an order of magnitude by 2025 as, every year, 4.8 to 12.7 million metric tons of plastic waste are continuously entering the oceans (Jambeck et al., 2015; Williams and Rangel-Buitrago, 2022).

Beyond the above-mentioned earth-shattering figures, plastic pollution has become a fertile research ground for scientists as diverse as chemists, physicists, oceanographers, biologists, ecologists, ecotoxicologists, molecular biologists, sociologists, economists, environmentalists, conservationists, managers, policy makers, politicians, and even citizens who are increasingly solicited through a variety of litter monitoring programs (Hidalgo-Ruz and Thiel, 2015). In this context, one may legitimately expect plastic pollution research to be highly interdisciplinary. Using the ever-growing topic of micro- and nanoplastic pollution and the emerging plastic leachate research—i.e. the desorption of molecules that are adsorbed onto the surface of a polymer and/or absorbed into the polymer matrix in the absence of plastic ingestion (Delaeter et al., 2022)—in the ocean as a case study, we argue that this is still far from being the case, and introduce a roadmap for future research. To address this issue in detail, we first used the Web of Science to assess the relative status of both plastic pollution research and plastic leachate research (Section 2). Subsequently, Section 3 illustrates how microplastics and nanoplastics may be considered as a more pernicious threat for marine organisms and ecosystems than macroplastics due to both their minute size and the fact that they tend to accumulate more persistent pollutants than large debris. In Section 4, we highlight how leachates may be considered as a somewhat sickly child of micro- and nanoplastic research despite their potential significant impact on marine life and ecosystems. Section 5 reviews the current state of the so-called plastic leachate literature, highlighting how the vast majority of the handful paper that explicitly assessed the effect of plastic leachates on various aspects of the biology of aquatic organisms did not conduct any sort of screening analysis on either the nature of the plastic polymer considered, their chemical content nor the nature of their leachates. Finally, in Section 6, we discuss that, given the growing availability of screening methods to assess the nature of plastic polymers and their chemical contents, the future of the growing research effort devoted to assess the effects of plastic waste lies in our ability to develop a sound and systematic approach to chemically define what we biologically compare. To that end we introduce a 5-step knowledge roadmap as an attempt to guide the future development and implementation of plastic leachate research towards a true interdisciplinary research field.

2. Plethoric plastic pollution research vs. a *still* anecdotal plastic leachate research

Over the last few decades, plastic pollution has become one of the most prolific field of research. The Web of Science (accessed June 30, 2022) returned a total of 45,471 articles containing the words *plastic* (i.e., ‘**plastic**’) and *pollution* (i.e., ‘*pollut**’) in their topic between 1950 (the beginning of industrialization of plastic production) and 2022, a figure that has been highly significantly exponentially growing ($r^2 > 0.99$, Fig. 1A). To assemble the literature search, we deliberately used the word *pollution*, instead of *contamination*—though they are often used interchangeably in the ocean litter literature—as there is a critical need to separate the anthropogenic presence of the materials (i.e., contamination) from pollution, the adverse biological effects at one of more levels of biological organization, that is sensu *lato* pollution (Borja and Elliott, 2019). Overall, these 45,471 articles were predominantly related to 3 subject areas (“*Environmental Sciences and Ecology*”, “*Public Environmental and*

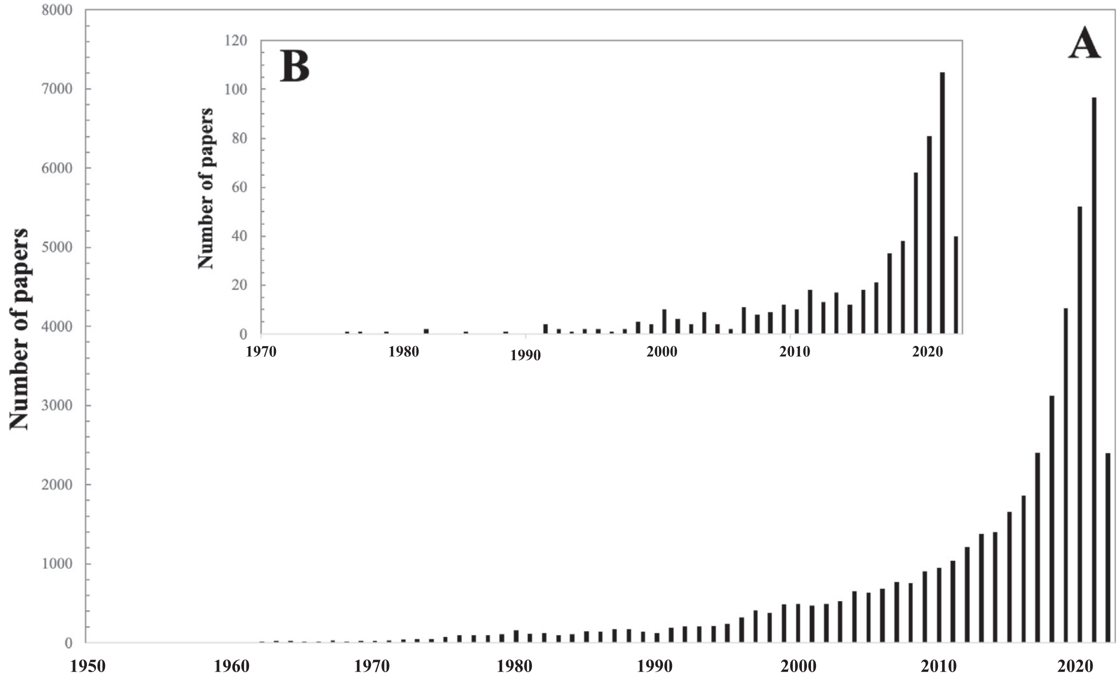


Fig. 1. Web of Science search (accessed June 30, 2022) of the articles containing the words '*plastic*' and '*pollution*' in their topic between 1950 and 2022 (A), and the words '*plastic*', '*pollution*' and '*leachate*' in their topic between 1950 and 2022 (B).

Occupational Health", and "*Toxicology*"; Fig. 2A). When using the words *plastic* and *pollution* and *leachate* the above-mentioned figure drastically drops down to 579 articles, which have a much more recent history (Fig. 1B), but also appear to be spread over a wider range of research areas (Fig. 2B). While these results may point towards a high interdisciplinary trend in the areas of research dealing with plastic pollution, we develop hereafter the idea that this is still not quite the case using the subject area "*Marine Ecology*" as a case study. Instead, we highlight that plastic pollution research rather seems to remain structured in mostly isolated monodisciplinary studies. Using the example of emerging topics on microplastics (i.e., particles smaller than 5 mm) and nanoplastics (i.e., particles smaller than 100 nm), we highlight how, in the face of the increasing research effort assessing the effects of plastic waste, the lack of a true interdisciplinary approach is likely to hamper the development of this research field.

3. Microplastics and nanoplastics as the invisible portion of the plastic iceberg

The public—and to a certain extent the scientific—perception of plastic pollution has strongly been biased towards the description and quantification of the presence, amount, nature and origin of large (typically >5 mm) plastic debris. For instance, the 'garbage patches' found in the Atlantic Ocean (Ryan, 2014) and Pacific Ocean (Lebreton et al., 2018)

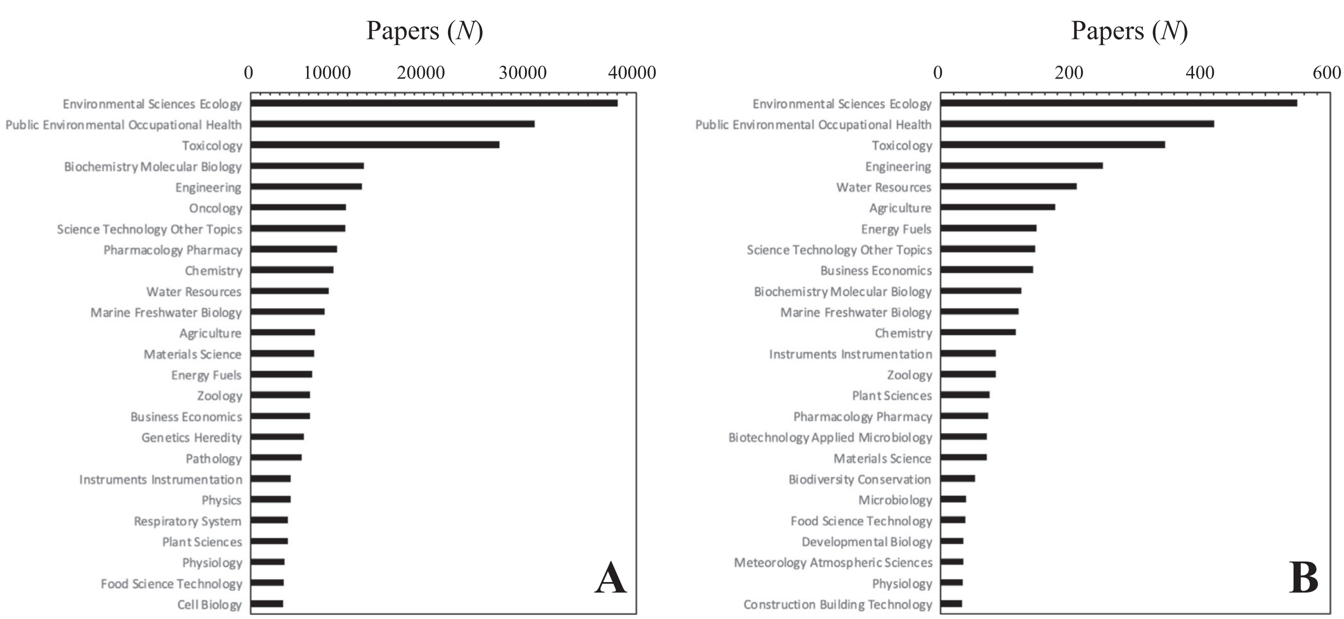
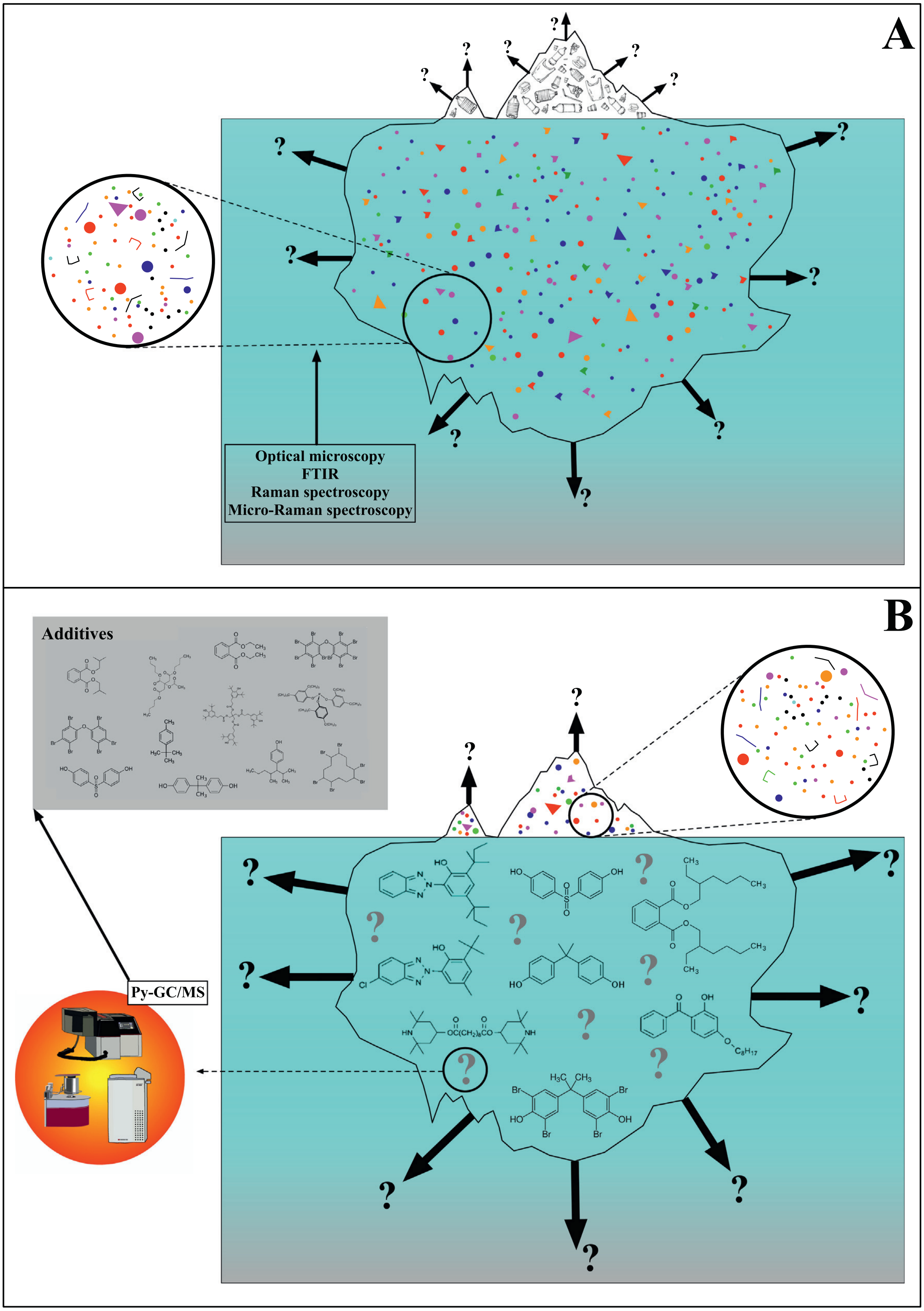


Fig. 2. Web of Science search (accessed June 30, 2022) showing the number of articles containing the words '*plastic*' and '*pollution*' in their topic between 1950 and 2022 (A), and the words '*plastic*', '*pollution*' and '*leachate*' in their topic between 1950 and 2021 (B) as a function of the 25 most represented research areas.


A
Optical microscopy
FTIR
Raman spectroscopy
Micro-Raman spectroscopy
B
Additives
Py-GC/MS

are some of the most infamous examples of plastic waste accumulation in oceans, that, together with what is found on the seafloor and wash up on beaches, is threating marine animals mostly in the form of entanglement, smothering and ingestion (Van Franeker et al., 2011; Kühn et al., 2015; Li et al., 2016; Thiel et al., 2018). Despite the ever-increasing number of scientific articles assessing plastic pollution (Fig. 1) and the conspicuous nature of large plastic debris (Fig. 3A), the quantification of their overall amount that either floats or accumulates on seashores and at the bottom of the ocean remains a daunting task. Global inventories are indeed (i) fundamentally limited by both the horizontal and vertical extent of the ocean, hence de facto rely on limited amount of actual concentration measurements, which are typically upscaled in model simulations (Cózar et al., 2014; Eriksen et al., 2014; Lau et al., 2020; Koelmans et al., 2017), and (ii) further complicated by the uneven distributions of debris (Halsband and Herzke, 2019; Lacerda et al., 2019; van Sebille et al., 2020). Coastal debris fundamentally undergoes cycles of beaching and remobilization (Lebreton and Andrady, 2019) that add a temporal dimension to its geographical unevenness. In addition, there is no standard definitions and protocols to qualify and quantify beached debris (Browne et al., 2015; Underwood et al., 2017). As a consequence, there still are noticeable uncertainties in the assessment of the fate of discarded plastics (Jambeck et al., 2015; Geyer et al., 2017; Williams and Rangel-Buitrago, 2022).

Microplastics (i.e., particles smaller than 5 mm; MPs), and in particular nanoplastics (i.e., particles smaller than 100 nm; NPs), have received an increasing scientific interest over the last decade (Thompson, 2015; Koelmans et al., 2015; Gigault et al., 2018; Mattsson et al., 2018; Ferreira et al., 2019; Gangadoo et al., 2020; Peng et al., 2020; Piccardo et al., 2020; Sana et al., 2020). These forms of plastic litter are now virtually found everywhere in the ocean (Chiba et al., 2018; Barrett et al., 2020). They are also considered by far the most numerically abundant form of solid waste on the planet (Eriksen et al., 2014) with maximum concentrations ranging from 10 to 10^2 particles l^{-1} for MPs (Di Mauro et al., 2017)—but typically in the range of 0.01 to 1 particles l^{-1} with a marked increase in concentration with decreasing size (Lenz et al., 2016; Alfaro-Núñez et al., 2021)—and 10^{10} to 10^{12} particles l^{-1} for NPs (Gallego Urrea et al., 2010). Note that microplastics have also been found in marine sediments, with concentrations ranging from 0.04 to 13.6 particles per gram of dry sediment (Barrett et al., 2020).

Despite the increasing scientific attention towards plastic pollution and contamination, no standard protocols exist to detect, enumerate and characterize microplastics (Löder and Gerdts, 2015). This fact may explain the observed discrepancy between MPs counts in the oceans (Andrady, 2017; Law, 2017; Fig. 3A). The situation is even more critical for NPs as they are still not accounted for in global plastic estimates due to a relative lack of detection and/or quantification techniques (Gigault et al., 2016; Ter Halle et al., 2017; Schwaferts et al., 2019; Rai et al., 2021). As such they may represent a significant proportion of the so-called 'missing plastic paradox' that has been riddling scientists since the seminal work of Richard Thompson's team in 2004 (Thompson et al., 2004). MPs and NPs are likely to have been critically under-estimated, hence their abundance will increase with the improvement of qualification and quantification methods. This issue is exacerbated by the ever-increasing evidence that micro- and nanoplastic pollution is ubiquitous, ranging from high-mountain lakes to deep-sea sediment; see e.g. Woodall et al. (2014), Napper et al. (2020), Rowlands et al. (2021), and Pastorino et al. (2022).

MPs and NPs may represent a far more pernicious threat for marine organisms and ecosystems than macroplastics (Galloway et al., 2017). The wide variety of deleterious effects that these smaller forms have on the environment and human health are, however, still debated (Chae and An, 2017; Galloway et al., 2017; Revel et al., 2018). In particular, it is stressed that the acute lethal and sub-lethal damages (e.g., entanglement, smothering and ingestion) caused by macroplastics to large marine organisms with low population growth rates are barely comparable to the chronic effects MPs and NPs may have on microorganisms such as copepods which are characterized by fast population growth rates and are fundamentally exposed to plastic items at much lower concentrations (i.e., 0.01 to 1 particles l^{-1}; Lenz et al., 2016; Alfaro-Núñez et al., 2021) than their actual microphytoplankton, which typically range from ca. 10^2 and 10^4 cell l^{-1} in offshore waters and 10^3 and 10^7 cell l^{-1} in estuarine and coastal waters; see e.g., Cloern (1996), Continuous Plankton Recorder Survey Team (2004), O'Boyle and Silke (2010) for reviews. In this context, it is noticeable that the vast majority of studies assessing the bioavailability of various microplastic items (i.e., beads, fibers and fragments) used concentrations orders of magnitude above MPs environmental concentrations; see Bai et al. (2021), their Table 1. As such, given the current limited and lack of knowledge of MP and NP concentrations in the ocean, any attempt to compare the relative effects of plastics at the base and the top of the food chain would be at best speculative. It is nevertheless stressed that though the resolution of this issue warrants the need for further work, it lies well beyond the scope of the present work.

MPs and NPs are typically divided into primary and secondary according to their sources. Primary MPs and NPs are released into the environment as nurdles, pellets and granules, biobeads, fibers either accidently or intentionally (Vandermeersch et al., 2015; Gasperi et al., 2018; Dris et al., 2016; Zhang et al., 2020). These plastics are produced as intended products, wastes from manufacturing processes, or derivatives from the erosion and tearing in the use of large plastic products such as tires, wheels and boards. Noticeably, increasing amounts of NPs have been manufactured for a variety of applications such as 3D printing, cosmetics, drug delivery and some of them are inevitably discharged into the environment (Peng et al., 2020). Secondary MPs are derived from the breakdown of large plastic litter in the environment under the action of physical, chemical and biological forces (Thompson, 2015; Ryan et al., 2009) such as mechanical abrasion, heat, UV-radiation light, oxidation, and biodegradation (Barnes et al., 2009; Browne et al., 2011; Imhof et al., 2012; Rillig et al., 2017). NPs also derive from the fragmentation and weathering of MPs (Gigault et al., 2016; Dawson et al., 2018; Ekvall et al., 2019; Enfrin et al., 2020; Piccardo et al., 2020; Wang et al., 2021), just as MPs are mostly derived from the fragmentation of macroplastics.

MPs and NPs have a radically different impact on both marine environments and organisms than larger pieces of plastic due to their abundance,

Fig. 3. Schematics of the visible and invisible portions of (A) the 'plastic iceberg' and (B) the 'micro- and nanoplastic iceberg'. Specifically, the visible portion of the plastic iceberg is made of highly conspicuous macro-debris that accumulate at the surface and the bottom of the ocean, as well as along the seashore, while its invisible portion is made of minute-sized debris that can barely be seen through the naked eye and often request tools ranging from optical microscopes to increasingly expansive pieces of equipment such as Fourier-transform infrared spectrometer (FTIR), Raman and micro-Raman spectrometers to quantify their size, shape, color and polymer. In contrast, the invisible part of the micro- and nanoplastic iceberg is made of the additives that are released from virtually any plastic item present in the ocean, and requests very specific tools such as pyrolysis-gas chromatography–mass spectrometry packages to identify and quantify them. Note that though the identification and quantification of known plastic additives, here clockwise a plasticizer (DEHP) a UV-light stabilizer (Chimassorb 81), a flame retardant (TBBPA), and three UV-light stabilizers, Tinuvin 770, 326 and 328; the two middle molecules are two common antioxidants, i.e. from top to bottom bisphenol S (BPS) and bisphenol A (BPA), this area of research necessitate continuous analytical development and improvement given the ever-increasing number of plastic additives used by the plastic industry (see www.echa.europa.eu). The thickness and the length of the black arrows in both panels are respectively an indication of the expected quantity and the related level of uncertainty in our understanding of the quantity of plastic debris (A) and plastic additives (B) present in the oceans. In addition, the question marks highlight the fact that the quantification of plastic items discarded in marine ecosystems is fundamentally limited by the horizontal and vertical extent of the ocean (hence essentially relies on a limited amount of actual concentration measurements and upscaling model simulations) and temporal and spatial unevenness of the distribution of plastic debris.

Table 1

Web of Science search (accessed June 30, 2022) of the articles containing the words '*plastic*', '*pollution*' and '*leachate*' in their topic in the Marine Biology Research area. List of abbreviations for chemical compounds. HCB: hexachlorobenzene; PAH: polycyclic aromatic hydrocarbons; PBDE: polybrominated diphenyl ethers; PCB: polychlorinated biphenyls; PE: polyethylene; PET: poly(ethylene terephthalate); PP: polypropylene; PS: polystyrene; PUR: polyurethane; PVC: polyvinyl chloride. List of abbreviations for analytical methods. ATR-FTIR: attenuated Total Reflection-Fourier Transform Infra-Red spectroscopy; EDS: energy dispersive spectroscopy; FTIR: Fourier Transform Infra-Red spectroscopy; LC-MS: Liquid Chromatography–Mass Spectrometry; ICP-MS: Inductively Coupled Plasma–Mass Spectrometry; ICP–OES: Inductively Coupled Plasma–Optical Emission Spectrometry; NMR: ^{1}H nuclear magnetic resonance spectroscopy; UPLC-ESI-MS: ultra-performance liquid chromatography coupled with tandem mass spectrometry, interfaced with an electrospray ionization source, operated in negative mode; TD: thermal desorption; SLE: supported liquid extraction; TXRF: total reflection X-ray fluorescence spectrometry.

Source	Plastic polymer			Leachate analysis	
	Product type	Polymer analysis	Chemical content analysis	Screened products	Analysis
Weis et al. (1992)	100 % curbside tailing[a]	–	–	Organic additives	GC–MS
	70 % curbside tailing and 30 % PS	–	–	Organic additives	GC–MS
Bejgarn et al. (2015)	New consumer plastic items (PE, PP, PVC, PS, PET, PUR)	–	–	–	–
Nobre et al. (2015)	Virgin PE granules	–	–	–	–
	Unidentified stranded pellets	–	–	–	–
Gandara e Silva et al. (2016)	Virgin PP pellets	–	–	–	–
	Stranded pellets (42 % PE and 58 % various polymers)	FTIR	–	–	–
Li et al. (2016)	Virgin PC, HDPE, LDPE, PET, PP, PS, PVC	–	–	Organic additives	HPLC-ESI
Martínez-Gómez et al. (2017)	Virgin PS microspheres and HDPE fluff	–	–	–	–
Seuront (2018)	Virgin PP pellets	–	–	–	–
	Stranded pellets	–	–	–	–
Ke et al. (2019)	New single-use plastic bags	FTIR	GC–MS/EDS	–	–
Oliviero et al. (2019)	Virgin PVC microplastics	–	–	Metals	ICP-MS
	Green toy	FTIR	–	Metals	ICP-MS
	Blue toy	FTIR	–	Metals	ICP-MS
	Orange toy	FTIR	–	Metals	ICP-MS
Tetu et al. (2019)	Grey grocery bag (HDPE)	–	–	Organic additives	LC-MS
	Textured black and yellow/green plastic matting (PVC)	–	–	Metals	ICP-MS/ICP-OES
				Organic additives	LC-MS
				Metals	ICP-MS/ICP-OES
Aminot et al. (2020)	Beached foam-like plastic macrodebris	FTIR	–	Brominated flame retardants, Plasticizer bisphenol A	UPLC-ESI-MS
Capolupo et al. (2020)	Virgin PET, PS, PP, PVC, CTR	–	–	Organic additives	GC–MS
				Metals	ICP-MS
Chae et al. (2020)	Virgin expanded polystyrene fragments and spheres	–	–	Chemical and carbon content	HPLC
Cormier et al. (2021)	Stranded microplastics	FTIR	GC + ICP-MS	–	–
Gardon et al. (2020)	Spat collectors (shade-mesh), new and aged	FTIR	TD/GC–MS + SLE/GC–MS	PAHs, organic additives	TD/GC–MS
	Synthetic ropes, new and aged	FTIR	TD/GC–MS + SLE/GC–MS	PAHs, organic additives	TD/GC–MS
Langlet et al. (2020)	Virgin PP pellets	–	–	–	–
Piccardo et al. (2020)	Virgin PET microparticles	FTIR	–	–	–
Sarker et al. (2020)	Grey grocery bag (HDPE), new and aged	–	–	Metals	ICP-MS/ICP-OES
	Textured black plastic matting (PVC), new and aged	–	–	Metals	ICP-MS/ICP-OES
Trestrail et al. (2020)	Oasis® Ideal floral foam	ATR-FTIR	–	–	–
	Oasis® Enhanced Biofoam	ATR-FTIR	–	–	–
Thomas et al. (2020)	Virgin PS and PMMA microparticles	–	–	–	–
Capolupo et al. (2021)	Virgin PET, PS, PP, PVC, CTR	–	–	Organic additives	GC–MS
				Metals	ICP-MS
Gewert et al. (2021)	Virgin LDPE, PP, PS and PET pellets	–	–	–	–
Koski et al. (2021)	Tire wear particles from: new tire, old tires, rubber granules	–	–	Metals	ICP-MS
Lehtiniemi et al. (2021)	Virgin LDPE granules	–	NMR	Metals	TXRF
	Post-consumer LDPE frozen vegetable package	–	NMR	Metals	TXRF
	Recycled styrene-butadiene rubber	–	NMR	Metals	TXRF
Rendell-Bhatti et al. (2021)	Beached pellets	FTIR	–	PCBs, HCB, PAHs	GC-ECD
	New White PVC prime plasticized pellets	FTIR	–	PCBs, HCB, PAHs	GC-ECD
	Virgin LDPE	FTIR	–	PCBs, HCB, PAHs	GC-ECD
Schiavo et al. (2021)	Virgin PP, PE and PS	–	–	Organic additives	GC–MS
				Metals	ICP-MS
Seuront et al. (2021)	Virgin PP pellets	–	–	–	–

[a] Curbsite tailing is ca. 80 % polyolefin (Polyethylene and polypropylene), with the remaining percentages made of poly(ethylene terephthalate), polystyrene, polyvinyl chloride, and other plastics (Phillips et al., 1989).

small size, high surface curvature and large surface area, which typically enhance any surface effect such as sorption/desorption dynamics (Mattsson et al., 2018). As such, their toxicity typically increases as their size decreases (Anbumani and Kakkar, 2018; Sjollema et al., 2016). A range of physical processes related to particle-algae cell wall interactions such as adsorption, complexation and agglomeration have been reported to hamper the integrity of phytoplankton cell walls (Shiu et al., 2020; Casabianca et al., 2020), hinder photosynthesis and result in an increase of the reactive oxygen species in algae cells (Bhattacharya et al., 2010; Nolte et al., 2017, Bellingeri et al., 2020). Furthermore, the secretion of extracellular polysaccharides to form algae clusters have been shown to polymerize and concentrate microplastic beads, hence potentially influencing microplastic vertical distribution and bioavailability (Long et al., 2015, 2017). Note, however, that the abovementioned effects of MPs and NPs on microalgae—as well as those reported on a range of invertebrates—have only been demonstrated in the laboratory, which typically used concentrations of plastic that are up to 100 g l^{-1} or 100,000 particles l^{-1} for plastic leachate studies; see Delaeter et al. (2022) for a review. Noticeably these concentrations are orders of magnitude higher than typical environmental concentrations, which are in the ng l^{-1} to μg l^{-1} range (Lenz et al., 2016) or 1 to 500 particles m^3 (Koski et al., 2021), and can even reach up to ca. 17 mg l^{-1} in the intertidal zone (Paul-Pont et al., 2018). Though the discrepancy between environmental and experimental plastic concentrations has previously been pointed out (Lenz et al., 2016; Haegerbaeumer et al., 2019), its resolution is still at the core of a sustainable and meaningful future of plastic leachate research and warrants the need for further work to assess the impact of ecologically relevant NPs and MPs concentrations in laboratory studies.

MPs and NPs also act as both sources and vectors of pollutants through respectively the intrinsic chemical additives (usually organic), used in their manufacture to improve their physical and chemical properties, and the variety of extrinsic chemical pollutants, adsorbed onto their surface while aging in the ocean (Bakir et al., 2014, 2016; Wang et al., 2021). As a result, they tend to accumulate more persistent pollutants than large debris (Law and Thompson, 2014), and have even been qualified as "*a sink for toxic chemicals*" (Engler, 2012).

4. Leachates as the invisible portion of the micro- and nanoplastic iceberg

Only 5 polymers—polyethylene (PE), polypropylene (PP), poly(vinyl chloride) (PVC), polystyrene (PS) and poly(ethylene terephthalate) (PET)—account for 90 % of the MPs and NPs found in the ocean (Andrady and Neal, 2009; Suaria et al., 2016). These polymers, however, include numerous—over 400 according to the European Chemical Agency initiative (see www.echa.europa.eu)—chemical additives (typically organic) used in their manufacture mainly to increase their durability. These low molecular weight additives (i.e., light and heat stabilizers, antioxidants, nucleating agents, UV stabilizers, pigment agents, antistatic molecules, flame retardants and plasticizers; see www.echa.europa.eu) are typically released in seawater from plastic polymers, as well as other chemical compounds adsorbed onto their surface such as hydrophobic organic contaminants (e.g., polycyclic aromatic hydrocarbons, PAHs, and polychlorinated biphenyls, PCBs) and heavy metals. Some plastic additives such as phthalates and brominated flame retardants can reach high concentrations in coastal waters (Hermabessiere et al., 2017; Sánchez-Avila et al., 2012) and substantially accumulate in marine organism tissues (Vered et al., 2019). For instance, 17 to 917 tons of phthalates are expected to be released annually in the oceans from leaching PVC, PS and PE (Suhrhoff and Scholz-Böttcher, 2016), and PE microplastics release endocrine disrupting chemicals (typically bisphenol A, bisphenol S, octylphenol, nonylphenol), with estrogen-like and anti-androgen effects inducing carcinogenesis and mutagenesis (Chen et al., 2019; Huang et al., 2021).

The chemicals associated to MPs and NPs can impact directly or indirectly marine life (Huang et al., 2021; Wang et al., 2021). Due to their minute size, MPs and NPs are ingested by a large range of organisms including filters feeders such as oysters and mussels (Van Cauwenberghe and Janssen, 2014; Li et al., 2018; Green, 2016; Catarino et al., 2018; Green et al., 2019; Abidli et al., 2019) and organisms as small as zooplankton; see Bai et al. (2021) for a review. Once ingested, the variety of chemical pollutants, either bounded to the polymer or adsorbed onto their surface are desorbed, cause adverse effects and accumulate into the tissues and organs (Hahladakis et al., 2018; Vered et al., 2019) with putative cascading effects through the food chain (Desforges et al., 2015). In this context, these chemicals can be thought as the Trojan Horse of plastic litter triggering toxic effects and responses in a wide range of marine organisms. NPs are probably the least known area of marine litter, though potentially the most hazardous (Koelmans et al., 2015) and we still barely have any idea of the quantity of these chemicals that may be lurking below the surface of our oceans (Fig. 3B). They are of particular concern as they are more likely to pass biological membranes and affect the functioning of cells including blood cells and photosynthesis (Koelmans et al., 2015), but also inhibit fertilization and embryogenesis and decrease larval performance and have intergenerational effects (Tallec et al., 2018, 2019). MPs and NPs, however, release intrinsic and extrinsic pollutants directly in seawater, which have also been shown to have various toxic effects; see e.g., Huang et al. (2021) for a review.

5. On the lack of interdisciplinarity in leachate research

A Web of Science search (accessed June 30, 2022) based on the words *plastic*, *pollution* and *leachate* in the subject area "*Marine Ecology*" returned 120 articles. Among these articles, only 27 (Table 1) explicitly assessed the effect of plastic leachates sensu Delaeter et al. (2022)—i.e., the desorption of molecules that are adsorbed onto the surface of a polymer and/or absorbed into the polymer matrix in the absence of plastic ingestion—from a range of virgin and aged plastic items on marine organisms ranging from microbes to large molluscs and crustaceans. Note that we deliberately excluded studies that were conducted on leachate solutions where plastic items were kept in suspension as they are implicitly potentially biased by the challenge to decipher the physical effects due to the presence of plastic particles from the chemical effects that can be exclusively related to the leachate, hence were not considered as dealing with leachate sensu *strictissimo*. The effects of plastic leachates were addressed through the consideration of various biological parameters including gene expression, physiology, behaviour, individual and population growth; see Delaeter et al. (2022) for a detailed review of the various effects (including the lack of effect) of various plastic leachates on invertebrate organisms ranging from microscopic organisms such as cyanobacteria and microalgae to large molluscs and crustaceans. As such, the observed effects of plastic leachates on marine invertebrates have not been rehearsed in the present work. Note, however, that the reported effect (or lack of effect) of plastic leachates are extremely difficult to synthesized as they are fundamentally organism-dependent, polymer-dependent and dose-dependent (Delaeter et al., 2022). Specifically, the effect of a given dose of a given polymer on a given organism may differ depending on the scrutinized physiological and biological process. For instance, the leachates of the same dose of polypropylene (PP) and polyethylene terephthalate (PET) respectively decrease and have no effect on the growth of the marine diatom *Skeletonema costatum* (Piccardo et al., 2020). Similarly, the leachates from virgin polypropylene pellets affect the motion and aggregation behaviour and byssal thread production of four species of intertidal mussels in species-specific ways (Seuront et al., 2021); see Delaeter et al. (2022), their Table S1 for a review of the effects (and lack of effects) of plastic leachates on marine microbes and invertebrates. Noticeably, 33 % of the 27 articles (i.e., 9) that addressed the issue of the effect of plastic leachates on marine microbes and invertebrates did not conduct any sort of screening analysis on either the nature of the plastic polymer considered, their chemical content or the nature of their leachates (Table 1). More specifically, nearly 50 % of these articles (i.e., 13) did not attempt to assess the nature of the so-called leachate (Table 1). Among the 18 papers assessing the type of the plastic polymer considered, its chemical content or the nature of its leachates, nine did not identify the polymer used in the assay. This may, however, reasonably be considered as a minor limitation especially when virgin polymer particles or single-use products of known origin and composition were used (Table 1). Only four papers run screening analyses of the chemical content of the plastic employed in the manipulative experiments, i.e., new single-use plastic bags (Ke et al., 2019), new and aged spat collectors and synthetic ropes used in in pearl-framing gear (Gardon et al., 2020), stranded microplastics (Cormier et al., 2021), and virgin low-density polyethylene, post-consumer LDPE frozen vegetable package and recycled styrene-butadiene rubber (Lehtiniemi et al., 2021). Finally, when leachates were screened (i.e., in 14 of the 27 papers returned by our literature survey), the screening essentially targeted either trace elements or organic additives, but barely both, tough exceptions exist (Tetu et al., 2019; Capolupo et al., 2020, 2021; Schiavo et al., 2021); see Table 1. For some, the screening was specifically directed towards specific chemical compounds such as brominated flame retardants, plasticizer bisphenol A or polycyclic aromatic hydrocarbons (PAHs; Table 1). There is now unequivocal evidence that exposure to most of the abovementioned chemicals is fundamentally hazardous—typical examples being heavy metals, phthalates, alkylphenols, PAHs and pesticides such as PCBs; see Balcioglu (2016), Honda and Suzuki (2020) and Zhang et al. (2021) for recent reviews—which further emphasizes the need to characterize the chemical composition of leachates.

The relative lack of chemical screening analyses reported here in studies addressing the effects of plastic leachates on marine organism—but the same reservation applies to the lack of identification of plastic polymers and polymer chemical screening—is potentially detrimental to a comprehensive and realistic understanding of the effect of plastic leachates on marine life, hindering the development of the field of research on plastic pollution. Specifically, our concern here is that comparing results obtained without chemical screening of leachates are at best questionable, as is any idiosyncratic false analogy epitomized by the idiomatic comparison of apples and oranges. The potential consequences of these limitations are discussed hereafter based on three papers reporting the effect of virgin and stranded pellets on the embryonic development of the sea urchin *Lytechinus variegatus* (Nobre et al., 2015) and the brown mussel *Perna perna* (Gandara e Silva et al., 2016) and the antipredator response of the intertidal gastropod *Littorina littorea* (Seuront, 2018). Note, however, that the criticisms implied by the following statements and analyses do not detract from the central point of their work.

Nobre et al. (2015) found that the leachate from PE pellets impair embryonic development in the sea urchin *L. variegatus*, in contrast to the leachate from unidentified stranded pellets. The authors subsequently legitimately emphasized the need to avoid direct loss of virgin plastic pellets into marine habitats during manufacturing and transport. In contrast, Gandara e Silva et al. (2016) observed a higher relative toxicity of leachate from beached pellets (42 % PE and 58 % made of other, unspecified polymers) in comparison to virgin PP pellets to mussel embryo. Finally, Seuront (2018) found that the behavioural vigilance and predator avoidance behaviour of *L. littorea* were impaired and inhibited when exposed to leachate from virgin PP pellets and stranded unidentified pellets (later identified as PE pellets following FTIR analysis; Seuront, unpublished data), respectively. The observed discrepancies from these three examples alone highlight how the toxic effects detected by naively comparing beached vs. virgin pellets may be potentially driven by much more complex undetermined differences between their leachate treatments. These include (i) the release of different additives by the virgin and weathered pellets which were made of a different polymer, (ii) the release of different quantities of additives depending on the weathering time, (iii) the pollutant's affinity to the plastic polymer and to the leaching, chemical cocktail (as opposed to a single-chemical solution), and (iv) the period of time that the plastic persists in the environment; see e.g., Teuten et al. (2009). Apart from a proper identification of polymer type and chemical content of stranded pellets, a chemical screening of the leachate solution would have contributed to clear and precise hypotheses and decipher any indistinct results by demixing apples and oranges.

6. A plea for an interdisciplinary approach to micro- and nanoplastic pollution

Even when scientific knowledge was in its very early infancy, Paracelsus stated in ca. 500 CE a noticeably still currently held, convenient and undistracting toxicological truth '*all things are poisons at the right dose*'. In this context, especially when referring to leachate studies, it appears critical to identify the so-called '*things*' we are dealing with as well as the related '*dose*'. The resolution of these specific issues is at the core of what we believe is the future of plastic pollution research. It is also timely given recent incentives to improve the quality and relevance of plastic-related research and to set-up new research priorities to inform policies aimed at reducing the harm of plastic pollution to biota, including:

(i) the increasing awareness of the scientific community on the need to shift our focus from one that solely describes levels and geographic location of plastic contaminations to one that identifies ecological effects and solutions of plastic contamination and predicts impacts to the projected increase of plastic litter (Borja and Elliott, 2019),
(ii) the acknowledged need to raise the publication bar for microplastic research through harmonized methods and quality assurance practices, and more incentive for journals, editors and reviewers to be more proactive in ensuring clear, repeatable methods of research articles, an absolute prerequisite to contribute to a constructive and factual discourse on plastic pollution (Provencher et al., 2020),
(iii) the growing demand from reviewers for the inclusion of polymer types to microplastic particles in environmental and biological samples, and plastic particle chemical characterization as a condition of final manuscript acceptance (Ivar do Sul, 2021), even though it may remain legitimate to consider that stating "*we found the amount Q of litter at location X*" is appropriate—hence worthy of publication—depending on the nature of the study and the scientific question.

As an example, the number of timely papers focusing on the potential environmental impact of face masks as a potential source of plastic pollution has been blooming following the beginning of the COVID-19 pandemic (Benson et al., 2021; Chowdhury et al., 2021; Schofield et al., 2021); yet none of them assessed the nature of the materials leaching from them in the environment. Noticeably, the same limitation applies to the body of research devoted to another major form of anthropogenic litter, cigarette butt leachate in terrestrial, freshwater and marine environments (Micevska et al., 2006; Slaughter et al., 2011; Booth et al., 2015; Gill et al., 2018; Caridi et al., 2020; Green et al., 2020). This knowledge gap was also glaring in the recent second and third yearly conference of the Research Group "Polymers and Oceans", respectively held online from February 8–11, 2021 and in-person from June 27–29, 2022. The objectives of this research group that includes 240 researchers from 56 laboratories were "*to develop new avenues of interdisciplinary research. To involve all the concerned scientific communities: chemists, physicists, biologists, ecologists, ecotoxicologists, oceanographers, economists and sociologists to support le development of multi-scale and transdisciplinary approaches*" (see https://www.gdr-po.cnrs.fr). Specifically, it appears that descriptive studies assessing the presence, quantification and future of plastic particles in various estuarine and marine systems are still well represented. There is also a clear dichotomy between the studies focusing on the ecological effects of plastics on various organisms (e.g. phytoplankton, zooplankton, bivalves and fish), and the development and improvement of screening methods to e.g. microplastic characterization through GT pyrolysis coupled with mass spectrometry, mass spectrometry quantification of small (25 μm–1 mm) microplastics or the characterization of additives using Gas Chromatography/Mass Spectrometry-Orbitrap. The former noticeably did not use the latter in their approach, and vice versa. We are well aware that assigning polymer types to single micro- and nanoparticles, screening the nature and quantity of the chemicals (i) bounded to virgin polymers and new consumer plastic products and adsorbed onto their surface once discarded in the marine environment, and (ii) released by these plastic items is a daunting task; this requires significant laboratory time, as well as expensive and specialized knowledge of the concepts and methods specific to polymer chemistry. The tools and techniques needed to qualify and quantify the chemicals leaching from plastic polymers are, however, now virtually available (particularly in western countries and/or in large research facilities) to any scientists working in a marine laboratory, hence the chemical screening needed to improve the relevance of the leachate solutions used in the type of papers reported in the present work is only at a stone's throw from becoming an indissociable component of any leachate-based biological assay. It is nevertheless stressed that the tools needed to screen the chemical composition of plastic polymers and their leachates are still relatively expansive, ranging from ca. $25,000 for a Fourier-transform infrared spectrometer (FTIR), $50,000 for a micro-Raman spectrometer and in the range $250,000–500,000 for a pyrolysis-gas chromatography–mass spectrometry package (Primpke et al., 2020), not to mention the relative lack of uniformity in the methods used to detect, enumerate and characterize microplastics (Löder and Gerdts, 2015). Small research groups and research groups based in many parts of Asia and Africa, where the issues related by plastic pollution are of growing concern (e.g. Matsuguma et al., 2017; Kapinga and Chung, 2020; Marnn et al., 2021; Akindele and Alimba, 2021), may not have the financial resources to purchase the needed tools. This situation may limit the research efforts by those who do not have access to the so-called 'standard' facilities, and ultimately be detrimental to the progress and evolution of the field of plastic pollution. This limitation may be overcome though the development of collaboration with other disciplines and/or laboratories mastering the needed tools, which (i) further strengthen our plea for more interdisciplinary in micro- and nanoplastic pollution research, and (ii) is very much in line with the recent claim to develop *both* collaboration and infrastructure in African research on micro- and nanoplastic pollution (Nel et al., 2021).

In this context, we claim—especially given the intrinsic interdisciplinary nature of microplastic research, and beyond the question of the pertinence of keeping publishing descriptive studies about the presence of plastics in virtually any location on Earth (see Borja and Elliott (2019) for an in-depth discussion on this specific issue)—that it is of utmost importance for microplastic research to bridge the gap between studies focusing on the biological and ecological effects of plastic pollution and characterizing and quantifying the intrinsic and extrinsic chemicals behind the observed so-called effects. We believe this is a legitimate plea for a few fundamental reasons. Firstly, plastics research is likely to play a pivotal role to the future of the Anthropocene, essentially due to the extreme persistence of plastic polymers in the environment, an intrinsic and unique property which is missing in most (if not all) anthropogenic particles (Andrady, 2015). Secondly, a significant amount of research has been centered around their persistence in the environment, and their subsequent impacts on marine health (Duis and Coors, 2016; Chatterjee and Sharma, 2019) noticeably through the interactions between various microplastic debris (Galloway et al., 2017) and between microplastics and persistent organic pollutants (Lohmann, 2017; Rodrigues et al., 2019). As a consequence, and as previously stressed in the context of both microplastic monitoring (Gago et al., 2018; Cutroneo et al., 2020) and the assessment of environmental and health impact of MPs (Coffin et al., 2021), standardized protocols and open collaboration between different fields are at the core of the future of microplastic leachate research.

Finally, because a founding principle behind any scientific approach is to use a common acknowledged nomenclature, assessing the effect of plastic leachates on a specific organism without chemically screening these leachates does not hold more wisdom than assessing the flavor of an apple by tasting an orange. As such, studies focusing on the biological and ecological effects of plastic leachates can only be comparable — and ultimately be credible and survive the test of time — if they are based on a sound assessment of the chemicals present within and released from plastic polymers to form the so-called leachate solutions in the form of the following '5K' roadmap:

(i) *Know your plastic particle*. The characterization of the size, shape, surface charge and morphology plastic particles is fundamental as these properties significantly drive key processes such as the dynamics of adsorption/desorption and absorption/desorption through e.g. their surface/volume ratio. These properties are typically characterized through a combination of light scattering and optical techniques. The former are typically used to characterize physical properties such as size and particle size distribution; see Xu (2015). In turn, the latter provide information on the morphology of a particle through three groups of particle imaging techniques, such as optical microscopy, electron microscopy and scanning probe microscopy; see Zhou and Wang (2007), Michler (2008) and Schmid et al. (2013) for reviews;

(ii) *Know your plastic polymer*. The identification of the polymer requires the use of spectroscopic methods. When MP particles are larger than 500 μm, analyses are typically conducted on a particle-by-particle basis with attenuated total reflection Fourier transform-infrared spectroscopy (ATR-FTIR). In contrast, smaller particles are collected on filters and identified with focal plane array micro-FT-IR (FPA-FT-IR) or Raman microspectroscopy with particles down to 10 μm to 1 μm (Schwaferts et al., 2019);

(iii) *Know the chemical content of your polymer*. This includes the molecules that are either absorbed into the polymer matrix such as the seemingly ever-increasing family of additives that enter the composition of plastics when manufactured (e.g. light and heat stabilizers, antioxidants, nucleating and antistatic agents, flame retardants, plasticizers and colorants) and adsorbed onto the surface of the polymer post-manufacture such as heavy metals, Polychlorinated biphenyls (PCBs), Polycyclic aromatic hydrocarbon (PAHs), Dichlorodiphenyltrichloroethane (DDT), and Polybrominated diphenyl ethers (PBDE). These chemical compounds can now routinely be assessed through a range of relatively well-established gas chromatographic-mass spectrometric methods; see e.g. Schwaferts et al. (2019) for a review;

(iv) *Know the chemical content of your leachate*. The chemical screening of leachate solutions, though still noticeably lacking in ca. 50 % of the studies identified in our literature survey (see Table 1), is nevertheless accessible through a range of coupled chromatographic and spectrometric methods such as gas chromatography–mass spectrometry, liquid chromatography-mass-spectrometry, and inductively coupled plasma–mass spectrometry and inductively coupled plasma–optical emission spectrometry; see e.g. Tetu et al. (2019), Capolupo et al. (2021), Koski et al. (2021), Lehtiniemi et al. (2021) and Rendell-Bhatti et al. (2021) for recent examples;

(v) *Know your dose*. This is probably the most challenging component of the present roadmap as the relevance of any biological assay relies on the use of environmentally realistic amounts of particles and/or chemicals (Lenz et al., 2016; Delaeter et al., 2022), which are typically highly variable in time and space.

Note, however, that despite the range of analytical methods available to characterize the chemical content of both plastic particles and their leachates, there is no a priori single protocol but a multitude of approaches that covers a complex field, requiring specific methods for samples of different types (e.g. virgin pellets vs. weathered pellets) and sizes (e.g. microplastic vs. nanoplastic particles). In addition, the type of analytical methods to be used is also fundamentally driven by the scientific question to be addressed (not to mention the chemical compound of interest), which warrants the need for increased interdisciplinary research efforts. Based on the examples of the evolution of ecotoxicology towards a fully interdisciplinary research area (e.g., Jorgensen, 2010; Zhou et al., 2019), and how the collaborations between two fields as different as archaeology and marine biology have significantly improved our understanding of past, present and future marine (especially coastal) ecosystems—see e.g. Crumley (2021), McCormack et al. (2021) and Slade et al. (2021)—it is stressed that our journey to understand the effect of micro- and nanoplastic pollution on life in the ocean has to evolve through the development of a sound and systematic ability to chemically define what we biologically compare.

CRediT authorship contribution statement

Laurent Seuront: Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing – original draft, Visualization, Supervision, Funding acquisition. **Gerardo I. Zardi:** Conceptualization, Writing – original draft. **Marine Uguen:** Conceptualization, Writing – review & editing. **Vincent M.P. Bouchet:** Writing – review & editing. **Camille Delaeter:** Writing – review & editing. **Solène Henry:** Writing – review & editing. **Nicolas Spilmont:** Writing – review & editing. **Katy R. Nicastro:** Conceptualization, Writing – original draft.

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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