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File no. T1022662

The Honourable Jonathan Wilkinson
Minister of the Environment and Climate Change

c/o Mr. Thomas Kruidenier,
Executive Director
Program Development and Engagement Division
Department of the Environment
Gatineau, Quebec K1A 0H3

Dear Minister Wilkinson:

Re: Notice of Objection and Request for Board of Review In relation to the Proposed Order to Add "Plastic Manufactured Items" to Schedule 1 of the *Canadian Environmental Protection Act, 1999;* Canada Gazette Part I, Vol. 154, No. 41 – October 10, 2020

We are writing on behalf of Responsible Plastic Use Coalition, Dow Chemical Canada ULC, Imperial Oil, a partnership, by its managing partner Imperial Oil Limited, and NOVA Chemicals Corporation. This submission responds to the October 10, 2020 Gazette Notice (the "**Notice**") in which the Governor in Council, on the joint recommendation of the Minister of the Environment and the Minister of Health (the "**Ministers**") proposed an Order to add "Plastic Manufactured Items" to Schedule 1 of the *Canadian Environmental Protection Act, 1999,* ("**CEPA 1999**"), (hereinafter referred to as the "**Proposed Order**").[1]

As provided for by section 332(2) of CEPA 1999, we are filing this Notice of Objection and respectfully request that a Board of Review be established, pursuant to section 333 of CEPA 1999, to "inquire into the nature and extent of danger" posed by "Plastic Manufactured Items", for the reasons set out below.

1) "Plastic Manufactured Items" are not a "Substance" (CEPA, Sections 3(1) & 90(1))

The term "Plastic Manufactured Items" does not meet the definition of a "Substance", set out in subparagraph 3(1)(f) of CEPA 1999. Accordingly, the Governor in Council does not have the authority to add "Plastic Manufactured Items" to the List of Toxic Substances ("**Schedule 1**").

Subsection 90(1) of CEPA 1999 provides that the Governor in Council may make an order adding a substance to Schedule 1, on the recommendation of the Ministers, if satisfied that the substance is toxic:[2]

[1] Order Adding a Toxic Substance to Schedule 1 to the Canadian Environmental Protection Act, 1999, (2020), C Gaz I, 2733.
[2] *Canadian Environmental Protection Act, 1999*, c 33 [CEPA], s 90(1).

> **90 (1)** Subject to subsection (3), the Governor in Council may, if satisfied that a substance is toxic, on the recommendation of the Ministers, make an order adding the substance to the List of Toxic Substances in Schedule 1.

The enabling authority provided to the Governor in Council relates to "a substance" singular. If satisfied that "a substance" is toxic, the Governor in Council may make an order to add "the substance" to Schedule 1.

Substances must be assessed individually, and added to Schedule 1 one at a time. The condition precedent to adding a substance to Schedule 1 is a determination by the Ministers that the substance in question is toxic. As such, the substance proposed for addition to Schedule 1 must be identified with sufficient precision that it is capable of assessment for toxicity.

Like the enabling authority in section 90(1), the definition of "substance" in CEPA 1999 also speaks in singular terms in relation to "manufactured item". A substance may be "any matter that is capable of being dispersed in the environment", "any element or free radical", "any combination of different molecules that originate in nature or are the result of chemical reactions", "any mixture that is a combination of substances", "any animate matter", or "any manufactured item" ("**Substance**").[3]

While it is possible for a manufactured item to be considered a Substance, this is only true for manufactured items considered one at a time:

> ***"substance*** means any distinguishable kind of organic or inorganic matter, whether animate or inanimate, and includes
>
> …..**(f) any manufactured item** that is formed into a **specific physical shape** or design during manufacture and has, for its final use, a function or functions dependent in whole or in part on its shape or design…"[4]

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[3] See generally CEPA, *ibid*, s 3(1), "***substance*** means any distinguishable kind of organic or inorganic matter, whether animate or inanimate, and includes
(a) any matter that is capable of being dispersed in the environment or of being transformed in the environment into matter that is capable of being so dispersed or that is capable of causing such transformations in the environment,
(b) any element or free radical,
(c) any combination of elements of a particular molecular identity that occurs in nature or as a result of a chemical reaction, and
(d) complex combinations of different molecules that originate in nature or are the result of chemical reactions but that could not practicably be formed by simply combining individual constituents,
and, except for the purposes of sections 66, 80 to 89 and 104 to 115, includes
(e) any mixture that is a combination of substances and does not itself produce a substance that is different from the substances that were combined,
(f) any manufactured item that is formed into a specific physical shape or design during manufacture and has, for its final use, a function or functions dependent in whole or in part on its shape or design, and
(g) any animate matter that is, or any complex mixtures of different molecules that are, contained in effluents, emissions or wastes that result from any work, undertaking or activity."

[4] CEPA, *ibid*.

In this case, the Proposed Order does not target a single chemical, nor even a single manufactured item. Instead, it proposes listing "Plastic Manufactured Items", which is a category containing thousands or even tens of thousands of individual products; including products that are fundamentally important in manufacturing, health care, automotive and renewable energy sectors. The category would include every product manufactured from plastic in Canada: from a child's action figures, to the structural parts of our cars, homes and offices, to the plexiglass shields being used to protect grocery store clerks from COVID-19, and the containers that we use to carry our lunch to work.

As a practical matter, by identifying a category containing thousands of substances, the Proposed Order does not identify "the substance" proposed for listing with sufficient precision to enable an assessment for toxicity. As such, the condition precedent to listing is not met.

More fundamentally, the term "Plastic Manufactured Items" does not satisfy the definition of "Substance" set out in subparagraph 3(1)(f), because it proposes to list a category containing thousands of manufactured items, rather than a single manufactured item, as required.

As such, the Proposed Order fails to propose a Substance for listing on Schedule 1, and is *ultra vires* the enabling authority set out in section 90(1).

2) "Plastic Manufactured Items" Are Not Toxic (Section 64)

Before a substance may be added to Schedule 1, it must not only be a Substance as defined in subsection 3(1), it must also meet the criteria for toxicity, pursuant to the criteria section 64 of CEPA 1999. A Substance is toxic pursuant to section 64 if it meets the following definition:[5]

> **64** …. **except where the expression "inherently toxic" appears**, a substance is toxic if it is entering or may enter the environment in a **quantity or concentration or under conditions that**
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> **(a)** have or may have an immediate or long-term **harmful effect on the environment** or its biological diversity;
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> **(b)** constitute or may constitute a **danger to the environment** on which life depends; or
>
> **(c)** constitute or may constitute a **danger in Canada to human life or health**. (section 64, CEPA 1999, emphasis added)

Simply put, section 64 requires harm, danger, or risk of harm or danger, either to human health or to the environment. It also requires a nexus between the harm, danger or risk posed by a Substance, and the concentration, quantity or conditions under which a Substance is entering the environment. Each of these criteria will be considered, in turn, below.

[5] CEPA, *ibid*, s 64.

The Proposed Order rests on a literature review, titled the Science Assessment of Plastic Pollution (the "**Literature Review")**.[6] The Literature Review does not actually consider the impact of "Plastic Manufactured Items" at all. Instead, it considers the existing literature regarding the impacts of pollution from "microplastics" and "macroplastics", which are two distinct size categories of plastic pollution.

Even in relation to microplastics and macroplastics, however, the Literature Review does not satisfy the criteria for toxicity set out in section 64.

First, the Literature Review does not identify any risk to human life or health, from microplastics or macroplastics.[7] Accordingly, section 64(c) of the definition of toxicity is not engaged. In relation to potential or actual risks to the environment under section 64(a) and (b), the Literature Review finds the evidence regarding microplastics to be "unclear" and "contradictory.[8]

Only in relation to macroplastics did the Literature Review identify actual or potential harm to the environment.[9] In particular, the Literature Review identified two potential harms: the risk of biota becoming entangled or smothered, and the risk of biota ingesting macroplastics.

Even in relation to these risks, it is not clear that section 64 criteria are satisfied. While a range of macroplastic items can potentially pose a risk of entanglement, in the overwhelming majority of cases (as high as 83% in the studies cited by the Literature Review), the cause of entanglement was fishing gear, in particular fishing ropes and nets.[10] While the Literature Review focused on macroplastics, the harm identified turns on the form of the item, and not on the material it was made from.

The Literature Review points to "ghost fishing", which occurs when fishing nets and ropes are lost at sea, creating a risk of entanglement for sea life as they drift. The risk of harm from ghost fishing flows not from the material used to make the fishing nets, but from the fact that they are drifting, uncontrolled. A ghost fishing net made of natural fiber poses the same risk of entanglement to sea life as a net made of plastic. The Substance that should be assessed, in that case, to determine if it is "toxic" under section 64, is the fishing net, not macroplastics.

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[6] Canada, Environment and Climate Change Canada ("**ECCC**") and Health Canada, *Science Assessment of Plastic Pollution*, (Ottawa, Minister of the Environment and Climate Change, October 2020) [Literature Review].

[7] Literature Review, *ibid* at 82, concluded that, based on the current scientific literature: "a concern for human health has not been identified…"

[8] Literature Review, *ibid* at 82. In particular: "…although there are reports indicating that exposure of environmental receptors to microplastics can lead to mortality, developmental and reproductive effects, effects on feeding and energy production, and biochemical or molecular-level effects**, a similar number of reports have found no effects**".

[9] Literature Review, *ibid* at 82. Only in relation to macroplastics did the Literature Review conclude that there was demonstrated evidence of "physical harm to environmental receptors on an individual level", along with the "potential to affect habitat diversity." In particular, organisms "have been shown to ingest macroplastics and to become entangled in macroplastics, which can result in direct harm and in many cases, mortality".

[10] Literature Review, *ibid* at 51. A study of 265 bird species found that "fishing gear was determined to be the cause of entanglement in 83% of species", and in a 2015 study of 30,896 individuals and 243 species, of the 79% of incidents linked to direct harm or mortality, the "majority of these incidents involved plastic rope and netting".

Risks related to ingestion or smothering by macroplastics were anecdotally connected to a wide variety of items, ranging from plastics bowls and bottle caps, to plastic straws, bags, and latex balloons.[11] However, as with fishing ropes and nets, there was no suggestion that these macroplastics items are inherently toxic. As such, under section 64(a) or (b), these macroplastic items must be shown to pose a risk to the environment due to the "quantity, concentration, or conditions under which" they are entering the environment, in order to be considered toxic.[12]

The Literature Review observed that the reports regarding macroplastic occurrence in Canada were often "limited to data from litter clean up initiatives as well as reports in the popular press".[13] It also frankly acknowledged that due to a lack of "standardized methods and analytical techniques", it was simply "**not possible to quantitatively characterize environmental or human exposure levels at this time**".[14]

The definition of toxicity is not met by proving that a single bottle cap, if littered, poses a risk to biota that might ingest it. Litter can be problematic, and not meet the threshold of being "CEPA toxic". Section 64 specifically requires that the identified risk of harm from a Substance be connected to the quantity, concentration or conditions under which the Substance is entering, or may enter, the environment.

A Substance for which exposure levels have not even been quantified – in the case of the Literature Review, a handful of macroplastic items - cannot demonstrate a risk of harm due to the "quantity, concentration or conditions under which" they are each entering the environment in Canada.

More fundamentally, there must be a legal nexus between the alleged harm, as set out in the Literature Review, and the Substance proposed for listing on Schedule 1. No such nexus exists for "Plastic Manufactured Items".

As set out above, the potential harm identified in the Literature Review relates to a handful of specific macroplastic items. However, the Proposed Order does not propose to list these specific macroplastic items, or all macroplastics. Instead, it proposes to list a category ("Plastic Manufactured Items"), which would contain every product manufactured from plastic in Canada.

Accordingly, the Literature Review identifies a potential harm for a Substance that is not proposed for listing, and the Substance proposed for listing ("Plastic Manufactured Items") is not the Substance for which a risk of harm to the environment has been identified.

The Literature Review did not study, review, or reach any conclusions in relation to "Plastic Manufactured Items", nor did the Literature Review link "Plastic Manufactured Items" to the handful of specific macroplastic wastes identified as posing a risk.

Therefore, "Plastic Manufactured Items" do not satisfy the criteria for toxicity set out in section 64, and cannot be listed on Schedule 1.

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[11] Literature Review, *ibid* at 51.
[12] CEPA, *supra* note 2, s 64: "…. except where the expression "inherently toxic" appears, a substance is toxic if it is entering or may enter the environment in a quantity or concentration or under conditions that…"
[13] Literature Review, *supra* note 6 at 32.
[14] Literature Review, *ibid* at 32.

3) The Proposed Order is Based on a False Premise (the 1% "Leakage" Assertion)

As set out above, "Plastic Manufactured Items" do not constitute a Substance under section 3(1)(f), and the term does not meet the definition of toxicity in section 64. As such, the Governor in Council does not have the enabling authority under subsection 90(1) to list "Plastic Manufactured Items" on Schedule 1. Given these fundamental legal concerns, it is worth stepping back to consider the regulatory issue being targeted. What problem is the Proposed Order trying to solve, by labelling every item manufactured from plastic in Canada as "toxic"?

Both the Literature Assessment and the Regulatory Impact Assessment Statement ("**RIAS**") for the Proposed Order make clear that the problem being targeted is plastic litter entering the Canadian environment. Both documents assert that 1% of the plastic waste generated annually in Canada – an estimated 29,000 tons or 29 kilotons ("**KT**"), in 2016 - ends up in the environment as litter, every year.[15]

Recall, as discussed above, that unless a Substance is inherently toxic, it is not "toxic" under section 64 unless there is both a risk of harm, and a link between that risk and the quantity, concentration, or conditions under which it is entering the environment. In relation to the Proposed Order, the ubiquity of plastics is not sufficient, nor is risk posed by particular macroplastics sufficient. In order to satisfy section 64, the Proposed Order relies on the assertion that 1% of the plastic waste generated annually, or 29 kt in 2016, ends up as litter in the environment.

This 1% figure is the basis for the Proposed Order that would list "Plastic Manufactured Items" on Schedule 1. It drives the entire policy exercise. However the 1% figure is not rooted in fact.

The 1% figure is cited to a report prepared for Environment and Climate Change Canada ("**ECCC**") by Deloitte, titled *Economic Study of the Canadian Plastic Industry, Market and Waste: Summary Report to Environment and Climate Change Canada*.[16] The full Economic Study has not been published by ECCC, but a Summary Report has been, and is available online (the "**Economic Study**").[17]

The Economic Study outlines a model of plastic waste management in Canada, using 2016 as a baseline year. This model, prepared by Deloitte, **estimates** that 1% of the plastic waste that was generated in Canada in 2016 was lost to the environment, and labels this loss as "leakage" (see Figure 5, Figure 19, Table 23 and the definition for "LEAK").[18]

However, Deloitte did not actually measure how much of the plastic waste generated in Canada in 2016 ended up as litter, nor does it rely on any study that did so. Instead, the amount of plastic waste that

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[15] Literature Review, *ibid* at 12; Order Adding a Toxic Substance to Schedule 1 to the Canadian Environmental Protection Act, 1999, (2020), C Gaz I, 2741-42 (Regulatory Impact Analysis Statement) [RIAS]. In particular, of the 4,667 kt of plastic wastes that entered the Canadian market in 2016, 3,268 kt were discarded as waste and: "an estimated 29 kt (or 1%) were discarded outside of the normal waste stream (i.e., not landfilled, recycled or incinerated) in 2016, through direct release to the environment or through dumps or leaks."

[16] Canada, Environment and Climate Change Canada, *Economic Study of the Canadian Plastic Industry, Markets and Wastes, Summary Report* (Ottawa: Government of Canada, 2019), as cited in the Literature Review, *supra* note 6 at 12 [Summary Report].

[17] The Economic Study itself is not a published document, but the Government of Canada has published a Summary Report.

[18] Summary Report, *ibid*, Fig 5 at 3, Fig 19 at 36, definition of "LEAK" in Table 23 at 38.

"leaked" into the environment in 2016 was estimated by Deloitte: "This second fraction, also called plastics leaked into the environment (LEAK) is **estimated** in the model" (emphasis added).[19]

In Table 23, where the term "LEAK" is defined, the Deloitte study cites a single paper in relation to the 1% estimate, authored by Jambeck et al:

> "Global **estimates** of plastic leakage into the environment were prepared by Jambeck et al. in 2015. In this study, the authors estimated that approximately 10,000 tonnes of plastic waste were mismanaged in coastal areas and nearly 29,000 tonnes across Canada".[20]

Both the Literature Review and the RIAS estimate that 1% of the plastic waste produced in Canada each year becomes litter. They do so based on Deloitte's Economic study, which also estimates a "leakage" rate of 1% of the plastic waste generated annually. However, the Jambeck et al study, cited by Deloitte, does not provide actual data for Canada, for 2016 or otherwise.

Like the Deloitte paper, the Jambeck paper is not a quantitative assessment of plastic waste. Instead, it is a paper describing an engineering model, to be used for prediction and management. The authors note that although the presence of plastic debris in the oceans is widely documented, "the "quantity of plastic entering the ocean from waste generated on land is unknown".[21]

In order to help model the impact of different regulatory scenarios, and project forward in time, the Jambeck model uses data on solid waste generation, population density, and economic status to estimate the quantity of plastic waste entering the ocean from land: "We **estimated** the annual input of plastic to the ocean from waste generated by coastal populations worldwide".[22]

Jambeck et al considered 192 countries, and the top 20 countries were estimated to be responsible for 83% of the annual total of "mismanaged" plastic waste in 2010.[23] **Canada was not among the offenders**.

Looking behind the Jambeck article at the Supplementary Materials for the article and the spreadsheet setting out Jambeck's detailed calculations, Canada was actually not attributed any inadequately managed waste at all, but was estimated to have a 2% rate of littering.[24] Once again, however, this littering rate was also estimated, not measured.

Jambeck's Supplementary Materials make clear that Jambeck's paper, in turn, cited a single study of littering, undertaken in relation to litter on roadways in the U.S, in 2008:

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[19] Summary Report, *ibid* at 38.
[20] Summary Report, *ibid* at 38.
[21] Jambeck et al, "Plastic waste inputs from land into the ocean" (2015) 347:6223 Science at 768–771 [Jambeck et al], see headnote.
[22] Jambeck et al, *ibid* at 770. Please note that while the Economic Study (Summary Report) quotes and references Jambeck, it does not provide a proper citation. Instead, it cites the unpublished, full-length version of Deloitte's Economic Study for ECCC, in which Jambeck et al is referenced (Deloitte. (2019a); Summary Report, *supra* note 16.
[23] Jambeck et al, *ibid* at 769.
[24] Jambeck et al, *ibid*, *Supplementary Materials* [Jambeck Supplementary Materials], consisting of a pdf file and "Data S1" (Excel file), available at www.sciencemag.org/content/347/6223/768/suppl/DC1.

> "We estimated percentage of waste littered using the **only available national estimate of litter mass** (25), which reported 4.17 million MT of litter generated in the United States in 2008, equivalent to approximately 2% of national waste generation. **For each country we estimated 2% of the total waste generated is littered**…" (emphasis added).[25][26]

In summary, the factual premise underlying the Proposed Order is that 1% of the plastic waste generated in Canada "leaks" into the environment. However, this factual premise is completely unsupported. It is not based on any measurement or quantitative data from Canada at all. Instead, it is an estimate, based on an estimate, **based on an estimate** – based on a single study, carried out in the U.S., in 2008.

This factual matrix cannot possibly provide a reasonable basis for the Governor in Council to conclude, in 2020, that 1% of the plastic waste generated in Canada annually "leaks" into the environment.

Section 64's criteria for toxicity, which require a risk of harm or danger to the environment by virtue of the quantity, concentration or conditions under which a Substance is entering or may enter the environment, in Canada, cannot be satisfied by an estimate. Nor can it be satisfied by a single, 12-year-old study, of conditions in a different country.

Likewise, subsection 90(1) requires the Governor in Council to be "satisfied" that a Substance is toxic. In order to do so, the Ministers, and the Governor in Council, must be able to point to occurrence data for the Substance in question. That occurrence data must be for the Canadian environment. It cannot be sufficient for the Governors in Council to guess that a problem may exist, based on an estimate three times removed, or based on conditions in another country.

For the reasons set out above, the Proposed Order is *ultra vires* both section 64 and subsection 90(1).

Request for Board of Review

As provided for by section 332(2) of CEPA, Responsible Plastic Use Coalition, Dow Chemical Canada ULC, Imperial Oil, a partnership, by its managing partner Imperial Oil Limited, and NOVA Chemicals Corporation respectfully request that a Board of Review be established, pursuant to section 333 of CEPA, to "inquire into the nature and extent of danger" posed by "Plastic Manufactured Items".

As set out above, the Proposed Order does not satisfy the definition of "Substance", or the criteria for toxicity. The reason that it fails to meet these statutory requirements is because the regulatory process leading up to the Proposed Order was a marked departure from the norm under CEPA 1999. In

[25] Jambeck Supplementary Materials, *ibid* at 3.

[26] As an aside, it should be noted, that Deloitte's figures do not actually line up with Jambeck's. Jambeck estimated a total of just 7,959 tonnes of mismanaged plastic waste in Canada in 2010. Even when Jambeck projected forward ten years, to 2025, the authors estimated less than 15,000 tonnes of plastic litter nationally. Deloitte does not make clear in the Summary Report where the figure of 29,000 tonnes comes from, nor do ECCC's Literature Review, or the Regulatory Impact Assessment Statement for the Proposed Order. Clearly, however, the figure being used to support the Proposed Order is not actually a figure that was produced by Jambeck. On the contrary, the estimates used by Deloitte exceed Jambeck's own estimates for Canada by a very significant margin.

particular, the Proposed Order is not supported by a substance risk assessment, or for that matter, by science, and weight of evidence analysis.

The American Chemistry Council noted, when commenting on the draft Literature Review, that it:

> "does not individually assess each 'plastic', either with respect to the specific polymer relevant to that plastic or the relevant and specific additives; each plastic as used in packing; or each plastic as used in a particular product.
>
> The Draft also does not adequately present specific findings that take into account use, exposure, and environmental fate specific to each plastic, plastic packaging, and resin. It does not support substance-specific findings related to the entry of the substance into the environment in a quantity or concentration or under conditions that justify further action."[27]

A Board of Review would provide an opportunity to remedy the above shortcomings. A targeted and more robust review would place science and risk assessment front and centre, to examine the scientific data available, including the quality and uncertainty associated with the data, and to determine, in a credible manner, if "Plastic Manufactured Items" in fact pose a risk of harm that satisfies the criteria set out in section 64.

1) The Importance of Science

One of CEPA's "Guiding Principles" is that CEPA 1999 "emphasizes the integral role of science" in decision-making.[28] The entire statutory scheme of Part 5 of CEPA 1999, Controlling Toxic Substances, is built on sound, scientific assessment of risk in relation to the management of toxic substances.

ECCC's own guide to CEPA 1999, which has been in place since 2004, underscores that risk assessment is the "**prelude to**, **and informs, the risk management stage for all programs**" under CEPA 1999.[29] As noted above, however, no risk assessment was undertaken for "Plastic Manufactured Items" to support the Proposed Order.

Risk assessments are carried out under Part 5 of CEPA 1999 in relation to "Existing Substances", those which are on the Domestic Substances List and subject to a screening level assessment or identified for inclusion on the Priority Substances List prior to assessment, and for "New Substances", which are assessed pursuant to the *New Substance Notification Regulations*. Regardless of which track a substance is on, ECCC advises that substance risk assessments must be based on "sound science", and incorporate the "weight of evidence" approach (discussed further below):

> "Risk Assessment – Substance risk assessments are based on sound science, which supports a better understanding of their impacts and exposure to the environment and human health. The assessments incorporate the precautionary principle and a weight of

[27] Letter from Chris Jahn, President and CEO of the American Chemistry Council to the Executive Director of Program Development and Engagement Divisions, Department of the Environment (31 March 2020), "American Chemistry Council Comments on Draft Science Assessment of Plastic Pollution" at 4-5.
[28] Canada, Environment and Climate Change Canada, *A Guide to Understanding the Canadian Environmental Protection Act, 1999* (Ottawa, Environment and Climate Change Canada, 2004) [CEPA Guide] at 3.
[29] CEPA Guide, *ibid* at 5.

> evidence approach. Risk assessment also helps to identify the sources of pollution that pose the greatest risk to the environment and human health…risk assessment is the prelude to, and informs, the risk management stage for all programs under CEPA 1999…"[30]

The Government of Canada makes clear in ECCC's online guidance related to CEPA 1999 that, in all cases, the determination as to whether a substance is toxic and should be added to Schedule 1 must either be based on a risk assessment, or on a finding that a substance is "CEPA-toxic equivalent", which must in turn be the result of a systematic, risk-based assessment:

> "Substances may also be added to the List of Toxic Substances in Schedule 1 of CEPA through section 90(1) of the act without having gone through a Priority Substances List assessment, a screening assessment, or the review of another jurisdiction's decision if, on the recommendation of the ministers of Environment and Health, the Governor in Council is satisfied that a substance is toxic. A substance is "CEPA-toxic equivalent" **if it satisfies the definition of "CEPA-toxic" as a result of a systematic, risk-based assessment.** Such assessments can include determinations made under other federal statutes, or can incorporate appropriate elements of assessments done by or for provinces or territories, international organizations or other appropriate scientific authorities" (emphasis added) [31]

The Government of Canada further advises that systematic, risk-based assessments found to be "CEPA-toxic-equivalent" could come out of international organizations, provinces, or "appropriate scientific authorities".[32] The examples provided in relation to subsection 90(1) are the Montreal Protocol on Substances that Deplete the Ozone Layer, in relation to the substance bromochloromethane, and the Stockholm Convention on Persistent Organic Pollutants, in relation to the substance DDT.

Both Conventions were implemented under the auspices of the United Nations Environment Program, and in both cases, rely on rigorous, science-based evaluation leading to multilateral action on global environmental issues of concern. Furthermore, it cannot be credibly asserted that bromochloromethane and DDT are in any way comparable to "Plastic Manufactured Items". In no case, in relation to Schedule 1, does ECCC assert that it is appropriate, or within the power of the Governor-in-Council pursuant to CEPA 1999, to find a substance "CEPA toxic", or "CEPA-toxic equivalent", without a proper, science based risk assessment. Yet this is precisely what the Proposed Order now purports to do in relation to "Plastic Manufactured Items".

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[30] CEPA Guide, *ibid* at 5.
[31] Government of Canada, "Risk assessments under section 90(1) of Canadian Environmental Protection Act, 1999" (accessed December 2020), online: Canada.ca < https://www.canada.ca/en/environment-climate-change/services/canadian-environmental-protection-act-registry/substances-list/risk-assessments-section-90-1.html > [https://web.archive.org/web/20201204020430/https://www.canada.ca/en/environment-climate-change/services/canadian-environmental-protection-act-registry/substances-list/risk-assessments-section-90-1.html].
[32] *Ibid*.

2) The Absence of Science, and Risk Assessment, in Support of the Proposed Order

As noted above, the Literature Review was not a risk assessment, and was "not intended as a substitute for" risk assessment. The Literature Review itself makes this point:

> "This report is a review of the current state of the science on plastic pollution. **It is not intended as a substitute for chemical risk assessment**…" [33]

ECCC acknowledges that it was unable to carry out a risk assessment in relation to "Plastic Manufactured Items" due to "significant data gaps". These data gaps included a "lack of standardized methods" for monitoring, and for "characterizing the environmental and health effects" of the substances under review". They also included "inconsistencies in reporting" in relation to both occurrence and effects data:

> Typically, a chemical risk assessment is conducted to assess the potential for risk to the environment and human health associated with a substance. However, **significant data gaps currently exist that preclude the ability to conduct a quantitative risk assessment**, including a **lack of standardized methods for monitoring** microplastics and **characterizing the environmental and human health effects** of plastic pollution, as well as inconsistencies in the reporting of occurrence and effects data in the scientific literature (Gouin et al. 2019).[34]

To give just one example, the Literature Review noted that studies on microplastics suffered from a "general lack of consistency and reliability" in relation to methods used to "sample and quantify" microplastics in the environment, and in media such as drinking water, or food. The Literature Review reported that visual identification was often used to determine if a particle was, in fact, plastic, which can lead to a "high false positive rate", as it "does not allow for proper characterization of plastics".

> "For instance, when fibres visually identified as microplastics from the GI tracts of eelpout (Zoarces viviparous) were analyzed…none of the fibre were determined to be of synthetic origin." [35]

In other words, the fibres that had been counted as 'microplastics' were not, in fact, plastic at all.

The American Chemistry Council wrote to ECCC in March 31, 2020, expressing concern about the failure to conduct a risk assessment, and noting that:

> "The Draft itself is not a risk assessment. For that matter, it is not a problem formulation or scope of a risk assessment…
>
> If a particular product, packaging or resin is evaluated, using best available science and weight of evidence, taking into consideration the quality of studies, and as a result, is deemed to present significant enough concern to warrant a risk assessment, then a robust risk assessment could proceed. It appears, however, that the proposed course of action

[33] Literature Review, *supra* note 6 at 14.
[34] Literature Review, *ibid* at 14.
[35] Literature Review, *ibid* at 75.

> is to skip over the risk assessment. This is unwise and wholly inconsistent with the provisions of CEPA"[36]

The Literature Review may not have been intended as a substitute for risk assessment, but that is precisely how it has been used in relation to the Proposed Order. This, notwithstanding the acknowledgement in the Literature Review that the very frameworks to be used to carry out risk assessment for plastics pollution are currently in the process of development:

> "risk assessment frameworks for evaluating the potential risks associated with plastics pollution are currently under development".[37]

The Governor in Council cannot reach a conclusion in relation to the potential risks of a substance for which no risk assessment has been carried out – and in relation to which the framework for undertaking such a risk assessment is still being developed.

3) The Precautionary Principle

The RIAS for the Proposed Order asserts that "Plastic Manufactured Items" are being added to Schedule 1 in accordance with the precautionary principle, in order to "address the potential ecological risks associated with certain manufactured items becoming plastic pollution".[38]

However, "Plastic Manufactured Items" are not, *ab initio,* plastic *waste.* All manufactured items have a life cycle, which can range from days to decades, between manufacture and final disposal, recycling or incineration. There is no analysis in the Literature Review regarding how, when, and under what conditions "Plastic Manufactured Items" enter and become part of the plastic waste stream, and from there, how they become plastic pollution posing a risk to the environment.

Even in relation to macroplastic *pollution*, only a handful of items were connected to any risk of harm in the Literature Review. ECCC offers no explanation of how the potential risk of harm from a handful of types of macroplastic *litter* can ground the assertion that all "Plastic Manufactured *Items*" pose a sufficient risk to the environment that they merit listing on Schedule 1. A manufactured item must first enter the waste stream, and then become litter, to even pose a potential risk to the environment.

Instead of analysis, all that is offered by ECCC is the assertion that 1% of the plastic waste generated annually in Canada becomes pollution. As set out above, this assertion is based on conjecture, unsupported by Canadian data, measurements or scientific study.

The precautionary principle cannot cure these failures. If it could, there would be no meaningful limits or bounds as to what could be characterized as "toxic" and added to CEPA Schedule I.

CEPA 1999 enshrines the precautionary principle in the preamble:

> "Whereas the Government of Canada is committed to implementing the precautionary principle that, **where there are threats of serious or irreversible damage**, lack of full

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[36] *Supra,* note 27.
[37] Literature Review, *supra* note 6 at 14.
[38] See RIAS, *supra* note 15.

> scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation" (CEPA 1999, emphasis added)

However, the precautionary principle only applies when there are "threats of serious or irreversible damage". In such cases, a "lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation".

The precautionary principle cannot be used*, in place of evidence,* to assert that a problem exists. Rather, the precautionary principle can be used to justify risk management measures, in relation to a problem that has been demonstrated to exist, but only where there are "threats of serious or irreversible damage".

ECCC's own policies on the application of precaution underscore that the evaluation of sound, scientific information must be the basis for applying the precautionary principle:

> "It is particularly relevant that **sound scientific information and its evaluation be the basis for (i) the decision to act or not to act** (i.e. to implement precautionary measures or not) and (ii) the measures taken once a decision is made" (emphasis added) [39]

These policies make clear that the existence of a threat of serious or irreversible damage must be evidenced by a credible body of scientific evidence:

> "In determining what constitutes a sufficiently sound or credible scientific basis, the emphasis should be on providing a sound and credible basis that a risk of serious or irreversible harm exists… [which] should be interpreted as **a body of scientific information**…**that can establish reasonable evidence of a theory's validity**, including its uncertainties and that indicates the potential for such a risk."[40]

They underscore that evidencing a threat of serious or irreversible damage requires sound scientific analysis of the body of the evidence, undertaken in a transparent and credible manner:

> "Scientific data relevant to the risk **must be evaluated through a sound, credible, transparent and inclusive mechanism** leading to the conclusion that expresses the possibility of occurrence of harm and the magnitude of that harm (including the extent of possible damage, persistency, reversibility and delayed effect).[41]

Finally, the available scientific evidence must be evaluated in order to secure quality science, and to summarize not only the state of knowledge but "scientific views on the reliability of the assessment", including areas of uncertainty:

> "Available scientific information must be evaluated with emphasis on securing high quality scientific evidence (not quantity). Reports should summarize the existing state of

---

[39] Canada, Canada Privy Council Office, *A Framework for the Application of Precaution in Science-Based Decision Making about Risk*, (Ottawa, 2003) [Framework] at 7: "…sound scientific information and its evaluation must be the basis for applying precaution", Government of Canada.
[40] Framework, *ibid* at 7.
[41] Framework, *ibid* at 7.

> knowledge, provide scientific views on the reliability of the assessment and address remaining uncertainties and areas for further scientific research or monitoring."[42]

The Proposed Order, and the Literature Review supporting it, do not respect or apply any of these principles. They do not rest on scientific data, evaluated through sound, credible and transparent scientific analysis. They do not "evaluate the available scientific information" with a view to securing "high quality scientific evidence", nor does it express scientific views on the reliability of the assessment, or express a conclusion regarding not only the possibility of harm but the magnitude of that harm. In short, the Proposed Order and Literature Review fail to establish a "threat of serious or irreversible damage" on the basis of science, or risk assessment.

On the contrary, the Literature Review does little more than catalogue the lack of evidence, the data gaps, and the lack of standardized methods, in relation to the problem of plastic pollution in the environment. It is precisely because of the lack of evidence and sound science that the Literature Review concedes that it is simply not possible to conduct a proper chemical risk assessment.

When "conducting and interpreting the results of" toxicity assessments, the Minister has a duty under CEPA section 76.1 to apply a "weight of evidence approach", alongside the precautionary principle.[43] Health Canada defines the weight of evidence approach as:

> A qualitative measure that takes into account the **nature and quality of scientific studies intended to examine the risk of an agent**. Uncertainties that result from the incompleteness and unavailability of scientific data frequently require scientists to make inferences, assumptions, and judgements in order to characterize a risk. Making judgements about risk based on scientific information is called "evaluating the weight of evidence"[44]

The weight of evidence approach is the preferred tool for risk *assessment.* Meanwhile, ECCC policy describes the precautionary principle as the approach for risk *management*. This distinction is supported by various international authorities.[45]

---

[42] Framework, *ibid* at 7.

[43] CEPA, *supra* note 2, s 76.1: Weight of evidence and precautionary principle

> 76.1 When the Ministers are conducting and interpreting the results of
>
> (a) a screening assessment under section 74,
>
> (b) a review of a decision of another jurisdiction under subsection 75(3) that, in their opinion, is based on scientific considerations and is relevant to Canada, or
>
> (c) an assessment whether a substance specified on the Priority Substances List is toxic or capable of becoming toxic,
>
> the Ministers shall apply a weight of evidence approach and the precautionary principle.

[44] Canada, Health Canada, *Weight of Evidence: General Principles and Current Applications at Health Canada*, (Ottawa: Health Canada, 2018) at 2.

[45] European Science and Technology Observatory, *On Science and Precaution in the Management of Technological Risk,* vol 1 (Luxembourg: Prepared for the European Commission, 1999) at 17-18 [ESTO Report]; World Health Organization, Inter-organization Programme for Sound Management of Chemicals, - *Uncertainty and Data Quality in Exposure Assessment* (World Health Organization, International Labour Organization, United Nations Environment Programme, 2008) at 1 [WHO Guide]; European Union – *Communication from the*

The distinction is important here because risk assessment and risk management deal with two different types of uncertainty – only one of which invokes the precautionary principle. During risk assessment, uncertainty relates to whether or not risks exist (referred to here as "threshold uncertainty").

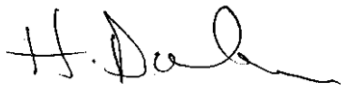
During risk management (where, by definition, risks have already been identified) the uncertainty relates to the prevalence, severity and nature of identified risks (referred to here as "impact uncertainty").

Because the precautionary principle presumes the existence of a risk, it is inappropriate to apply it during risk assessment, which is what has been done in relation to the Proposed Order. To do so presupposes the outcome, and (in the words of the European Science and Technology Observatory) confuses a state of "risk" with a state of "ignorance" – creating a "pretence at knowledge".[46]

In conclusion, for the reasons set out above, we respectfully request that a Board of Review be convened under section 332(2) of CEPA 1999.

Yours very truly,

Gowling WLG (Canada) LLP



Harry Dahme
Certified Specialist (Environmental Law)

Commission on the precautionary principle (Brussels: Commission of the European Communities, 2000) at 2, 3 & 12 [EU Communication].

[46] ESTO Report, *ibid* (citing Hayek's Nobel acceptance speech in relation to the phrase "pretence at knowledge").

HJD:ac

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Part I

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OTTAWA, LE SAMEDI 10 OCTOBRE 2020

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REGULATORY IMPACT ANALYSIS STATEMENT

(*This statement is not part of the Order.*)

Issues

Plastic manufactured items that are released into the environment outside of a managed waste stream, or that enter a managed waste stream but are accidentally released into the environment, constitute plastic pollution. Current scientific evidence confirms that plastic pollution is ubiquitous in the environment, and that macroplastic pollution poses an ecological hazard, such as physical harm to some animals and their habitat. Current scientific literature also suggests that microplastic pollution may pose an ecological hazard to some animals, though further research is needed. In order to address the potential ecological risks associated with certain plastic manufactured items becoming plastic pollution, the Minister of the Environment and the Minister of Health (the ministers) are recommending to the Governor in Council to make an order adding "plastic manufactured items" to Schedule 1 (i.e. the List of Toxic Substances) to the *Canadian Environmental Protection Act, 1999* (CEPA or the Act), in accordance with the precautionary principle.

Background

Description

Broadly speaking, plastics (which are the main ingredients in the manufacture of plastic items) are materials that can be created from a wide range of synthetic or semi-synthetic organic compounds. Plastics are formed from long-chain polymers of high molecular mass and often

Décret d'inscription d'une substance toxique à l'annexe 1 de la Loi canadienne sur la protection de l'environnement (1999)

Fondement législatif
Loi canadienne sur la protection de l'environnement (1999)

Ministères responsables
Ministère de l'Environnement
Ministère de la Santé

RÉSUMÉ DE L'ÉTUDE D'IMPACT DE LA RÉGLEMENTATION

(*Le présent résumé ne fait pas partie du Décret.*)

Enjeux

La pollution plastique est la résultante du rejet d'articles manufacturés en plastique dans l'environnement hors d'un flux de déchets géré ou entrant dans un flux de déchets géré, mais étant accidentellement rejetés dans l'environnement. Les données scientifiques actuelles confirment que la pollution plastique est omniprésente dans l'environnement, et que la pollution par les macroplastiques présente un danger pour l'environnement, comme des blessures physiques pour certains animaux et des dommages à leur habitat. La littérature scientifique actuelle suggère également que la pollution par les microplastiques peut présenter un danger pour l'environnement de certains animaux, bien que des recherches supplémentaires soient nécessaires. Afin de contrer les risques environnementaux potentiels associés à la pollution plastique découlant de certains articles manufacturés en plastique, le ministre de l'Environnement et la ministre de Santé (les ministres) recommandent à la gouverneure en conseil de prendre un décret pour ajouter « articles manufacturés en plastique » à la Liste des substances toxiques de l'annexe 1 de la *Loi canadienne sur la protection de l'environnement (1999)* [LCPE ou la Loi], conformément au principe de la prudence.

Contexte

Description

De façon générale, les plastiques (qui sont les principaux ingrédients des articles manufacturés en plastique) sont des matériaux qui peuvent être créés à partir d'un vaste éventail de composés organiques synthétiques ou semi-synthétiques. Les plastiques sont constitués de longues

contain chemical additives. Different polymers can be manufactured using different compositions of petroleum products, plant-based starting material, or recycled and recovered plastics.

Plastic manufactured items are any items made of plastic formed into a specific physical shape or design during manufacture, and have, for their intended use, a function or functions dependent in whole or in part on their shape or design. They can include final products, as well as components of products. All plastic manufactured items have the potential to become plastic pollution.

Plastic pollution is often categorized by size in the scientific literature. Individual pieces of plastic that are less than or equal to 5 mm in size can be defined as microplastics, while those that are greater than 5 mm in size can be defined as macroplastics. Microplastic pollution can be primary (smaller items that are manufactured to be that size), or secondary (smaller items resulting from the breakdown of larger plastic manufactured items).

Uses

Plastic manufactured items are a part of the everyday lives of Canadians and support economies around the world. Since the 1950s, the production and uses of plastics (to form plastic manufactured items) have increased at a faster rate than those of any other manufactured material, due to properties such as their versatility, durability, low cost, inert nature (i.e. non-chemical reactivity) and benefits to human health (e.g. in food and medical supplies packaging).

In order to better understand the quantities, uses, and end-of-life management of plastic manufactured items in the Canadian economy, the Department of Environment (the Department) commissioned the *Economic Study of the Canadian Plastic Industry, Market and Waste: Summary Report to Environment and Climate Change Canada* (the Commissioned Study), which was published in 2019. The Commissioned Study found that the majority of plastic manufactured items in Canada are concentrated in a number of sectors. The percentage of plastic manufactured items and corresponding amount of plastic waste generated by each of these sectors is detailed in Table 1.

chaînes de polymères de masse moléculaire élevée et renferment souvent des additifs chimiques. Différents polymères peuvent être fabriqués en utilisant diverses compositions de produits pétroliers, de matériaux de départ d'origine végétale ou de matières plastiques recyclées ou récupérées.

Les articles manufacturés en plastique comprennent tous les articles en plastique ayant une forme physique ou une conception spécifique durant leur fabrication et qui ont, pour leur utilisation prévue, une fonction ou des fonctions qui dépendent en tout ou en partie de leur forme ou de leur conception. Ces articles peuvent inclure des produits finis, ainsi que les composantes des produits. Tous les articles manufacturés en plastique ont le potentiel de devenir de la pollution plastique.

La pollution plastique est souvent catégorisée par la taille dans la littérature scientifique. Les morceaux de plastique individuels dont la taille est inférieure ou égale à 5 mm peuvent être définis comme des microplastiques, tandis que ceux dont la taille est supérieure à 5 mm peuvent être définis comme des macroplastiques. La pollution par les microplastiques peut être primaire (les articles plus petits étant manufacturés à cette taille) ou secondaire (les petites pièces provenant de la fragmentation d'articles en plastique manufacturés plus gros).

Utilisations

Les articles manufacturés en plastique font partie de la vie quotidienne de la population canadienne et soutiennent les économies du monde entier. Depuis les années 1950, la production et l'utilisation des plastiques (pour former des articles manufacturés en plastique) se sont accrues à un rythme plus élevé que celui de tout autre matériau manufacturé, en raison de propriétés telles que leur polyvalence, durabilité, faible coût, nature inerte (non chimiquement réactif), et des avantages pour la santé humaine (par exemple dans l'emballage des aliments et des fournitures médicales).

Afin de mieux comprendre les quantités, les utilisations et la gestion des articles manufacturés en plastique en fin de vie dans l'économie canadienne, le ministère de l'Environnement (le Ministère) a commandé l'*Étude économique sur l'industrie, les marchés et les déchets du plastique au Canada : Rapport sommaire à Environnement et Changement climatique Canada* (l'étude demandée), qui a été publiée en 2019. L'étude demandée a révélé que la majorité des articles manufacturés en plastique au Canada sont concentrés dans un certain nombre de secteurs. Le pourcentage des articles manufacturés en plastique et les quantités correspondantes de déchets de plastique générées par chacun de ces secteurs sont présentés de façon détaillée dans le tableau 1.

Table 1: Share of end-use plastic market in 2016 by sector and examples of plastic manufactured items

Sector	Share of End-use Plastic Market (%)	Plastic Waste (kt)	Share of Plastic Waste (%)	Examples of Plastic Manufactured Items
Packaging	33	1 542	47	Bags, drink bottles, toiletries, pharmaceutical product packaging
Construction	26	175	5	Siding, window applications, floor and wall coverings, thermal insulation, pipes and pipe fittings, glass substitutes, reconstituted wood, plywood
Automotive	10	309	9	Interior trims, seats, seat parts, body panels
Electronic and electrical equipment	6	214	7	Electric wires, cables, computer and phone parts
Textile	6	235	7	Carpets, rugs, mats, clothing
White goods (electric domestic appliances)	3	130	4	Major and small appliances, such as fridges, stoves, food processors, electric kettles
Agriculture	1	45	1	Fertilizer and pesticide packaging
Other	15	617	19	Chemical products, toys, household furniture

Tableau 1 : Part relative des marchés d'utilisation finale des produits en plastique en 2016 par secteur et exemples d'articles manufacturés en plastique

Secteur	Part du marché d'utilisation finale des produits en plastique (%)	Déchets de plastique (kt)	Part des déchets de plastique (%)	Exemples d'articles manufacturés en plastique
Emballage	33	1 542	47	Sacs, bouteilles en plastique utilisées pour les boissons, produits de toilette, emballages des produits pharmaceutiques
Construction	26	175	5	Recouvrements extérieurs, vitrages des fenêtres, revêtements de plancher et revêtements muraux, isolation thermique, tuyaux et raccords de tuyauterie, substitut du verre, bois reconstitué, panneaux de contre-plaqué
Automobile	10	309	9	Garnitures intérieures, sièges, parties de siège, panneaux de carrosserie
Équipements électriques et électroniques	6	214	7	Fils électriques, câbles, pièces d'ordinateurs et de téléphones
Textile	6	235	7	Tapis, moquettes, vêtements
Produits blancs (appareils électroménagers)	3	130	4	Gros et petits appareils électroménagers comme les réfrigérateurs, les poêles-cuisinières, les robots culinaires, les bouilloires électriques
Agriculture	1	45	1	Emballage des engrais et des pesticides
Autres	15	617	19	Produits chimiques, jouets, mobilier d'habitation

Sources of release

In Canada, the majority of plastic manufactured items that become plastic waste enter a managed waste stream (i.e. intended for landfilling, recycling, or incineration). Plastic waste that is released into the environment outside of a managed waste stream, or that enters a managed waste stream but is accidentally released into the environment,

Sources de rejet

Au Canada, la majorité des articles manufacturés en plastique qui deviennent des déchets de plastique entrent dans un flux de déchets (c'est-à-dire destinés à l'enfouissement, au recyclage ou à l'incinération). Les déchets de plastique qui sont rejetés dans l'environnement en dehors d'un flux de déchets, ou qui entrent dans un flux de

constitutes plastic pollution. The Commissioned Study estimated that the total amount of plastic waste generated in Canada in 2016 was 3 268 kilotonnes (kt), of which 2 795 kt (86%) ended up in a landfill, 305 kt (9%) was recycled, 137 kt (4%) was incinerated, and 29 kt (1%) was released into the environment as plastic pollution.

Plastic manufactured items can be released into the environment as plastic pollution through a wide range of activities including littering, environmental emergencies (e.g. flooding events), and through the wear and tear, abrasion, or maintenance of certain items. They can also be accidentally released into the environment while moving through a managed waste stream, for example by falling out or being blown away during transport, transfer, or processing, or due to inadequate waste, wastewater, and stormwater management practices. Plastic pollution can be released into terrestrial or aquatic environments and can move from one to the other over its lifetime.

Risk management activities

National

Plastic manufactured items encompass a wide range of product categories within many sectors of the plastics end-use market, some of which may already be subject to federal risk management activities. For example, aspects of plastic manufactured items relating to consumer safety, energy efficiency, and human health may already be regulated under various Acts of Parliament.[1] Limited federal risk management exists for plastic manufactured items with respect to environmental protection, with one example being the *Microbeads in Toiletries Regulations*, enacted under CEPA, that prohibit the manufacture, import, and sale of toiletries containing plastic microbeads.

Other jurisdictions in Canada are currently taking a range of actions consistent with the Strategy on Zero Plastic Waste of the Canadian Council of Ministers of the

déchets, mais qui sont accidentellement rejetés dans l'environnement, constituent de la pollution plastique. L'étude demandée a estimé que la quantité totale de déchets de plastique générés au Canada en 2016 était de 3 268 kilotonnes (kt), dont 2 795 kt (86 %) se sont retrouvées dans un site d'enfouissement, 305 kt (9 %) ont été recyclées, 137 kt (4 %) ont été incinérées, et 29 kt (1 %) ont été rejetées dans l'environnement sous forme de pollution plastique.

Les articles manufacturés en plastique peuvent être rejetés dans l'environnement sous forme de pollution plastique par le biais d'un large éventail d'activités, y compris l'abandon de détritus, les urgences environnementales (par exemple les inondations) et par l'usure normale, l'abrasion et l'entretien de certains articles. Ces articles peuvent être également rejetés accidentellement dans l'environnement alors que ceux-ci circulent dans un flux de déchets géré, par exemple en tombant ou en étant emportés par le vent pendant le transport, le transfert ou la transformation, ou en raison de pratiques de gestion inadéquate des déchets, des eaux usées et des eaux pluviales. Les déchets associés à la pollution plastique peuvent être rejetés dans les milieux terrestres ou aquatiques et circuler de l'un à l'autre tout au long de leur durée de vie.

Activités de gestion des risques

À l'échelle nationale

Les articles manufacturés en plastique comprennent un large éventail de catégories de produits dans de nombreux secteurs du marché de l'utilisation finale des plastiques, dont certains peuvent déjà être visés par des activités de gestion des risques mises en œuvre par le gouvernement fédéral. Par exemple, certains aspects des articles manufacturés en plastique liés à la sécurité des consommateurs, à l'efficacité énergétique et à la santé humaine peuvent déjà être réglementés par diverses lois du Parlement[1]. Une gestion fédérale limitée des risques existe pour les articles manufacturés en plastique en ce qui concerne la protection de l'environnement, par exemple le *Règlement sur les microbilles dans les produits de toilette*, promulgué en vertu de la LCPE, qui interdit la fabrication, l'importation et la vente de produits de toilette contenant des microbilles de plastique.

D'autres administrations au Canada prennent actuellement une série de mesures conformes à la Stratégie visant l'atteinte de zéro déchet de plastique du Conseil canadien

[1] Some of these Acts of Parliament may include the *Canada Consumer Product Safety Act*, the *Consumer Packaging and Labelling Act*, the *Pest Control Products Act*, the *Customs Act*, the *Energy Efficiency Act*, and the *Food and Drugs Act* (e.g. through the *Food and Drugs Regulations*, the *Natural Health Products Regulations*, and the *Medical Device Regulations*).

[1] Certaines de ces lois du Parlement peuvent comprendre la *Loi canadienne sur la sécurité des produits de consommation*, la *Loi sur l'emballage et l'étiquetage des produits de consommation*, la *Loi sur les produits antiparasitaires*, la *Loi sur les douanes*, la *Loi sur l'efficacité énergétique* et la *Loi sur les aliments et drogues* (par exemple au moyen de l'application du *Règlement sur les aliments et drogues*, du *Règlement sur les produits de santé naturels* et du *Règlement sur les instruments médicaux*).

Environment. For instance, recycling systems that process plastic waste exist in all provinces, and some domestic jurisdictions have established or are developing requirements to make producers responsible for the collection of the products and packaging they place on the market. Some domestic jurisdictions at the provincial or municipal level have announced local prohibitions or restrictions on certain single-use plastics, such as plastic bags.

These provincial and municipal risk management measures were designed and implemented to address jurisdictional waste reduction and waste management needs, and thereby result in strictly localized impacts. There is currently no existing Canada-wide integrated management of plastics that cover a range of lifecycle stages (e.g. design and manufacture, import, use, waste management) and different plastic sectors (e.g. packaging, single-use items).

International

Several international jurisdictions are pursuing measures to address plastic pollution. For example, the European Union (EU) adopted a directive to prevent production of packaging waste and to promote the reuse, recycling, and other forms of recovering packaging waste, alongside another directive to ban nine single-use plastic items for which alternatives exist on the market (e.g. cutlery, plates, beverage stirrers, and cotton bud sticks). Germany and France, in particular, are leading the way with national bans on several single-use plastic manufactured items. The United States does not have any federal laws or requirements for plastic waste, recycling, or extended producer responsibility, though many individual states (e.g. California, Maine) have implemented waste reduction and recycling programs concerning plastic products and packaging, and eight states thus far have passed bans on single-use plastic bags.[2] Other international jurisdictions, notably Australia and China, have announced actions such as sector-based targets for plastic waste.

[2] As of June 2020, these states are California, Oregon, Hawaii, New York, Vermont, Maine, Connecticut, and Delaware. In light of the novel coronavirus (COVID-19) pandemic, some of these jurisdictions have announced delays to their coming-into-force dates or enforcement of those bans, though none have been repealed.

des ministres de l'environnement. Par exemple, des systèmes de recyclage qui transforment les déchets de plastique existent dans toutes les provinces, et certaines administrations au pays ont établi ou élaborent des exigences pour rendre les producteurs responsables de la collecte des produits et des emballages qu'ils mettent en marché. Certaines administrations au pays au palier provincial ou municipal ont annoncé des interdictions ou des restrictions locales sur certains articles de plastique à usage unique, comme les sacs de plastique.

Ces mesures de gestion des risques provinciales et municipales ont été conçues et mises en œuvre pour répondre aux besoins de réduction et de gestion des déchets dans leur administration, et par conséquent entraînent des incidences strictement localisées. À l'heure actuelle, il n'existe aucune gestion pancanadienne intégrée des plastiques qui couvre l'ensemble des étapes du cycle de vie (par exemple la conception et la fabrication, l'importation, l'utilisation, la gestion des déchets) et les différents secteurs des produits en plastique (par exemple l'emballage, les articles à usage unique).

À l'échelle internationale

Plusieurs administrations internationales mettent en œuvre des mesures pour contrer la pollution plastique. Par exemple, l'Union européenne a adopté une directive visant à prévenir la production de déchets d'emballage et à promouvoir la réutilisation, le recyclage et d'autres formes de récupération des déchets d'emballage, parallèlement à une autre directive interdisant neuf articles en plastique à usage unique pour lesquels des solutions de rechange existent sur le marché (par exemple ustensiles, assiettes, bâtonnets à mélanger et bâtonnets de coton). L'Allemagne et la France, en particulier, ouvrent la voie avec l'interdiction nationale de plusieurs articles manufacturés en plastique à usage unique. Les États-Unis n'ont aucune législation ou exigence fédérale concernant les déchets de plastique, le recyclage ou la responsabilité élargie des producteurs, bien que de nombreux États individuels (par exemple la Californie et le Maine) aient mis en œuvre des programmes de réduction et de recyclage des déchets visant les produits et les emballages en plastique, et à ce jour, huit États ont adopté des interdictions sur les sacs en plastique à usage unique[2]. D'autres administrations internationales, notamment l'Australie et la Chine, ont annoncé des mesures ciblées pour les déchets de plastique pour certains secteurs.

[2] En date de juin 2020, ces États sont la Californie, l'Oregon, Hawaii, New York, le Vermont, le Maine, le Connecticut et le Delaware. En raison de la pandémie du nouveau coronavirus (COVID-19), certaines de ces administrations ont reporté l'entrée en vigueur et l'application de ces programmes, quoiqu'aucun n'ait été abrogé.

Science assessment of plastic pollution

On October 7, 2020, a science assessment of plastic pollution was published on the Canada.ca (Chemical Substances) website. The purpose of the assessment was to summarize the current state of the science regarding the potential impacts of plastic pollution on the environment and human health, as well as to inform future research and decision making on plastic pollution in Canada.[3] The science assessment recommends pursuing action to reduce macroplastics and microplastics that end up in the environment, in accordance with the precautionary principle.

Summary of the state of the science with respect to the environment

The degradation of plastic pollution in the environment can be a slow chemical and physical process, influenced by factors such as exposure to sunlight, oxidants, physical stress and the chemical composition of the specific plastic item. Many plastic manufactured items identified as "biodegradable" only break down when exposed to high temperatures for prolonged periods that are only achievable in industrial composting facilities.

Studies have confirmed the widespread occurrence of plastic pollution in many aquatic and terrestrial environments around the globe, including surface waters, sediments, and shorelines. For example, in Canada, studies have found an abundance of plastic pollution in surface waters and sediments within the Great Lakes, as well as in Arctic surface waters and in sea ice. Plastic pollution has also been detected in several international study locations, including the Adriatic Sea, the South Pacific, the North Pacific, the North Atlantic, the South Atlantic, the Indian Ocean, and in the waters surrounding Australia. In 2018, the Great Canadian Shoreline Cleanup removed over 100 tonnes of litter from Canadian shorelines, with 7 out of the top 10 most commonly collected items either being made of plastic or containing plastics (cigarette butts, tiny plastics or foam, bottle caps, plastic bags, plastic bottles, straws, and food wrappers).

Certain types of macroplastic pollution (e.g. ropes, nets, cable ties, plastic bags, packaging rings) have been widely reported in the scientific literature to exhibit adverse effects on some animals through entanglement. It has also been observed that large pieces of plastic pollution (e.g.

[3] The science assessment followed a similar approach to that taken for the science summary on microbeads.

Évaluation scientifique de la pollution plastique

Le 7 octobre 2020, une évaluation scientifique de la pollution plastique a été publiée sur le site Web Canada.ca (Substances chimiques). L'objectif de l'évaluation était de résumer l'état actuel des connaissances scientifiques sur les impacts potentiels de la pollution plastique sur l'environnement et la santé humaine, ainsi que d'orienter les futures recherches et de contribuer à la prise de décision à ce sujet au Canada[3]. L'évaluation scientifique recommande la prise de mesures visant à réduire les macroplastiques et les microplastiques qui se retrouvent dans l'environnement, conformément au principe de la prudence.

Résumé de l'état des connaissances relatives à l'environnement

La dégradation des déchets de plastique qui polluent l'environnement peut être un lent processus chimique et physique, influencé par des facteurs comme l'exposition à la lumière du soleil, les oxydants, le stress physique et la composition chimique des articles de plastique spécifiques. De nombreux articles manufacturés en plastique désignés comme « biodégradables » se décomposent uniquement lorsqu'ils sont exposés à des températures élevées durant des périodes prolongées qui ne sont réalisables que dans les installations de compostage industrielles.

Des études ont confirmé la présence généralisée de la pollution plastique dans de nombreux milieux aquatiques et terrestres partout dans le monde, y compris les eaux de surface, les sédiments et les zones côtières. Par exemple, au Canada, des études ont révélé une abondance de pollution plastique dans les eaux de surface et les sédiments dans les Grands Lacs, ainsi que dans les eaux de surface et les glaces océaniques dans l'Arctique. La pollution plastique a aussi été détectée dans de nombreux emplacements lors d'études internationales, y compris la mer Adriatique, le Pacifique Sud, le Pacifique Nord, l'Atlantique Nord, l'Atlantique Sud, l'océan Indien et dans les eaux entourant l'Australie. En 2018, l'initiative du Grand nettoyage des rivages canadiens a contribué à l'élimination de 100 tonnes de détritus dans les zones côtières canadiennes; parmi les 10 objets les plus fréquemment recueillis, 7 étaient des articles de plastique ou des articles contenant du plastique (mégots de cigarette, fines particules de plastique ou de mousse, bouchons de bouteille, sacs et bouteilles de plastique, pailles et papiers d'emballage alimentaire).

Certains types d'articles qui contribuent à la pollution macroplastique (par exemple les cordes, les filets, les attaches de câbles, les sacs de plastique, les anneaux de plastique) ont été largement décrits dans la littérature scientifique comme ayant des effets néfastes sur certains

[3] L'évaluation scientifique a adopté une approche semblable à celle décrite dans le document Microbilles – Résumé scientifique.

bags, sheets, films) can smother marine plants, sponges, and coral, affecting biological processes such as photosynthesis. In addition, the scientific literature depicts that macroplastic pollution has been found in the gastrointestinal tracts of several marine species all around the world, which has been linked to several adverse ecological impacts such as organ damage and starvation from blocked intestinal systems. Macroplastic pollution can also impact the integrity of habitats, for example by transporting non-native species, invasive species, or species containing diseases into well-established ecosystems, disrupting their structures and dynamics. In contrast to macroplastic pollution, the potential impact of microplastic pollution on animals is less understood in the scientific literature.

Summary of the state of the science with respect to human health

Exposure to macroplastics (as pollution or otherwise) is not expected to be of concern for human health. There is some scientific literature to suggest that humans may be exposed to microplastics through the inhalation of air, and the ingestion of food and drinking water. The potential hazards of microplastics from inhalation remain uncertain, and there is need for further research in this area. Current knowledge of the occurrence of microplastic particles in food is limited, with little to no Canadian data. The World Health Organization carried out an assessment of human exposure to microplastic particles in drinking water, and the Food and Agriculture Organization of the United Nations and the European Food Safety Authority conducted similar assessments of exposure to microplastic particles in seafood, which concluded that potential ingestion of chemicals associated with microplastics is of low concern for human health. Although the current literature does not identify a concern for human health with respect to microplastics, there is need for further research in this area.

Objective

The objective of the proposed *Order Adding a Toxic Substance to Schedule 1 to the Canadian Environmental Protection Act, 1999* (the proposed Order) is to add "plastic manufactured items" to Schedule 1 to CEPA. The proposed Order would enable the ministers to propose risk management measures under CEPA on certain plastic manufactured items in order to manage the potential

animaux qui s'enchevêtrent dans ces matériaux. Il a également été observé que les gros morceaux contribuant à la pollution plastique (par exemple les sacs, les feuilles, les films) peuvent étouffer les plantes marines, les éponges et les coraux, ce qui affecte ainsi les processus biologiques tels que la photosynthèse. De plus, la littérature scientifique indique que de la pollution macroplastique a été détectée dans le tractus gastro-intestinal de plusieurs espèces marines partout dans le monde. Ces observations sont liées à plusieurs impacts écologiques négatifs tels que des dommages aux organes et la mort par la faim à la suite de l'obturation des systèmes intestinaux. La pollution macroplastique peut aussi avoir des incidences sur l'intégrité des habitats, par exemple par le transport d'espèces exotiques, d'espèces envahissantes ou d'espèces transportant des maladies dans des écosystèmes bien établis, ce qui perturbe leurs structures et leur dynamique. Contrairement à la pollution macroplastique, l'impact potentiel de la pollution microplastique sur les animaux est moins bien compris dans la littérature scientifique.

Résumé de l'état des connaissances relatives à la santé humaine

L'exposition aux macroplastiques (sous forme de pollution ou autrement) ne devrait pas être une source de préoccupation pour la santé humaine. Certaines publications scientifiques suggèrent que les humains peuvent être exposés à des microplastiques par l'inhalation d'air et par l'ingestion d'aliments et d'eau potable. Les dangers potentiels des microplastiques par l'inhalation demeurent incertains et des recherches supplémentaires sont nécessaires dans ce domaine. Les connaissances actuelles sur la présence de particules de microplastiques dans les aliments sont limitées, et il existe peu ou aucune donnée canadienne. L'Organisation mondiale de la Santé a effectué une évaluation de l'exposition humaine aux particules de microplastiques dans l'eau potable, et l'Organisation des Nations Unies pour l'alimentation et l'agriculture et l'Autorité européenne de sécurité des aliments ont mené des évaluations similaires de l'exposition aux particules de microplastiques présentes dans les fruits de mer. Les auteurs de ces évaluations ont conclu que l'ingestion de substances chimiques due à l'exposition aux microplastiques suscite un faible degré de préoccupation pour la santé humaine. Bien qu'aucun rapport n'ait formulé d'inquiétudes pour la santé humaine relativement à l'ingestion de particules de microplastiques, des recherches supplémentaires sont nécessaires dans ce domaine.

Objectif

Le *Décret d'inscription d'une substance toxique à l'annexe 1 de la Loi canadienne sur la protection de l'environnement (1999)* proposé (le projet de décret) vise à ajouter « articles manufacturés en plastique » à l'annexe 1 de la LCPE. Le projet de décret permettrait aux ministres de proposer des mesures de gestion des risques en vertu de la LCPE, qui s'appliqueraient à certains articles

ecological risks associated with those items becoming plastic pollution.

Description

The proposed Order would add "plastic manufactured items" to Schedule 1 to CEPA.

Regulatory development

Consultation

From April 2018 to May 2020, the Government of Canada undertook broad stakeholder engagement on achieving zero plastic waste. During that period and across those engagements, the Department received input from multiple stakeholder groups (e.g. industry, industry associations, non-government organizations, provinces, territories, the general public) on options, barriers, and solutions to achieve zero plastic waste in Canada, including adding a substance relating to plastics to Schedule 1 to CEPA. The Department consulted on various policy initiatives through

- consultations on "Moving Canada toward zero plastic waste" using the online platform PlaceSpeak from April 22, 2018, to September 21, 2018;
- a 2018 ministerial plastics advisory group composed of stakeholders from industry and civil society; and
- the development, with the Canadian Council of Ministers of the Environment, of the Canada-wide Strategy on Zero Plastic Waste (2018), and both phase 1 (2019) and phase 2 (2020) of the Canada-wide Action Plan on Zero Plastic Waste.

Other engagement activities included public calls for input, two industry sector webinars, workshops, meetings, teleconferences, and presentations to industry stakeholders across the entire plastics value chain.

On February 1, 2020, the ministers published a notice with a summary of the draft science assessment of plastic pollution (which included a link to the complete draft assessment) in the *Canada Gazette*, Part I, for a 60-day public comment period, which was extended until May 1, 2020, in light of the novel coronavirus (COVID-19) pandemic. Over 70 comments were received from different stakeholder groups, including over 50 from businesses and industry associations. Several of these comments related to potential risk management measures, and will be considered in the event that the ministers propose to develop such measures. A table summarizing all comments received and the

manufacturés en plastique afin de gérer les risques écologiques potentiels associés au fait que ces articles deviennent de la pollution plastique.

Description

Le projet de décret ajouterait « articles manufacturés en plastique » à l'annexe 1 de la LCPE.

Élaboration de la réglementation

Consultation

D'avril 2018 à mai 2020, le gouvernement du Canada a entrepris une vaste mobilisation des parties prenantes visant l'atteinte de l'objectif de zéro déchet de plastique. Tout au long de cette période d'engagement, le Ministère a reçu des commentaires de multiples groupes de parties intéressées (par exemple l'industrie, des associations industrielles, des organisations non gouvernementales, des provinces, des territoires, le grand public) concernant les options, les obstacles et les solutions pour atteindre l'objectif de zéro déchet de plastique au Canada, dont l'ajout d'une substance relative aux plastiques à l'annexe 1 de la LCPE. Le Ministère a mené des consultations sur diverses initiatives stratégiques :

- Consultations sur « Direction : zéro déchet de plastique au Canada » par l'entremise de la plateforme PlaceSpeak, du 22 avril 2018 jusqu'au 21 septembre 2018;
- Le groupe consultatif sur les plastiques ministériels de 2018, composé d'intervenants de l'industrie et de la société civile;
- Le développement, élaboré avec le Conseil canadien des ministres de l'environnement, de la Stratégie pancanadienne visant l'atteinte de zéro déchet de plastique (2018), ainsi que la phase 1 (2019) et la phase 2 (2020) du Plan d'action pancanadien visant l'atteinte de zéro déchet de plastique.

Les activités d'engagement additionnelles comprenaient des consultations avec le public, deux webinaires avec le secteur de l'industrie, des ateliers, des réunions, des téléconférences et des présentations aux intervenants de l'industrie dans toute la chaîne de valeur des plastiques.

Le 1er février 2020, les ministres ont publié un avis et un sommaire de l'ébauche d'évaluation scientifique de la pollution plastique (qui comprenait un lien vers l'ébauche d'évaluation complète) dans la Partie I de la *Gazette du Canada*, pour une période de commentaires du public de 60 jours, qui a été prolongée jusqu'au 1er mai 2020 en raison de la pandémie du nouveau coronavirus (COVID-19). Plus de 70 commentaires ont été reçus de différents groupes d'intervenants, dont plus de 50 d'entreprises et d'associations industrielles. Plusieurs de ces observations portaient sur d'éventuelles mesures de gestion des risques et seront examinées dans le cas où les ministres proposeraient

departments' responses to those comments is available on the Canada.ca (Chemical Substances) website.

Of the comments pertaining to a potential new addition to Schedule 1 to CEPA, some industry stakeholders expressed concerns with adding a substance relating to plastics, arguing that such an addition could lead to the stigmatization of plastics in the Canadian economy. Other stakeholders questioned whether enough evidence had been presented in the draft science assessment of plastic pollution to warrant adding a substance to Schedule 1 to CEPA, and expressed concerns regarding subsequent use of the regulatory authorities for controlling toxic substances, under Part 5 of CEPA, to address plastic pollution. These stakeholders argued that a Schedule 1 listing could lead to the over-regulation of the plastics value chain in Canada.

The Department acknowledges the concerns and suggestions raised by stakeholders, and considered all comments received, except those pertaining to risk management, in the development of the proposed Order. Since the proposed Order would not introduce new regulatory requirements, consideration of potential impacts to plastics-related industries and the broader Canadian economy would occur only in the event that the ministers propose risk management measures for plastic manufactured items (as discussed in the "Benefits and costs" paragraph below). Additionally, the Department maintains that the science assessment of plastic pollution provides the ministers with the evidence to recommend the addition of "plastic manufactured items" to Schedule 1 to CEPA in accordance with the precautionary principle, which would be consistent with the recommendation from the science assessment to take action to address plastic pollution.

On October 7, 2020, the Department published a discussion paper entitled A Proposed Integrated Management Approach to Plastic Products to Prevent Waste and Pollution to engage with stakeholders on the design and implementation of potential risk management measures for certain plastic manufactured items, including regulatory instruments to ban single-use plastic items that cause harm to the environment, where warranted and supported by scientific evidence.

d'élaborer de telles mesures. Un tableau résumant tous les commentaires reçus et les réponses des ministères à ces commentaires est disponible sur le site Web de Canada.ca (Substances chimiques).

Parmi les commentaires concernant l'ajout d'une nouvelle substance à l'annexe 1 de la LCPE, certains intervenants de l'industrie ont exprimé des préoccupations quant à l'ajout d'une substance relative aux plastiques, faisant valoir qu'une telle inscription pourrait entraîner la stigmatisation des plastiques dans l'économie canadienne. D'autres intervenants ont remis en doute s'il y avait suffisamment de preuves présentées dans l'ébauche d'évaluation scientifique de la pollution plastique pour justifier l'ajout d'une substance à l'annexe 1 de la LCPE, et ont exprimé des préoccupations au sujet de l'utilisation subséquente des autorités réglementaires pour contrôler les substances toxiques, en vertu de la partie 5 de la LCPE, pour lutter contre la pollution plastique. Ces intervenants ont soutenu qu'une inscription à l'annexe 1 pourrait mener à une surréglementation de la chaîne de valeur des plastiques au Canada.

Le Ministère reconnaît les préoccupations et les suggestions soulevées par les intervenants et a tenu compte de tous les commentaires reçus, à l'exception de ceux relatifs à la gestion des risques, dans l'élaboration du projet de décret. Étant donné que le projet de décret n'introduirait pas de nouvelles exigences réglementaires, l'examen des répercussions possibles sur les industries liées aux plastiques et l'économie canadienne dans son ensemble ne se produirait que si les ministres proposaient des mesures de gestion des risques pour les articles manufacturés en plastique (comme il est expliqué dans le paragraphe « Avantages et coûts » ci-dessous). De plus, le Ministère soutient que l'évaluation scientifique de la pollution plastique fournit aux ministres les éléments de preuve qui recommandent l'ajout d'« articles manufacturés en plastique » à l'annexe 1 de la LCPE conformément au principe de la prudence, ce qui serait conforme à la recommandation de l'évaluation scientifique qui vise à prendre des mesures pour contrer la pollution plastique.

Le 7 octobre 2020, le Ministère a publié un document de consultation intitulé « Une approche proposée de gestion intégrée des produits de plastique visant à réduire les déchets et à prévenir la pollution » en vue de susciter la participation des parties intéressées à la conception et à la mise en œuvre des mesures de gestion des risques potentielles s'appliquant à certains articles manufacturés en plastique, y compris les instruments réglementaires pour interdire les articles de plastique à usage unique qui sont dommageables pour l'environnement, dans les cas où il existe des éléments scientifiques probants et où cela est justifié.

Modern treaty obligations and Indigenous engagement and consultation

The assessment of modern treaty implications conducted in accordance with the *Cabinet Directive on the Federal Approach to Modern Treaty Implementation* concluded that orders making additions to Schedule 1 to CEPA do not impose any new regulatory requirements, and, therefore, do not result in any impact on modern treaty rights or obligations. As a result, specific engagement and consultations with Indigenous peoples were not undertaken. However, the prepublication comment period, which is open to all Canadians, is an opportunity for Indigenous peoples to provide feedback on the proposed Order. In the event that the ministers propose risk management measures for plastic manufactured items, the departments would assess any associated impact on modern treaty rights or obligations, and requirements for Indigenous engagement and consultations, during the development of such measures.

Instrument choice

The Government of Canada has initiated a comprehensive agenda to achieve zero plastic waste and eliminate plastic pollution by 2030, which will require implementing a range of risk management measures. The Department determined that non-regulatory measures (e.g. voluntary agreements, guidelines, codes of practice) alone would not be sufficient to implement this agenda, and that regulatory measures would also be required.

The addition of a substance to Schedule 1 to CEPA enables the ministers to propose risk management measures. A substance may be listed if it is found to meet any of the criteria set out in section 64 of the Act (i.e. if the substance poses a risk to the environment, human health, or both). The science assessment of plastic pollution provided the ministers with the evidence to recommend adding "plastic manufactured items" to Schedule 1 to CEPA, an action which would help address the potential ecological risks associated with plastic manufactured items becoming plastic pollution. The use of CEPA over other existing Acts of Parliament would enable the ministers to access the full range of authorities needed to manage plastic manufactured items along their entire lifecycle. Therefore, adding "plastic manufactured items" to Schedule 1 to CEPA is the preferred option.

The addition of "plastic manufactured items" to Schedule 1 to CEPA would be made in accordance with paragraph 2(1)(a) of the Act, which requires the Government

Obligations relatives aux traités modernes et consultation et mobilisation des Autochtones

L'évaluation des incidences sur les traités modernes menée conformément à la *Directive du Cabinet sur l'approche fédérale pour la mise en œuvre des traités modernes* a conclu que les décrets visant l'ajout de substances à l'annexe 1 de la LCPE n'imposent aucune nouvelle exigence réglementaire et, par conséquent, n'ont aucune incidence sur les droits ou obligations découlant des traités modernes. Par conséquent, un engagement et des consultations spécifiques avec les peuples autochtones n'ont pas été entrepris. Cependant, la période de commentaires avant la publication, qui est ouverte à tous les Canadiens, est une occasion pour les peuples autochtones de faire part de leurs commentaires sur le projet de décret. Dans l'éventualité où les ministres proposeraient des mesures de gestion des risques pour les articles manufacturés en plastique, les ministères évalueraient tout impact connexe sur les droits ou obligations issus de traités modernes, et les exigences en matière d'engagement et de consultations des Autochtones, pendant l'élaboration de ces mesures.

Choix de l'instrument

Le gouvernement du Canada a amorcé un programme exhaustif pour atteindre l'objectif de zéro déchet de plastique et éliminer la pollution plastique d'ici 2030, ce qui exigera la mise en œuvre d'une gamme de mesures de gestion des risques. Le Ministère a déterminé que les mesures non réglementaires seules (par exemple les ententes volontaires, les directives, les codes de pratique) ne suffiraient pas à mettre en œuvre ce programme, et que des mesures réglementaires seraient aussi requises.

L'ajout d'une substance à l'annexe 1 de la LCPE permet aux ministres de proposer des mesures de gestion des risques. Une substance peut être inscrite sur la liste si celle-ci répond à l'un des critères énoncés à l'article 64 de la Loi (c'est-à-dire si la substance présente un risque pour l'environnement, la santé humaine ou les deux). L'évaluation scientifique de la pollution plastique fournit aux ministres les éléments probants pour recommander l'ajout d'« articles manufacturés en plastique » à l'annexe 1 de la LCPE, une mesure qui aiderait à contrer les risques écologiques potentiels associés aux articles manufacturés en plastique qui contribuent à la pollution plastique. La préséance de la LCPE sur d'autres lois du Parlement existantes permettrait aux ministres d'avoir accès à l'ensemble des pouvoirs requis pour gérer les articles manufacturés en plastique tout au long de leur cycle de vie. Par conséquent, l'ajout d'« articles manufacturés en plastique » à l'annexe 1 de la LCPE est l'option privilégiée.

L'ajout d'« articles manufacturés en plastique » à l'annexe 1 de la LCPE se ferait conformément à l'alinéa 2(1)a) de la Loi, ce qui exige du gouvernement du Canada qu'il

of Canada to exercise its powers in the administration of the Act in a manner that

- protects the environment;
- applies the precautionary principle where, if there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason to postpone cost-effective measures to prevent environmental degradation; and
- promotes and reinforces enforceable pollution prevention approaches.

Regulatory analysis

Benefits and costs

The addition of "plastic manufactured items" to Schedule 1 to CEPA would not on its own impose any regulatory requirements on businesses or other entities, and would therefore not result in any incremental compliance costs for stakeholders or enforcement costs for the Government of Canada. The proposed Order would grant the ministers the authority to develop risk management measures under CEPA for plastic manufactured items. If pursued, these measures could result in incremental costs for stakeholders and the Government of Canada. In the event that the ministers propose risk management measures for plastic manufactured items, the departments would assess their benefits and costs, and would conduct consultations with stakeholders, Indigenous peoples, the public, and other interested parties during the development of such measures.

Small business lens

The small business lens analysis concluded that the proposed Order would have no associated impact on small business, as it does not impose any administrative or compliance costs on businesses. In the event that the ministers propose risk management measures for plastic manufactured items, the departments would assess any associated impact on small businesses during the development of such measures.

One-for-one rule

The one-for-one rule does not apply to the proposed Order, as there are no incremental changes in administrative burden imposed on businesses. In the event that the ministers propose risk management measures for plastic manufactured items, the departments would assess any associated administrative burden during the development of such measures.

exerce ses pouvoirs relatifs à l'administration de la Loi de manière à :

- protéger l'environnement;
- appliquer le principe de la prudence si bien qu'en cas de risques de dommages graves ou irréversibles, l'absence de certitude scientifique absolue ne doit pas servir de prétexte pour remettre l'adoption de mesures effectives visant à prévenir la dégradation de l'environnement;
- promouvoir et affermir les méthodes applicables à la prévention de la pollution.

Analyse de la réglementation

Avantages et coûts

L'ajout d'« articles manufacturés en plastique » à l'annexe 1 de la LCPE n'imposerait pas en soi des exigences réglementaires aux entreprises ou à d'autres entités, et, par conséquent, n'entraînerait aucun coût supplémentaire associé à la conformité pour les parties intéressées ou de coûts d'application pour le gouvernement du Canada. Le projet de décret donnerait aux ministres le pouvoir d'élaborer des mesures de gestion des risques en vertu de la LCPE pour les articles manufacturés en plastique. Si ces mesures étaient mises en œuvre, cela pourrait entraîner des coûts supplémentaires pour les parties intéressées et le gouvernement du Canada. Advenant le cas où les ministres proposeraient des mesures de gestion des risques pour les articles manufacturés en plastique, les ministères évalueraient leurs avantages et leurs coûts, et mèneraient des consultations auprès des intervenants, des peuples autochtones, du public et d'autres parties intéressées.

Lentille des petites entreprises

L'analyse de la lentille des petites entreprises a conclu que le projet de décret n'aurait aucune incidence associée sur les petites entreprises, car celui-ci n'impose pas de fardeau administratif ou de coûts associés à la conformité pour les entreprises. Advenant le cas où les ministres proposeraient des mesures de gestion des risques pour les articles manufacturés en plastique, les ministères évalueraient toute incidence associée sur les petites entreprises lors de l'élaboration de telles mesures.

Règle du « un pour un »

La règle du « un pour un » ne s'applique pas au projet de décret, car il n'y a aucun changement progressif du fardeau administratif imposé sur les entreprises. Advenant le cas où les ministres proposeraient des mesures de gestion des risques pour les articles manufacturés en plastique, les ministères évalueraient tout fardeau administratif associé lors de l'élaboration de telles mesures.

Regulatory cooperation and alignment

The proposed Order would not directly relate to any domestic or international agreements or obligations. The proposed Order would enable the ministers to propose risk management measures that could align and complement actions undertaken by provincial, territorial, and municipal governments toward a coordinated effort to achieve zero plastic waste and eliminate plastic pollution by 2030.

Strategic environmental assessment

In accordance with the *Cabinet Directive on the Environmental Assessment of Policy, Plan and Program Proposals*, a strategic environmental assessment (SEA) was completed for the Government of Canada's comprehensive zero plastic waste agenda. The analysis concluded that the full implementation of this agenda will have a significant positive effect on the environment and on Canada's ability to deliver on its Federal Sustainable Development Strategy, Canada-wide Strategy on Zero Plastic Waste, and commitments under the Ocean Plastics Charter. The SEA found that inaction, or limited action, can have significant negative impacts on the environment, including increased stresses on marine life, compromised economic viability of municipal recycling, and increased stress on Canada's limited landfill disposal capacity.

Gender-based analysis plus

The gender-based analysis plus (GBA+) assessment concluded that the proposed Order would not affect sociodemographic groups (based on factors such as gender, sex, age, language, education, geography, culture, ethnicity, income, ability, sexual orientation, or gender identity), as it would not introduce new regulatory requirements.

Implementation, compliance and enforcement, and service standards

As no specific risk management measures are recommended as part of the proposed Order, developing an implementation plan and a compliance and enforcement strategy, as well as establishing service standards, are not necessary at this time. In the event that the ministers propose risk management measures for plastic manufactured items, the departments would assess these elements during the development of such measures.

Coopération et harmonisation en matière de réglementation

Le projet de décret ne serait pas directement lié à des obligations ou à des accords nationaux ou internationaux. Le projet de décret permettrait aux ministres de proposer des mesures de gestion des risques qui pourraient s'harmoniser et compléter les mesures prises par les gouvernements provinciaux et territoriaux et les administrations municipales en vue d'un effort coordonné pour atteindre l'objectif de zéro déchet de plastique et éliminer la pollution plastique d'ici 2030.

Évaluation environnementale stratégique

Conformément à la *Directive du Cabinet sur l'évaluation environnementale des projets de politiques, de plans et de programmes*, une évaluation environnementale stratégique (EES) a été réalisée pour le programme entier du gouvernement du Canada destiné à l'atteinte de l'objectif de zéro déchet de plastique. L'analyse a conclu que la pleine mise en œuvre du programme aura des effets positifs considérables sur l'environnement et la capacité du Canada à mettre en œuvre sa Stratégie fédérale de développement durable, sa Stratégie pancanadienne visant l'atteinte de zéro déchet de plastique et ses engagements liés à la Charte sur les plastiques dans les océans. L'EES a conclu que l'inaction ou les mesures limitées peuvent avoir des effets négatifs importants sur l'environnement, y compris accroître les facteurs de stress sur la vie marine, compromettre la viabilité économique du recyclage par les municipalités et augmenter la pression sur la capacité de stockage limitée des sites d'enfouissement au Canada.

Analyse comparative entre les sexes plus

L'évaluation de l'analyse comparative entre les sexes plus (ACS+) a conclu que le projet de décret n'aurait pas d'incidence sur les groupes sociodémographiques (fondés sur des facteurs comme le genre, le sexe, l'âge, la langue, le niveau d'éducation, la géographie, la culture, l'ethnicité, le revenu, la capacité, l'orientation sexuelle ou l'identité de genre), car celui-ci n'introduirait pas de nouvelles exigences réglementaires.

Mise en œuvre, conformité et application, et normes de service

Comme aucune mesure de gestion des risques précise n'est recommandée dans le cadre du projet de décret, l'élaboration d'un plan de mise en œuvre et d'une stratégie de conformité et d'application ainsi que l'établissement de normes de service ne sont pas nécessaires pour le moment. Advenant le cas où les ministres proposeraient des mesures de gestion des risques pour les articles manufacturés en plastique, les ministères évalueraient ces éléments lors de l'élaboration de ces mesures.

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PROPOSED REGULATORY TEXT

Notice is given, pursuant to subsection 332(1)[a] of the *Canadian Environmental Protection Act, 1999*[b], that the Governor in Council, on the recommendation of the Minister of the Environment and the Minister of Health, pursuant to subsection 90(1) of that Act, proposes to make the annexed *Order Adding a Toxic Substance to Schedule 1 to the Canadian Environmental Protection Act, 1999*.

Any person may, within 60 days after the date of publication of this notice, file with the Minister of the Environment comments with respect to the proposed Order or a notice of objection requesting that a board of review be established under section 333 of that Act

[a] S.C. 2004, c. 15, s. 31
[b] S.C. 1999, c. 33

PROJET DE RÉGLEMENTATION

Avis est donné, conformément au paragraphe 332(1)[a] de la *Loi canadienne sur la protection de l'environnement (1999)*[b], que la gouverneure en conseil, sur recommandation du ministre de l'Environnement et de la ministre de la Santé et en vertu du paragraphe 90(1) de cette loi, se propose de prendre le *Décret d'inscription d'une substance toxique à l'annexe 1 de la Loi canadienne sur la protection de l'environnement (1999)*, ci-après.

Les intéressés peuvent présenter au ministre de l'Environnement, dans les soixante jours suivant la date de publication du présent avis, leurs observations au sujet du projet de décret ou un avis d'opposition motivé demandant la constitution de la commission de

[a] L.C. 2004, ch. 15, art. 31
[b] L.C. 1999, ch. 33

and stating the reasons for the objection. All comments and notices must cite the *Canada Gazette*, Part I, and the date of publication of this notice, and be sent to the Executive Director, Program Development and Engagement Division, Department of the Environment, Gatineau, Quebec K1A 0H3 (fax: 819-938-5212; email: eccc.substances.eccc@canada.ca).

A person who provides information to the Minister of the Environment may submit with the information a request for confidentiality under section 313 of that Act.

Ottawa, October 5, 2020

Julie Adair
Assistant Clerk of the Privy Council

révision prévue à l'article 333 de cette loi. Ils sont priés d'y citer la Partie I de la *Gazette du Canada*, ainsi que la date de publication, et d'envoyer le tout au directeur exécutif, Division de la mobilisation et de l'élaboration de programmes, ministère de l'Environnement, Gatineau (Québec) K1A 0H3 (téléc. : 819-938-5212; courriel : eccc.substances.eccc@canada.ca).

Quiconque fournit des renseignements au ministre de l'Environnement peut en même temps présenter une demande de traitement confidentiel aux termes de l'article 313 de cette loi.

Ottawa, le 5 octobre 2020

La greffière adjointe du Conseil privé
Julie Adair

Order Adding a Toxic Substance to Schedule 1 to the Canadian Environmental Protection Act, 1999

Amendment

1 Schedule 1 to the *Canadian Environmental Protection Act, 1999*[1] is amended by adding the following in numerical order:

163 Plastic manufactured items

Coming into Force

2 This Order comes into force on the day on which it is registered.

Décret d'inscription d'une substance toxique à l'annexe 1 de la Loi canadienne sur la protection de l'environnement (1999)

Modification

1 L'annexe 1 de la *Loi canadienne sur la protection de l'environnement (1999)*[1] est modifiée par adjonction, selon l'ordre numérique, de ce qui suit :

163 Articles manufacturés en plastique

Entrée en vigueur

2 Le présent décret entre en vigueur à la date de son enregistrement.

[1] S.C. 1999, c. 33

[1] L.C. 1999, ch. 33

Government of Canada Gouvernement du Canada

Canada

Science Assessment of Plastic Pollution

Environment and Climate Change Canada
Health Canada

October 2020

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List of abbreviations

Abbreviation	Meaning
µ-FTIR	Micro-Fourier transform infrared spectroscopy
AAB	Adopt-a-Beach™
ABS	Acrylonitrile butadiene styrene
AKT	Protein kinase B
ARG	Antibiotic resistant gene
ATP	Adenosine triphosphate
BALF	Bronchoalveolar lavage fluid
BW	Body weight
CaPSA	Canada's Plastics Science Agenda
CBD	Convention on Biological Diversity
CCME	Canadian Council of Ministers of the Environment
CMC	Carboxymethylcellulose
DDE	Dichlorodiphenyldichloroethylene
DNA	Deoxyribonucleic acid
DW	Dry weight
DWTP	Drinking water treatment plant
EC_{10}	10% effect concentration
EC_{50}	Median effect concentration
ECCC	Environment and Climate Change Canada
ECHA	European Chemicals Agency
EDS	Energy-dispersive X-ray spectroscopy
EFSA	European Food Safety Agency
ERK	Extracellular signal-regulated kinase
EU	European Union
FAO	Food and Agriculture Organization
FTIR	Fourier Transform Infrared Spectroscopy
GCMS	Gas chromatography mass spectrometry
GCSC	Great Canadian Shoreline Cleanup
GD	Gestational day
GESAMP	Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection
GI	Gastrointestinal
GM	Geometric mean
GPx	Glutathione peroxidase
GR	Glutathione reductase
GSH	Glutathione
GST	Glutathione-S-transferase
HDPE	High-density polyethylene
HGT	Horizontal gene transfer
HOC	Hydrophobic organic compound
HPF	Hours post fertilization
IR	Infrared
IUCN	International Union for Conservation of Nature

LC_{50}	Median lethal concentration
LDH	Lactate dehydrogenase
LDPE	Low-density polyethylene
LLDPE	Linear low-density polyethylene
LoD	Limit of detection
LOEC	Lowest observed effect concentration
LOEL	Lowest observed effect level
LT_{50}	Median lethal time
MAPK	Mitogen-activated protein kinase
MEDITS	International bottom trawl survey in the Mediterranean
MEK	Mitogen-activated protein kinase
MGE	Mobile genetic element
MMAD	Mass median aerodynamic diameter
MP	Microplastic
NOEC	No observed effect concentration
NOEL	No observed effect level
OECD	Organisation for Economic Co-operation and Development
PA	Polyamide
PAA	Poly(N-methyl acrylamide)
PAH	Polycyclic aromatic hydrocarbon
PAME	Protection of the Arctic Marine Environment
PAN	Polyacrylonitrile
PBDE	Polybrominated diphenyl ethers
PBT	Polybutylene terephthalate
PC	Polycarbonate
PCB	Polychlorinated biphenyl
PE	Polyethylene
PET	Polyethylene terephthalate
PLA	Polylactic acid
PMMA	Poly(methyl methacrylate)
POP	Persistent organic pollutant
PP	Polypropylene
PS	Polystyrene
PTFE	Polytetrafluoroethylene
PU	Polyurethane
PUF	Polyurethane foam
PVC	Polyvinyl chloride
qPCR	Quantitative polymerase chain reaction
ROS	Reactive oxygen species
SAMP	Suspended atmospheric microplastic
SEM	Scanning electron microscopy
SOD	Superoxide dismutase
TEF	Toxic equivalency factor
UNEP	United Nations Environment Programme
UV	Ultra-violet
WHO	World Health Organization

WW	Wet weight
WWTS	Wastewater treatment system

Executive summary

Plastics are among the most universally used materials in modern society. Since the 1950s, the production and use of plastics has been increasing faster than that of any other material, mostly due to their durability and low cost. However, the improper management of plastic waste has led to plastics becoming ubiquitous in all major compartments of the environment. Plastic that is discarded, disposed of, or abandoned in the environment outside of a managed waste stream is considered plastic pollution. Plastic pollution has been detected on shorelines, and in surface waters, sediment, groundwater, soil, indoor and outdoor air, food and drinking water. In Canada, it is estimated that 1% of plastic waste enters the environment. In 2016, this amounted to 29 000 tonnes of plastic pollution. Since plastic degrades very slowly and is persistent in the environment, the amount of plastic pollution is anticipated to continue to increase over time. There are growing concerns that plastic pollution may adversely impact the health of the environment and humans.

The purpose of this report is to summarize the current state of the science regarding the potential impacts of plastic pollution on the environment and human health, as well as to guide future research and inform decision-making on plastic pollution in Canada. It provides a review of the available information on plastic pollution, including its sources, occurrence, and fate, as well as on the potential effects of plastic pollution on the environment and human health. This report is not intended to quantify the risks of plastic pollution on the environment or human health, but rather to survey the existing state of science in order to guide future scientific and regulatory activities.

In an environmental context, plastics are often categorized by their size, with macroplastics being larger than 5 mm and microplastics being less than or equal to 5 mm. Plastic waste can be released into the environment as complete materials (e.g., discarded single-use or short-lived products, such as plastic bags and straws), as large pieces of plastics (e.g., fragments of plastic products) or as microplastics (e.g., microfibres released from washing of clothes or microbeads released through wastewater). Microplastics can also be formed through the breakdown of larger plastic items in the environment.

While plastics can degrade, the rate at which they break down is slow and can be affected by multiple factors, such as temperature and light. In water, the rate of degradation is temperature dependent, being slower in cold water. The lack of exposure to sunlight also slows down the degradation of plastics. While oxidation can promote the degradation of plastics in soil, the rate of degradation is still slow. Although biodegradable, compostable, biobased, and oxo-degradable plastics are increasingly being used as alternatives to conventional plastics, there is a lack of significant evidence that they will fully degrade in natural environments. Further studies would assist in understanding their environmental impact, particularly in comparison to conventional plastics.

Plastic pollution is found in aquatic and terrestrial environments, as well as in indoor and outdoor air, and arises from various sources. For example, plastic may enter the aquatic environment as a result of litter, mismanaged waste, and abandoned, lost or discarded fishing gear, or may be deposited in the terrestrial environment from agricultural activities. Additionally, microplastics removed from wastewater settle in sewage sludge can be released to land through the application of biosolids.

Moreover, release of microfibres from wastewater treatment systems is known to represent a source of microplastic pollution. Products available to consumers that are discarded to the environment or not properly managed may also represent a source of plastic pollution. Sources of microplastic pollution to indoor air include the shedding of fibres from clothing, furniture, carpeting and household goods, while microplastics in outdoor air are influenced by various sources including vehicle tire wear and tear.

In Canada as well as internationally, single-use plastics make up the bulk of macroplastics found on shorelines. The most common litter items collected on Canadian shorelines include cigarette butts, bottle caps, plastic bags, plastic bottles, and straws. Microplastic particles such as fragments and pellets are also found on shorelines where they accumulate within the organic matter along the strandline. Generally, a greater abundance of plastic pollution has been found in areas with high human and industrial activity, notably in the Great Lakes.

Microplastic particles are also found in fresh and marine surface waters. Globally, microfibres are one of the most common types of microplastics found in water. However, it is recognized that there is a lack of standardized, high-quality methods for sampling plastics, particularly for measuring, quantifying and characterizing microplastics.

Microplastics are also found in sediment and soil. Through various mechanisms, such as the formation of biofilms—layers of microorganisms that form on a surface—microplastics in surface waters may eventually sink, leading to the accumulation of microplastics in the sediment of both freshwater and marine environments. Soils are also expected to act as a major sink for plastic particles, as microplastics are likely to remain in soils for long periods due to factors such as vertical transport, which pulls particles down from the surface, slowing their degradation. Currently, only limited evidence is available on the occurrence of microplastics in groundwater, although it has been hypothesized that microplastics may travel from soil into groundwater.

Air is also anticipated to be an important pathway for microplastic transport, and microplastics have been detected in both indoor and outdoor air. While there are no Canadian data available on the occurrence of microplastics in air, limited data from other parts of the world suggest that concentrations may be higher in indoor air than in outdoor air. Indoors, microplastics are also found in settled house dust.

Current data on the occurrence of microplastics in food are limited, and most available information concerns microplastics found in seafood, specifically fish and shellfish from marine environments. In fish, microplastics have been found in both muscle tissue and the gastrointestinal tract, mostly as fragments and fibres. Microplastics have also been detected in mussels, clams, oysters, scallops and snails, and in a very small number of other foods, such as salt.

Internationally, a limited number of studies have investigated the presence of microplastics in tap and bottled water. Of these studies, few are considered reliable due to concerns with quality assurance measures. While the available studies indicate that microplastics have been detected in bottled water samples purchased from outside of Canada, the concentrations do not correlate with bottle type (i.e., plastic, glass or beverage carton) and vary depending on the use conditions (i.e., single-use versus multi-

use bottles). The sources of microplastics in bottled water are still unknown and further research is required. In the case of tap water, some studies have detected microplastics while others have not. It is anticipated that drinking water treatment will remove a large proportion of microplastic particles.

Plastic pollution has been shown to impact organisms and their habitats. Macroplastics have been demonstrated to cause physical harm to environmental receptors on an individual level and to have the potential to adversely affect habitat integrity. Physical harm to biota is often a result of entanglement or ingestion. Entanglement can lead to suffocation, strangulation, or smothering, and a high frequency of reported entanglements have led to direct harm or mortality. Ingestion can lead to direct harm through physical damage; it can block airways or intestinal systems leading to suffocation or starvation. An increasing amount of plastic pollution in water bodies may also adversely affect ecosystem function, biodiversity, and habitat integrity. Plastics can act as transport mediums for organisms, microorganisms, or other organic matter, which can alter ecosystem dynamics.

The observed effects of microplastics on biota are primarily driven by physical effects. Published studies on the ecotoxicological effects of microplastics report conflicting observations, even for the same endpoint in the same species. This conflicting information could be addressed by developing and using standard approaches for testing the effects of microplastics on environmental organisms, using environmentally relevant testing materials, and developing an understanding of the impact of shape, size and chemical composition on ecotoxicological effects.

Humans may be exposed to microplastics via the ingestion of food, bottled water, and tap water, as well as through the inhalation of indoor and outdoor air. However, information on the human health effects of microplastics is limited, and further research is required to better inform target tissues, threshold doses, and mode of action. Some associations between exposures to high levels of microplastics and adverse health effects in laboratory animals and in humans have been reported, but these health effects cannot be linked to exposure in the general population. Occupational inhalation exposure studies show associations between work in microplastic-related industries and increased incidence of various respiratory symptoms and diseases. Conflicting observations have been made for cancers of the respiratory tract and digestive system.

Effects observed in animal studies are primarily associated with tissues related to where particles enter the body (e.g., effects on the digestive system after oral exposure and on the respiratory tract after inhalation). Effects following oral exposure include inflammation of the liver, oxidative stress, metabolic changes, and altered gut microbiota. Movement of a small fraction of microplastic particles to lymphatic tissues has also been observed. Although the current scientific literature does not identify a concern for human health, there are insufficient data to allow for a robust evaluation of the potential human health risks of ingested microplastics.

Effects in the respiratory tract are likely related to the physical impact of microplastics as particulate matter and include oxidative stress, cytotoxicity, inflammation, and development of foreign body granulomas. In inhalation studies, movement of a small fraction of microplastic particles to lymphatic or

systemic tissues has also been observed. No dose-response relationship has been observed in mortality, survival time, behaviour, clinical observations, or tumour incidence from inhalation exposures.

In addition to physical impacts, there are concerns that plastics may serve as a means of transport for other chemicals. Since plastics can contain unbound monomers and chemical additives and can sorb persistent organic pollutants from the environment, it is possible that these substances may be transported to organisms or humans, where they may then be released. The extent of release is expected to depend on a variety of factors, such as the properties of the receiving environment, the plastic particle, and the bound chemical. The current literature suggests that, while the transport of chemicals via plastics is possible, the impact to biota is likely limited, and recent international reviews indicate that there is likely a low health concern for human exposure to chemicals from ingestion of microplastics from food or drinking water (EFSA 2016; FAO 2017; WHO 2019). Many of the chemicals observed to be bound to plastic particles have been assessed by various programs at Environment and Climate Change Canada and Health Canada.

Plastics can also provide a habitat for microorganisms, including potential pathogens, through the formation of biofilms. There is currently no indication that microplastic-associated biofilms would impact human health. In addition, despite very limited data, it is anticipated that drinking water treatment would inactivate biofilm-associated microorganisms.

In order to advance the understanding of the impacts of plastic pollution on the environment and human health, it is recommended that research be carried out in the following areas to address the key knowledge gaps identified in this report:

- Developing standardized methods for sampling, quantifying, characterizing, and evaluating the effects of macroplastics and microplastics;
- Furthering the understanding of human exposure to microplastics;
- Furthering the understanding of the ecotoxicological effects of microplastics;
- Furthering the understanding of the effects of microplastics on human health; and
- Expanding and developing consistent monitoring efforts to include poorly characterized environmental compartments such as soil.

Given the increasing amounts of plastic pollution in the environment and the demonstrated ability of macroplastics to harm biota, it is anticipated that the frequency of occurrence of physical effects on individual environmental receptors will continue to increase if current trends continue without mitigation measures.

In accordance with the precautionary principle, action is needed to reduce macroplastics and microplastics that end up in the environment.

1. Introduction

Plastics are part of the everyday lives of Canadians and populations around the world. Plastics are low cost, durable materials and can be used in a variety of applications (CCME 2018). For these reasons, global plastic production has been increasing over the past several decades at a rate faster than that of any other material (Geyer et al. 2017; CCME 2018). In Canada, total sales of plastic were estimated to be $35 billion in 2017, with approximately 4 667 kt introduced to the Canadian market in 2016 (ECCC 2019a). Plastics are used in a variety of industrial sectors, and demand for plastic products continues to grow.

Of the 4 667 kt of plastics that entered the Canadian market in 2016, an estimated 3 268 kt were discarded as waste (ECCC 2019a). Of that plastic waste, an estimated 29 kt (or 1%) were discarded outside of the normal waste stream (i.e., not landfilled, recycled or incinerated) in 2016, through direct release to the environment or through dumps or leaks. An estimated 9% of the remaining plastic waste was recycled, 86% was landfilled, and 4% was incinerated for energy recovery (ECCC 2019a).

In a global context, Geyer et al. (2017) have estimated that only 30% (2 500 000 kt) of all plastics ever produced are still in use. This means that 6 300 000 kt of global cumulative plastic waste was created between 1950 and 2015. As illustrated in Figure 1, if plastic manufacturing continues at its current pace, the accumulation of plastics will continue to accelerate. It is estimated that by 2050, 12 000 000 kt of plastic waste will have been discarded globally to landfills or the environment (Geyer et al. 2017).

With the growing public and scientific concern about the ubiquity of plastic pollution, there has been increasing global media attention on the potential impacts of plastic pollution on human health and the environment (CCME 2018; ECCC 2019b; SAPEA 2019). The Government of Canada has put forward Canada's Plastics Science Agenda (CaPSA), which aims to align current and future research investments across a range of disciplines (ECCC 2019b). The CaPSA framework identifies several key research priorities, including the detection of plastics in the environment, understanding and mitigating potential impacts on wildlife, human health and the environment, plastic design and alternatives, sustainable plastic production, and recycling and recovery.

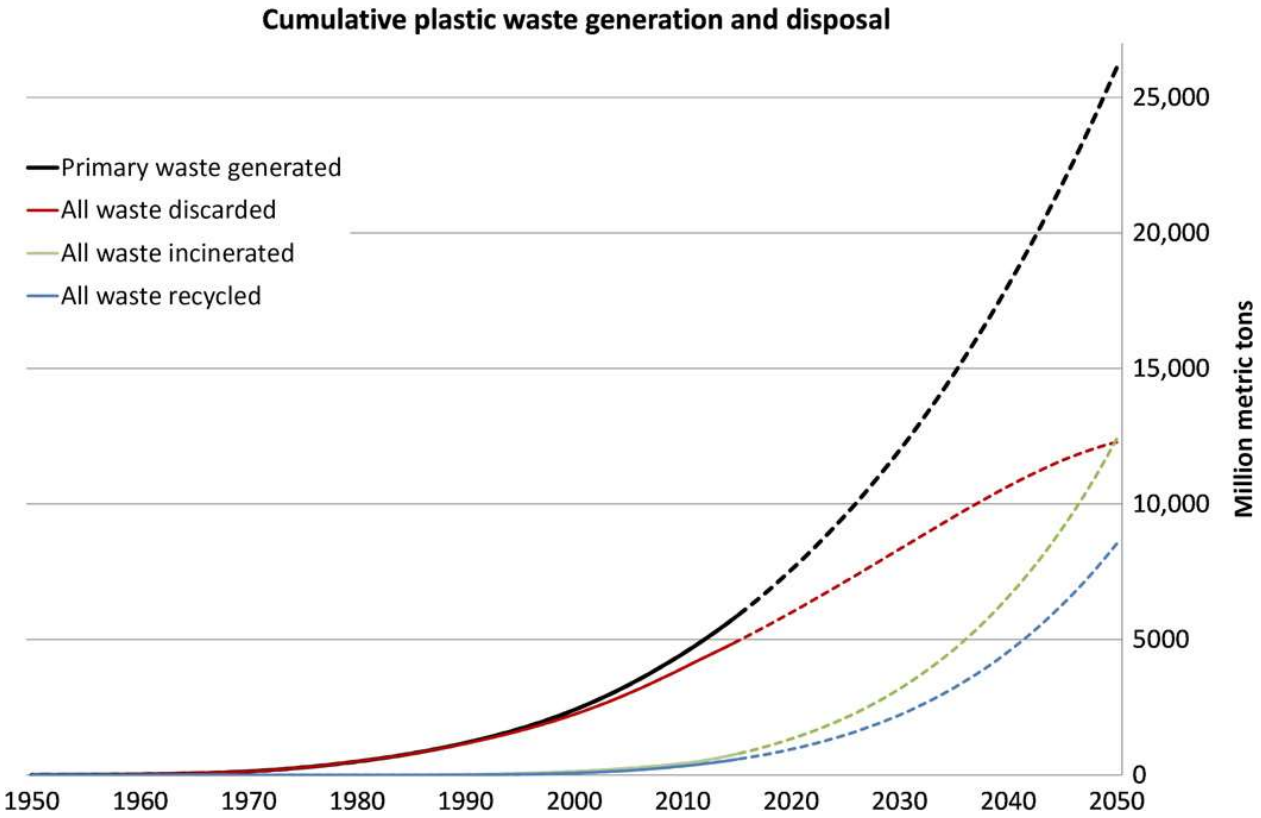

Figure 1: Global cumulative plastic waste generation and disposal. Solid lines show historical data from 1950 to 2015, and dashed lines show projections of historical trends to 2050 (reproduced with permission from Figure 3 of Geyer et al. 2017)

Long Form Description: This figure shows the trend in global cumulative plastic waste generation and disposal. Increasing trend lines are shown for primary waste generated, all waste discarded, all waste incinerated and all waste recycled (reproduced with permission from Figure 3 of Geyer et al. 2017)

1.1 Scope

This report summarizes the current state of the science on the potential impacts of plastic pollution on the environment and human health and informs future research and decision-making on plastic pollution in Canada. For the purposes of this report, plastic pollution has been divided into two main types: macroplastics (plastics greater than 5 mm in size) and microplastics (plastics less than or equal to 5 mm in size). This report discusses the sources, occurrence, and fate of plastic pollution in the environment, as well as the potential impacts of plastics on human health and the environment. In the draft science assessment, published in January 2020, information identified up to June 2019 was considered for inclusion, in addition to the August 2019 World Health Organization (WHO) report on microplastics in drinking water and the October 2019 Ocean Wise report on microfibres (Vassilenko et al. 2019). In finalizing the science assessment, a review of information published up to March 2020 was conducted (see Appendix A and Section 5.1.1 for relevant new information), in addition to considering comments submitted through public consultation.

This report is a review of the current state of the science on plastic pollution. It is not intended as a substitute for chemical risk assessment, and it is similar to the approach taken for the Science Summary on Microbeads (ECCC 2015). Typically, a chemical risk assessment is conducted to assess the potential for risk to the environment and human health associated with a substance. However, significant data gaps currently exist that preclude the ability to conduct a quantitative risk assessment, including a lack of standardized methods for monitoring microplastics and characterizing the environmental and human health effects of plastic pollution, as well as inconsistencies in the reporting of occurrence and effects data in the scientific literature (Gouin et al. 2019). Indeed, risk assessment frameworks for evaluating the potential risks associated with plastic pollution are currently under development. For example, see Gouin et al. (2019) for a discussion on the development of an environmental risk assessment framework for microplastics.

As the focus of this report is on plastic pollution, it is limited to a review of the occurrence of macroplastics and microplastics resulting from plastic waste entering the environment and does not examine non-environmental sources (e.g., via direct exposure from products available to consumers or self-care products[1]). Moreover, it does not review the economics of waste management practices or evaluate the efficacy of waste management streams (e.g., recycling).

1.2 Terminology

This report discusses plastic pollution in an environmental context. In this context, plastics are often categorized by their size. The term microplastic was originally used to differentiate between substances that could only be visualized through a microscope and larger macroplastics (ECCC 2015). However, there is no single definition of what constitutes a microplastic. For the purpose of this report, plastic particles less than or equal to 5 mm in size are defined as microplastics, while plastics greater than 5 mm are defined as macroplastics. Microplastics can be further defined as primary or secondary microplastics. Primary microplastics are intentionally produced plastic particles (such as pellets, powders, and beads) that are either intended for use as microplastics or as precursors for the production of plastic or plastic-containing products. Primary microplastics are widely used as abrasives in a variety of applications (UNEP 2016). Secondary microplastics are not produced intentionally, but are the result of the breakdown and fragmentation of larger plastic items (SAPEA 2019). Furthermore, microfibres are a specific type of secondary microplastic defined as being fibrous in shape and less than or equal to 5 mm in length. Nanoplastics are considered to be a subset of microplastics. They are primary or secondary microplastics that range from 1 to 100 nm in size in at least one dimension. Nanoplastics occur largely as a result of secondary sources of plastic pollution (i.e., the breakdown of larger plastics) (Rist and Hartmann 2018). This report will focus on plastics greater than 100 nm in size (i.e., microplastics and macroplastics).

[1] Self-care products are products available for purchase without a prescription from a doctor, and fall into one of three broad categories: cosmetics, natural health products, and non-prescription drugs.

In this report, plastic waste is considered to be plastics that enter the waste stream (e.g., landfilled, recycled or incinerated), whereas plastic pollution is considered to be plastic that is discarded, disposed of, or abandoned in the environment outside of a managed waste stream. In the scientific literature, plastic pollution has been referred to by a number of terms, such as plastic debris or plastic litter. This report will use the terms plastic pollution or plastic pollutants. Furthermore, in this report the term litter refers to any persistent, manufactured, or processed solid material discarded, disposed of, lost, or abandoned in the environment, including plastics, textiles, glass, metal, ceramics, and other persistent synthetic materials. This term will be used when the proportion of plastic pollution reported in the literature is unclear.

2. Composition, properties, and uses

All plastic materials are formed from long-chain polymers of very high molecular weight, often measured in the hundreds of thousands of kilodaltons (Sperling 2006). Synthetic polymers first appeared in the early 20th century, leading to the manufacture of plastic products such as Bakelite and nylon (commercial name for polyamides). Since then, polymer science has evolved, with a greater mechanistic understanding of the interrelationships between polymer structure, morphology, and physical and mechanical behaviour. This has resulted in the production of a myriad of plastic materials with varying physical and chemical properties.

Polymerization, the synthesis of polymers, can occur following one of two main processes: chain polymerization or stepwise polymerization. The process used to form polymers greatly influences their physical properties. Common chain polymer structures include polyethylene (PE), polypropylene (PP), polystyrene (PS), and polyvinyl chloride (PVC), whereas common stepwise polymers include nylons, polyethylene terephthalate (PET), polycarbonate (PC), and polyurethane (PU). These represent many of the most common forms of plastics typically found as environmental contaminants (Sperling 2006).

The physical properties of plastic, such as rigidity, flexibility, and elasticity, are influenced by the polymer's molecular weight distribution and organization of polymer chains (Sperling 2006; Verschoor 2015). Generally, high molecular-weight polymers with a complex organization that leads to strong covalent bonds between the polymers can result in the formation of a rigid plastic with a high melting point. In contrast, linear polymer organization with low molecular-weight distribution results in a more flexible plastic with a lower melting point. Combinations of different molecular weight distributions, different polymer chain organization, and/or blends of different types of polymers can produce a material that will be effective for its intended use (Sperling 2006).

Furthermore, many polymers are subject to additional processes aimed at enhancing efficacy with respect to an intended functionality. For instance, when heated, a linear polymer will flow, resulting in the formation of a thermoplastic (Sperling 2006). Thermoplastics are polymers commonly found in plastics that can be melted and reshaped into new objects. Commonly used thermoplastics include PVC, PE, PS, and PC (ECCC 2019c). To prevent flow upon heating, polymers can be cross-linked to produce a thermoset plastic (Sperling 2006). Thermoset plastics are polymers that are used for their resistance to

mechanical forces, chemicals, wear, and heat, but they cannot be re-melted to form new objects. Examples include PU and unsaturated polyester polymers (ECCC 2019c).

Chemical additives can be added to polymers during production to alter the properties of plastics (Rochman et al. 2019). There are several categories of additives, including stabilizers and functional agents. Polymer stabilizers maintain the inherent properties of the material by protecting it against oxidative degradation. They include substances such as anti-oxidants, light stabilizers, metal deactivators, and ultra-violet absorbers. Functional agents can enhance the mechanical strength of a polymer or impart new characteristics. Examples of functional agents include flame-retardants, anti-static agents, lubricants, and plasticizers (ECCC 2019c). For instance, plasticizers can be added to soften a polymer by lowering its glass transition temperature or reducing the degree of crystallinity or melting point (Sperling 2006).

Polymer production can begin with either the use of recycled or recovered plastics or with natural resources (i.e., petroleum or plant-based starting material). These polymers are then used to manufacture plastic products (ECCC 2019c). Although many different types of plastic polymers are used in Canada, domestic plastic production is dominated by five polymer types. PE accounts for the majority of plastic production, with 3 700 kt produced in 2017, followed by PVC (210 kt), PET (166 kt), PU (122 kt) and polyamides (PA) (116 kt) (ECCC 2019c). Of the 4 800 kt of plastic polymers produced in Canada in 2016, 77% was exported. Further, there is a domestic demand of 3 800 kt, 71% of which is fulfilled through imports (ECCC 2019a).

The majority of plastic products in Canada are found in the packaging and construction sectors. Other major sectors include the automotive, electronic and electrical equipment, textiles, and agriculture sectors (ECCC 2019c). Examples of applications of various polymers are presented in Table 2-1.

Table 2-1: *Selected polymer applications*

Acronym	Name	Main application[a]
PP	Polypropylene	Rigid, semi-rigid and flexible packaging Automotive Housewares Electrical insulation
PE	Polyethylene	Rigid, semi-rigid and flexible packaging Agricultural film Housewares Electrical insulation Construction (pipes) Self-care products
PS	Polystyrene	Packaging (thermoformed containers) Foams
PMMA	Poly(methyl methacrylate)	Transparent applications in the automotive and construction industries Medical Electronics
PC	Polycarbonate	Transparent applications in the automotive and construction industries Medical Electronics
PLA	Polylactide – a specific type of polyester	Rigid, semi-rigid and flexible packaging
PET	Polyethylene terephthalate – a specific type of polyester	Rigid, semi-rigid and flexible packaging Textile synthetic fibres
PVC	Polyvinylchloride	Construction (pipes, profiles, flooring) Sheet and coated fabrics Electrical insulation
PTFE	Polytetrafluoroethylene	Anti-adhesive coatings Engineering parts

[a] Personal communication, email from the Transportation and Manufacturing Division, National Research Council Canada, to the Ecological Assessment Division, Environment and Climate Change Canada, dated August 15, 2019; unreferenced

The packaging sector is the largest user of plastics in Canada, accounting for 33% of end-use plastics introduced to the market in 2016. Examples of plastic packaging products include plastic bags, water and soft drink bottles, as well as various packaging used for pharmaceuticals, toiletries, and cleaning compounds. PE is very commonly used in packaging, specifically for films and flexible packaging. The main types of PE are low-density polyethylene (LDPE), linear low-density polyethylene (LLDPE), and high-density polyethylene (HDPE). These types of PE form the majority of PE production globally. Some examples of LDPE/LLDPE applications are squeeze bottles, toys, carrier bags, and general packaging. LDPE is generally used in heavier duty films, such as high durability bags and protective sheeting, due to its toughness, flexibility, and relative transparency. HDPE possesses good chemical resistance and is used for packaging many household and industrial chemicals such as detergents and bleach. It is also used in thin-gauge carrier bags, chemical drums, toys, food wrapping material, and kitchenware. In

addition to PE, other plastic polymers are used in the packaging sector, such as PVC, PET and PP (ECCC 2019c).

Construction is the second-largest end-use market for plastics in Canada, accounting for approximately 26% of all end-use plastics generated in 2016 (ECCC 2019a). Primary uses of plastics in the construction sector include plastic and foam building and construction materials, paints and coatings, profile shapes, and reconstituted wood and plywood. Plastics are broadly used in the construction of all types of buildings and are especially used in thermal insulation materials, as well as waterproofing and sealant materials. PVC is widely used in siding and window applications, floor and wall covering products, as well as pipe and pipe fittings. Clear PC sheets are used as a substitute for glass in greenhouses, transit shelters, and covered walkways due to its resistance to weathering. PU foam is used as insulation in commercial and residential properties (ECCC 2019c).

In an effort to improve fuel efficiency through weight reduction, the automotive sector has increased its use of plastics. While many different types of plastics are used in the sector, PU, PP, and PVC make up the vast majority of total plastics used in a vehicle. PU is used in cushioning applications such as seating, PP is used in automobile interiors, and PVC is used for faux leather. PC can be used to replace glass in cars, while foam, plastic, and fibre composites can be used in door panels, dashboards, and hoods (ECCC 2019c). In this report, vehicle tires are considered a source of plastic pollution via the release of tire wear particles. Yet, it is recognized that whether or not rubber is considered a type of plastic is the subject of some debate at this time, and while some scientific publications consider rubber to be plastic, others do not.

Other end-use sectors include the electronic and electrical equipment, textile, and agriculture sectors. Plastics are used in the electronics sector for computer and phone parts, as well as items such as electrical wires and cables. The textile sector uses plastics for fibres in carpets, rugs, mats, and clothing. In the agriculture sector, plastics are used for fertilizer and pesticide packaging (e.g., agricultural films, mulches, and greenhouses) (Ekebafe et al. 2011; ECCC 2019c).

Given the variety of plastic materials that can be produced, the physical and chemical properties of plastic particles present in the environment will be complex (Rochman et al. 2019). With respect to shape and size, primary microplastics are intentionally engineered to be a particular size (e.g., virgin resin pellets used in plastic manufacturing processes) and will therefore likely show less variation than secondary microplastics. Secondary microplastics can have a range of shapes, including spheres and cylinders, but also fragments, fibres, and films (Kooi and Koelmans 2019). Secondary microplastics are also highly variable in size and density. Recognizing the inherent challenge associated with defining the physical properties of microplastic particles observed in the environment, Kooi and Koelmans (2019) suggest a method aimed at defining and characterizing the distributions of properties most commonly encountered. The approach proposed by Kooi and Koelmans (2019) may prove useful in developing tools for monitoring plastics in the environment, providing a greater mechanistic understanding of the environmental fate of microplastics, and allowing for easy comparison between studies.

Microplastics can exist as fibres, fragments, spheres, pellets, films, and foams, as shown in Figure 2. In general, certain shapes of microplastics originate from certain plastic products. For example, fibres are typically shed from fabrics, such as clothing and upholstery, whereas pellets are typically from industrial feedstock (Rochman et al. 2019).

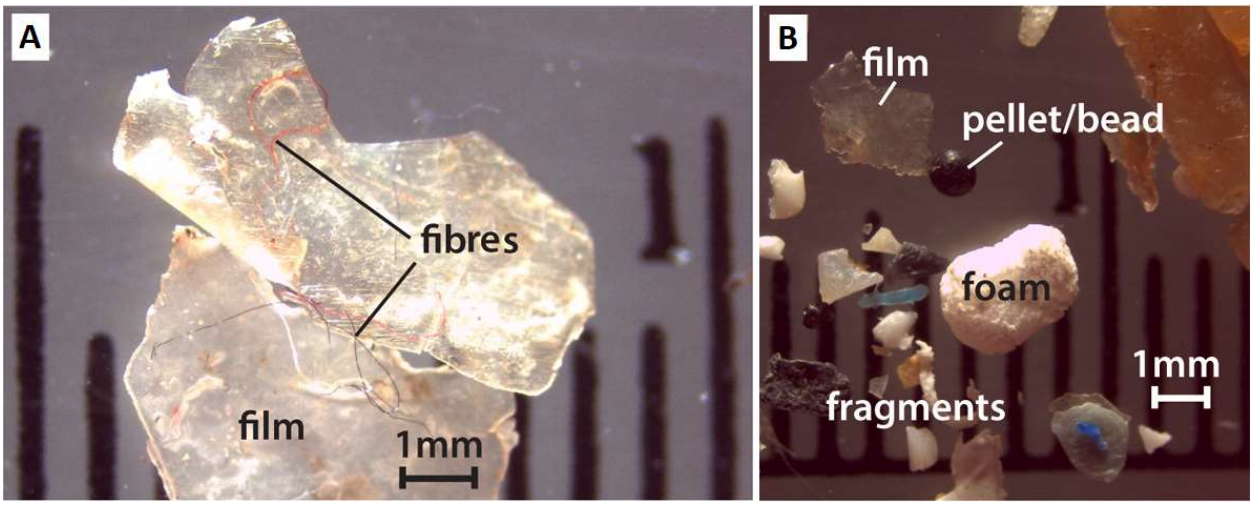

Figure 2: Microplastics found in the environment (reproduced and adapted with permission from Figure 2 of Baldwin et al. 2016)

Long Form Description: This figure shows microplastics found in the environment. Image A contains film and fibres. Image B contains film, pellet/bead, fragments, and foam. (Reproduced and adapted with permission from Figure 2 of Baldwin et al. (2016))

Density is a key property that influences the environmental fate of plastics (Rochman et al. 2019). Densities of plastic polymers such as PE, PS and PVC can range from 0.9 to 2.3 g/cm^3 (WHO 2019).

Table **2-2** presents the densities of various plastic polymers. Polymers with a density greater than 1 are denser than water and are expected to sink, while those with a density less than 1 are expected to float. By analogy, the environmental fate and transport of macroplastics or microplastics released to the atmosphere are also likely to be influenced by their density. For example, denser microplastics are less likely to be readily dispersed by the wind (Rochman et al. 2019). The density of plastics and their buoyancy in water can also be influenced by the coating of plastics with microorganisms, algae, or plants (i.e., biofilms) (Woodall et al. 2014). Other factors, such as shape and size, can also govern the fate of plastics in the environment (Rochman et al. 2019).

Table 2-2: *Selected polymer densities*

Name	Density (g/cm^3)[a]
Polyethylene	0.965 – 0.971
Polypropylene	0.90 – 0.91
Polystyrene	1.04 – 1.10
Polyamides (nylon)	1.02 – 1.05
Acrylic	1.09 – 1.20
Polyvinylchloride	1.16 – 1.58
Poly methylacrylate	1.17 – 1.20
Polyurethane	1.20
Polyester	1.23 – 2.3
Polyethylene terephthalate – a specific type of polyester	1.37 – 1.45

[a] Hidalgo-Ruz et al. 2012

3. Sources of plastic waste and pollution

3.1 Sources of plastic waste

In Canada, the main industrial sectors contributing to the estimated 3 268 kt of plastic waste discarded in 2016 are presented in **Error! Reference source not found.**. Plastic packaging is the single largest contributor of plastic waste, followed by the automotive, textile and electrical and electronic equipment sectors. In 2016, 33% of the plastics entering the Canadian marketplace was for use in packaging. Based on international data, it is estimated that 40% of all plastic production is used for packaging, a significant portion of which is used for the food and drink sector (UNEP 2016). Due to the extremely short lifecycle of plastics from packaging (i.e., most plastic packaging is single-use in nature) compared to plastics from other sectors, packaging accounted for 47% of the plastics discarded in Canada in 2016. Plastics generated from other industrial sectors, such as the automotive (vehicle parts and components, excluding tire wear) and construction sectors, have longer lifecycles and therefore represent a smaller proportion of annual plastic waste as compared to packaging, which is typically discarded shortly after use (ECCC 2019a).

Table 3-1: *Main industrial sectors generating plastic waste in Canada in 2016 (ECCC 2019a)*

Sector	Proportion of total plastic waste
Packaging[a]	47%
Automotive (vehicle parts and components, excluding tire wear)	9%
Textiles	7%
Electrical and electronic equipment	7%
Construction	5%
White goods (e.g., large and small appliances)	4%
Agriculture	1%
Other[b]	19%

[a] Films (including plastic bags), bottles and other items from sectors including food and beverage, healthcare, consumer packaged goods, and cosmetics and personal care, among countless other applications.
[b] Includes chemical products, toys, household furniture, etc. See ECCC (2019a) for a complete description.

3.2 Sources of plastic pollution

The sources of global plastic pollution are varied, and actual quantities of plastic pollution are largely unknown (UNEP 2016). Plastics that are discarded to the environment or not properly managed represent sources of plastic pollution. Land-based sources of macroplastics to the marine environment include packaging, construction materials, household goods, and items related to coastal tourism (UNEP 2016). Land-based sources of microplastics around the world include cosmetics and personal care products, synthetic textiles and clothing, terrestrial transport (i.e., tire wear), and plastic producers and fabricators (i.e., accidental loss of resin pellets) (UNEP 2016; SAPEA 2019). With respect to sea-based sources of plastic pollution, the fisheries, aquaculture, and shipping sectors are major contributors (GESAMP 2016; UNEP 2016). Plastics in these sectors may be lost at sea by accident, abandonment, or deliberate disposal (UNEP 2016; SAPEA 2019). Macroplastics and microplastics from land- and sea-based sources can enter the ocean through various entry points (e.g., wastewater, rivers, coasts), depending on the region (UNEP 2016).

Products available to consumers discarded to the environment or not properly managed may also represent a source of plastic pollution in the environment. While knowledge of the source of primary microplastics (i.e., the type and amount of microplastics intentionally used in products available to consumers) in Canada is limited, secondary microplastics may arise from the breakdown and fragmentation of macroplastics released to the environment. This may include items such as toys, plastic gloves, appliances, electronics, mattress covers and flooring, as well as plastic materials used in packaging (**Error! Reference source not found.**).

3.2.1 Sources to water

Plastic pollution in the aquatic environment can arise from plastics released during land-based activities (e.g., through littering, inadequate waste management, landfill leachate, the use of plastics in agriculture, land application of biosolids, or direct release following abrasion or maintenance of plastic products (Boucher and Friot 2017; Alimi et al. 2018), from the deposition of airborne microplastics onto water (Hendrickson et al. 2018), from runoff and stormwater (Grbić et al. 2020)), or from water-based sources (e.g., fishing-related litter (Driedger et al. 2015)). Plastic pollution in water may also arise from the accidental release of raw plastic materials, such as spillage during transport (Driedger et al. 2015) and from releases from wastewater effluent (Murphy et al. 2016; Boucher and Friot 2017; Kay et al. 2018).

The Arctic Council's Protection of the Arctic Marine Environment (PAME) Working Group recently released the *Desktop Study on Marine Litter including Microplastics in the Arctic* as part of the first phase of a Marine Litter Project. The major sectors highlighted as sources of marine litter in the Arctic were fisheries, aquaculture, shipping, cruise tourism, and offshore resource exploration and exploitation. It is estimated that approximately 640 kt of abandoned, lost or discarded fishing gear are released to marine

waters globally each year, accounting for 10% of all marine litter. In addition, releases from communities that are not connected to large waste management systems have been flagged as sources of marine litter (PAME 2019).

Wastewater treatment

When wastewater containing plastics from domestic, commercial, and industrial sources passes through wastewater treatment systems[2] (WWTSs), most of the plastics are removed prior to discharge to the aquatic environment.

Based on a review of several published studies, Sun et al. (2019) reported significant reductions in microplastic concentrations when comparing influent and effluent in various WWTSs: concentrations ranged from 1 to 10 044 particles/L for influent and from 0 to 447 particles/L for effluent. While large variations in microplastic concentrations can be observed between WWTSs, this may be due to differences in sample collection and analysis methods, as there are currently no standardized methods for the detection and quantification of microplastics in water. Other factors, such as catchment size, population served, wastewater source (residential, commercial, or industrial), and treatment technology, may also contribute to variations in influent and effluent concentrations and treatment efficiencies (Sun et al. 2019).

According to available data on the microplastic removal efficiencies of WWTSs, standard wastewater treatment systems using primary and secondary treatment processes can effectively remove most microplastics from the effluent before it is released to receiving waters (WHO 2019). Sun et al. (2019) estimated that 50% to 98% of microplastics can be removed during primary treatment, which involves skimming processes and settling stages, with larger particles being preferentially removed. Secondary treatment, which typically involves biological treatment to remove organic compounds, can increase microplastic removal to approximately 86% to 99.8% of microplastics (Sun et al. 2019; Raju et al. 2018). The addition of tertiary treatment can lead to the removal of 98% to 99.8% of microplastics, but removal efficiency is dependent on the type of treatment technology used (Sun et al. 2019). Advanced technologies such as rapid-sand filters, membrane bioreactors, and dissolved-air flotation can remove 95% to 99.9% of microplastics greater than 20 µm (Lares et al. 2018; Talvitie et al. 2017). Mintenig et al. (2019) observed complete removal of microplastics greater than 500 µm and 95% of microplastics less than 500 µm using tertiary filtration.

Given the large volumes of effluent water leaving a WWTS, even a small fraction of microplastics remaining in the effluent water after treatment can translate into high absolute numbers of particles being released to the environment. Effluent discharges have therefore been identified as an important pathway for the entry of microplastics into freshwater sources (Murphy et al. 2016).

[2] The term “wastewater treatment system” refers to a system that collects domestic, commercial and/or institutional household sewage and possibly industrial wastewater (following discharge to the sewer), typically for treatment and eventual discharge to the environment. Unless otherwise stated, the term wastewater treatment system makes no distinction of ownership or operator type (municipal, provincial, federal, Indigenous, private, partnerships).

It is estimated that a single WWTS discharges an average of 2 million microplastic particles per day (Sun et al. 2019). In a study conducted at a WWTS near Vancouver, it was estimated that 32 million to 97 million microplastics per day are discharged in effluent (Gies et al. 2018), with fibres and fragments being the most abundant microplastic in the effluent. The study also estimated that of the 1.76 trillion microplastic particles that enter the WWTS each year, 1.28 trillion settle into primary sludge, 360 billion exit in secondary sludge, and 30 billion pass into the secondary treatment effluent and are released into the environment, corresponding to up to 99% removal of microplastics in the WWTS.

The most frequent polymers in WWTS influent and effluent are polyester, PE, PET and PA, with fibres accounting for approximately 52.7% of the microplastics found in wastewater, which is likely attributable to the large amount of fibres released during domestic laundering (Sun et al. 2019). A study conducted by the Swedish Environmental Research Institute found that microfibres were the predominant type of microplastics found in sewage sludge from WWTSs, which is consistent with observations in other studies (Magnusson and Norén 2014; Mahon et al. 2017; Li et al. 2018a).

Microfibres from laundering of textiles also represent a significant source to waterbodies. A report by Ocean Wise detailed the results of a study in which 38 different textile samples were tested for their shedding properties using a custom-designed washing machine test facility. The extent of microfibre shedding varied with the type of textile, with polyester, wool and cotton textiles releasing the largest amounts of microfibres. The report also estimates that the average Canadian household releases 533 million microfibres from laundry every year and that an estimated 878 tonnes of microfibres are released to water following wastewater treatment in Canada and the United States annually (Vassilenko et al. 2019).

Synthetic textiles and clothing are a large source of microplastic pollution (SAPEA 2019). Microfibres can be released from synthetic fabrics during wear and laundering, as well as from sources such as fishing gear (e.g., fishing nets) (ECCC 2019d). Carney Almroth et al. (2018) and De Falco et al. (2018) counted the number of microfibres released from different types of fabric under different laundering conditions. Both studies found that the use of a detergent increases the number of fibres released during washing. Powdered detergents, which often contain insoluble compounds that are able to create friction with the fabric, enable an even greater number of fibres to be released (De Falco et al. 2018). It has also been noted that powdered detergents have a higher pH compared to liquid detergents. While this is effective for soil removal, it can damage polyester fabrics by way of slow surface hydrolysis (Bishop 1995). Furthermore, exposure of fabrics to chemical detergents can cause the breakdown of synthetic fibres into smaller fibres (SAPEA 2019). The studies found that fleece garments and tightly knit fabrics released the greatest number of fibres during washing. It was found that on average, an adult-sized PET fleece garment releases an estimated 110 000 fibres during washing (Carney Almroth et al. 2018). A wash load of 5 kg of polyester garments was found to release 6 000 000 to 17 700 000 fibres, for an approximate weight of 0.43 to 1.27 g (De Falco et al. 2018).

3.2.2 Sources to soil

Plastic pollution can enter terrestrial environments through various sources including litter, plastic products used in agriculture, such as plastic seed casings, ground covers, and crop mulch. Sources of microplastics to soil include land application of biosolids, plastic pollution, and poorly managed landfills (Alimi et al. 2018).

The settling stages of the wastewater treatment process result in the production of sewage sludge that contains large amounts of microplastics (Mahon et al. 2017). It is estimated that 99% of microplastics are removed from the influent but are retained in sewage sludge (Magnusson and Norén 2014) and that the properties of microplastics, such as their hydrophobicity and surface charge, can affect their accumulation in the solid phase (Murphy et al. 2016). However, the configurations of WWTSs differ, and thus the removal of microplastics from the influent vary from study to study (Novotna et al. 2019). Microplastics can therefore enter terrestrial environments through the application and use of sewage sludge as fertilizers for agriculture or landscaping purposes (Raju et al. 2018). In Europe and North America, around 50% of sewage sludge is recycled for use as fertilizer, and it is estimated that 44 to 300 kt of microplastics are added to farmlands in North America annually (Nizzetto et al. 2016).

3.2.3 Sources to air

Road traffic-related releases of particles from tire wear and tear are a source of microplastics to outdoor air (Panko et al. 2013, 2019; Kole et al. 2017; Prata 2018). Additional sources of microplastics in outdoor air are thought to include airplane tires, artificial turf, thermoplastic road markings, waste incineration, construction, landfills, industrial emissions, and tumble dryer exhaust, although their relative contributions have not been well established (Dris et al. 2016; Magnusson et al. 2016; Kole et al. 2017; Prata 2018). Deposition and dispersion of all airborne plastic particles from the air may result in accumulations of microplastics in water. Current estimates of the contribution of airborne tire wear and tear particles to water bodies and oceans are varied (e.g., Kole et al. 2017; Sieber et al. 2020; Unice et al. 2019a, 2019b) and additional research is necessary. However, findings suggest that tire wear particles that occur as road dust (i.e., particles that settle rapidly and that are less prone to air dispersion) are a more important contributor to total microplastic pollution in oceans than those found in ambient air.

The primary source of microplastic particles in indoor air is thought to be the shedding of polymeric textile fibres from clothing, furniture, carpeting, and household goods due to wear and tear or abrasion (Sundt et al. 2014; Dris et al. 2016). For example, washing clothing made from synthetic materials has been shown to release microplastics into wastewater, and it is hypothesized that air- or tumble-drying these garments would also cause fragments to be transferred to indoor air, household dust or dryer lint (Wright and Kelly 2017; Prata 2018). Synthetic textile fibres have also been retrieved from a variety of surfaces, including outdoor surfaces, suggesting that clothing and other fabrics may be additional sources of microplastics in both outdoor and indoor air (Rauert et al. 2014; Dris et al. 2016; Prata 2018).

4. Environmental fate

This section reviews the available data on the fate of macroplastics and microplastics in three environmental compartments: water, soil, and air. It then discusses the persistence of plastics in the environment and the conditions under which they will break down (e.g., transition from macroplastics to microplastics). The fate of biodegradable plastics and biobased plastics is also addressed.

The transport of plastic pollution often follows hydrological pathways (Windsor et al. 2019), with rivers being a key transport pathway (see Figure 3) (Alimi et al. 2018). From rivers, it is expected that the majority of plastic pollution will eventually be transported to the ocean. The mechanisms of transport are poorly understood, but are thought to be influenced by the shape, density, size, and surface condition (i.e., degree of weathering) of the plastic particle. It is also thought that the behaviour of macroplastics differs from microplastics since more energy would be required to transport larger plastics through an ecosystem even if the same transport mechanism is used (Windsor et al. 2019).

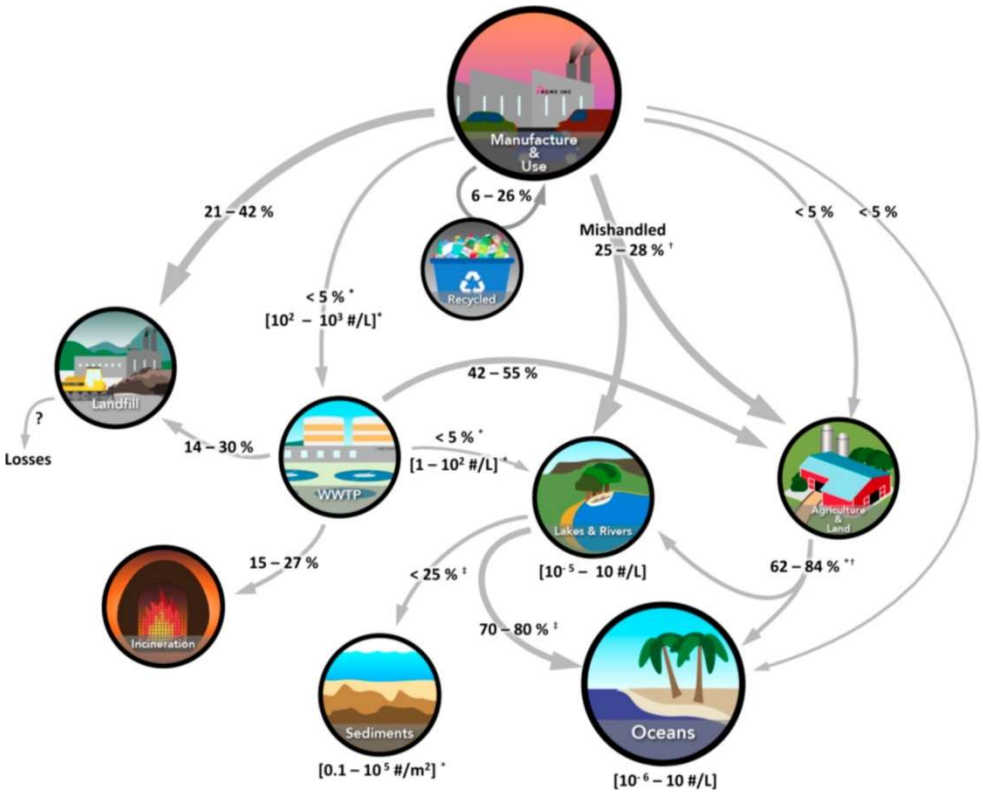

Figure 3: Estimated plastic loading and transport pathways in the environment (reproduced with permission from Alimi et al. 2018, © 2018 American Chemical Society).

Long Form Description: This figure shows the estimated plastic loading and transport pathways in the environment. Manufacture and use of plastic can result in plastics being landfilled, recycled, mishandled, or ending up in wastewater treatment plants. Mishandled plastics can end up on land, or in lakes and rivers, and subsequently in oceans and sediment. Plastics in wastewater treatment plants can

be incinerated, end up in lakes and rivers or on land (reproduced with permission from Alimi et al. (2018). Copyright 2018 American Chemical Society.)

4.1 Degradation

Plastic degradation in the environment is slow and can be affected by multiple factors (Andrady 2015; Gewert et al. 2015). Plastics that are exposed to sunlight, oxidants, and physical stress over time will weather and degrade, although the extent of degradation depends on both the environment and the chemical composition of the plastic (Eubeler et al. 2010).

Owing to their chemical structure, common synthetic polymers are durable and can be resistant to degradation. However, there are multiple processes that can bring about the degradation of polymers. These include solar UV-induced photodegradation, thermo-oxidation, hydrolysis, and biodegradation (i.e., degradation by microorganisms). The most common polymers in the environment, such as PE, PP, PS and PVC, possess a carbon backbone that is resistant to biodegradation. Therefore, in order for biodegradation of these polymers to occur, an abiotic degradation step is needed to first break them down into smaller, lower molecular weight fragments (Gewert et al. 2015; Ng et al. 2018). Given that plastic degradation occurs primarily through exposure to sunlight, degradation is most intensive in environments such as the sea surface and beaches (Andrady 2015).

The first visual effects of polymer degradation are changes in colour and cracking of the surface (Gewert et al. 2015). Surface cracking causes the inside of the plastic material to be further available for degradation, eventually leading to embrittlement and physical breakdown upon exposure to abrasive or mechanical forces, such as wind, waves, and physical impacts (Gewert et al. 2015; Ng et al. 2018; SAPEA 2019). Over time, fragmentation can result in plastics no longer being visible in the environment (Selke et al. 2015).

It is important to note that several degradation pathways may take place simultaneously since various factors initiate degradation. For that reason, degradation products may be more diverse than those expected for any specific pathway. In the marine environment, most plastics degrade first at the polymer surface that is exposed and available for chemical or enzymatic attack. Microplastics have higher surface-to-volume ratios than macroplastics and therefore degrade faster than macroplastics, but the process is still slow (Andrady 2015).

There are numerous gaps in research on plastic degradation. To estimate biodegradation, many studies examine factors such as weight loss, decrease in tensile strength, visual disappearance, or the growth of different microorganisms (Zumstein et al. 2019).

Plastics often include additives that, when released, may degrade to form other chemicals. In addition, additives such as stabilizers may enhance resistance to degradation. A study by Selke et al. (2015) evaluated the effect of biodegradation-promoting additives on the biodegradation of PE and PET in compost, landfill, and soil environments. They found that none of the additives significantly increased biodegradation in any of the conditions, and there was no evidence that these additives promoted or enhanced biodegradation of PE or PET polymers (Selke et al. 2015).

4.1.1 Biodegradable, compostable, biobased and oxo-degradable plastics

Biodegradable, compostable, biobased and oxo-degradable plastics are often regarded as potential solutions to the accumulation of plastic litter and waste (European Commission 2019; Napper and Thompson 2019). Some of these terms are explicitly defined elsewhere in the context of various certifications (e.g., ASTM D6400, ASTM D6868-19, and ASTM D883-20a). The following provides a brief overview of these plastics as they relate to the issue of plastic pollution.

Biodegradable plastics are a type of plastic that possess heteroatoms along their backbone that render them more susceptible to hydrolytic or enzymatic reactions (Ng et al. 2018). These processes cause the structure to break down into lower molecular weight fragments that microbial cells can assimilate and subsequently mineralize[3] either aerobically or anaerobically. The conditions for biodegradable plastics to break down vary and there is a need to differentiate degradation pathways under different conditions (Lambert and Wagner 2017). For example, some types of biodegradable plastics will not mineralize unless they are exposed to temperatures above 50°C for long periods of time, conditions that are rarely found in the natural environment, but rather in industrial composting facilities (UNEP 2015).

Compostable plastics are a type of biodegradable plastic that are designed to biodegrade in a managed composting process through the action of naturally occurring microorganisms, typically within a specified time frame (Napper and Thompson 2019). While composting of these products has been explored in Canada, very little post-consumer plastic is managed through industrial composting facilities (ECCC 2019e). Difficulties distinguishing compostable from non-compostable plastics can also create contamination problems for processors. As a result, some certified compostable plastics are not accepted by many composting facilities in Canada (ECCC 2019e).

Biobased plastics are plastics that are synthesized from biomass or renewable resources. Many do not necessarily biodegrade more readily than conventional plastics (European Commission 2019), and unless demonstrated through a complete lifecycle analysis, they do not necessarily confer any superiority to petroleum-based plastics with respect to environmental factors (Vert et al. 2012).

Oxo-degradable plastics, which are sometimes referred to as oxo-biodegradable plastics (UNEP 2015), are formulated using conventional polymers with the addition of heat and UV-activated additives to accelerate their fragmentation into microplastics. While it is expected that accelerated fragmentation would also accelerate degradation, the degree and speed of fragmentation are dependent on environmental conditions such as temperature and light intensity, which vary from day to day, and according to local conditions. Therefore, there is no conclusive evidence that accelerating fragmentation will enable degradation. Given that fragmentation of oxo-degradable plastic requires oxygen and that the majority of plastics in landfills will not have direct access to oxygen, little to no biodegradation of oxo-degradable plastics is expected in deeper landfill layers. In addition, there is insufficient evidence to

[3] Mineralization is the complete breakdown of a polymer as a result of abiotic and microbial activity into inorganic compounds (e.g., CO_2, H_2O, and methane) (UNEP 2015).

indicate that oxo-degradable plastics will biodegrade in a reasonable timeframe in the marine environment (European Commission 2018).

Overall, there is a lack of significant evidence that biodegradable, compostable, biobased, and oxo-degradable plastics will fully degrade in natural environments (UNEP 2015; European Commission 2018, 2019). Further studies would assist in understanding the environmental impacts of different types of biodegradable, biobased, compostable and oxo-degradable plastic, particularly in comparison to conventional, petroleum-based plastics.

4.2 Fate in water

The proportion of plastics present in surface waters and sediments varies depending on the biological (e.g., attachment of bacteria/algae), physicochemical (e.g., plastic density), and hydrodynamic conditions (e.g., mixing of the water column) (Alimi et al. 2018). Factors such as wind, surface water circulation, temperature and salinity influence the distribution of microplastics (Zbyszewski et al. 2014; Corcoran et al. 2015; Anderson et al. 2016).

In the aquatic environment, the rate of degradation of plastics is temperature-dependant, with degradation proceeding more slowly in cold water (Andrady 2015). Plastics found below the photic zone in the water column degrade very slowly, resulting in high persistence of plastics in the aphotic zone, particularly at the seafloor. In addition, biodegradation of plastics by microorganisms is negligible because of the slow kinetics of biodegradation at sea and the limited oxygen supply for these processes (Andrady 2015).

A study by Leonas and Gorden (1993) looked at disintegration rates of LDPE, PS, and a 2% ethylene-carbon monoxide polymer, as well as other blends in aqueous media. The results showed that while the ethylene-carbon monoxide polymer disintegrated[4] more rapidly than the other films evaluated, the aqueous environment significantly delayed, if not inhibited, the degradation of the other polymers.

Biber et al. (2019) studied the deterioration of different plastics in air and seawater. Macro-sized pieces of PE, PS, PET, and a commercial material marketed as degradable plastic were exposed to environmental conditions in air and water. All materials deteriorated more slowly in seawater than in air, likely due to reduced exposure to sunlight and thus reduced photooxidation in seawater. The authors found that PS showed the most rapid deterioration and is likely to break down into microplastics faster than the other materials evaluated, but that all materials tested did deteriorate to microplastics. Given the requirements for breakdown, it is expected that plastic items likely remain in seawater and that the formation of microplastics would occur in areas where plastic pollution is exposed to oxygen and UV radiation, such as intertidal habitats and at the water surface.

[4] Disintegration is the breakdown of the polymer material as evidenced by the loss of physical and mechanical properties.

4.2.1 Sediment

Plastics may remain in benthic systems of lakes and rivers or be transferred along an altitudinal gradient towards marine ecosystems (e.g., oceans). As plastics move from source to sink, they interact with the physical, chemical, and biological environments in ways that depend on the characteristics of the plastics (e.g., density) (Windsor et al. 2019).

Besseling et al. (2019) found that microplastic concentrations on a volume basis are higher in sediments than in surface water. This can be explained by the settling of particles either as singular particles or in aggregated or fouled form. The authors also found that concentrations in beach sediments were higher than in subtidal sediments, which may be explained by the relatively low density of plastics compared to seawater, causing floating and suspended plastics to be washed ashore.

Sinking fecal matter from zooplankton that have ingested microplastics represents a mechanism by which floating plastics can be vertically transported away from surface waters and into deeper waters and the benthos, thus providing food for sediment-dwelling biota (Cole et al. 2016). Wieczorek et al. (2019) found that microplastics significantly altered the size, density, and sinking rates of zooplankton fecal pellets. In oceanic conditions, fecal pellets with reduced sinking velocities are more prone to consumption, fragmentation, and microbial degradation, resulting in their mineralization within the upper regions of the water column and therefore in reduced particulate organic matter export to deeper waters (Cole et al. 2016).

Fecal pellets containing microplastics that reside at the sea surface for a prolonged period are also more readily available for ingestion by other organisms, resulting in the trophic transfer of microplastics. Wieczorek et al. (2019) note that despite this, microplastics have been found in deep-sea sediments and benthic deep-sea organisms. Thus, an unknown proportion of microplastics are likely being transported to the seabed from fecal pellets where they become available to the benthos communities.

While sediment is largely expected to be a sink for macroplastics and microplastics (Eriksen et al. 2014; Woodall et al. 2014), there is significant mobilization and removal of microplastics in sediment during high flow events such as flooding (Hurley et al. 2018). Plastics in benthic sediments may be temporarily stored and remobilized by physical and biological processes. However, there is limited research on these mechanisms of plastic transport in aquatic systems (Windsor et al. 2019).

4.2.2 Impact of biofouling on aquatic distribution

Biofouling, also known as biofilm formation, is the coating of plastics with microorganisms, algae, or plants. This process can lead to a loss of buoyancy and thus promote the sinking of microplastics to the bottom of the water body (Weinstein et al. 2016; SAPEA 2019). It has been hypothesized that phytoplankton aggregates act as potential sinks for microplastics (Long et al. 2015). Kaiser et al. (2017) found that the sinking velocities of PS particles increased by 16% in estuarine water and 81% in marine water after a six-week incubation period, which allowed for the particles to become coated with biofilms. The sinking of PE particles was not impacted by biofouling during 14 weeks of incubation in estuarine water, but in coastal water, their sinking velocity increased after six weeks. These results

indicate that biofouling can enhance deposition of plastics to sediments and ocean beds (Kaiser et al. 2017). Further, Weinstein et al. (2016) indicated that biofilm formation on plastics decreases their UV transmittance, which could also inhibit the degradation of plastics in the environment.

4.3 Fate in soil

Although limited scientific information is available on the fate of plastics in the soil compartment, studies indicate that biodegradation can play a role in the fate of plastics in soil. Certain organisms, such as bacteria (Huerta Lwanga et al. 2018) or insect larvae (e.g., moths), can degrade plastics; however, this is not likely a relevant process in natural agroecosystems since they may not be naturally present in these environments (Ng et al. 2018). Alternatively, co-metabolism (i.e., the degradation of a compound in the presence of another compound used as a carbon source) is more likely to occur due to the abundance of carbon resources in soil (Ng et al. 2018).

The physicochemical state of plastics is also likely to be very dynamic in soil due to interactions with soil components, including organic matter (Ng et al. 2018). Interactions with certain pesticides can facilitate photodegradation or embrittlement of plastic particles (Schettini et al. 2014). As photo- and thermo-oxidative degradation pathways both rely on the combination of free radicals and oxygen, these processes only occur near or at the surface of soil (Ng et al. 2018).

In a study by Cosgrove et al. (2007), PU was observed in soil at different organic carbon contents and different pH levels, and their fungal communities were compared. PU appeared to be highly susceptible to biodegradation in soil and was degraded almost completely after five months (Cosgrove et al. 2007; Eubeler et al. 2010). In another study, biodegradation in compost was investigated for irradiated ethylene propylene copolymers, LDPE, and isotactic PP films (Eubeler et al. 2010). The results showed that degradation increased with increasing irradiation time; however, after six months of exposure, LDPE was still the slowest sample to be degraded as measured by weight loss (Eubeler et al. 2010). Ohtake et al. (1995) found no evidence of biodegradation for PS, PVC and urea formaldehyde resin that had been buried under soil for over 32 years. Another study found that an LDPE bottle buried in shallow soil under aerobic conditions for over 30 years underwent degradation on the surface, but the inner part was almost unchanged (Ohtake et al. 1996).

Following the release of microplastics to the terrestrial environment, particles can be transported to surface water bodies by wind and water erosion or dispersed through ingestion by organisms (Maaß et al. 2017; Hurley and Nizzetto 2018). There is also the potential for microplastics to leach into groundwater aquifers due to downward drainage from soils (Re 2019).

Soils are also expected to act as a major sink for plastic particles (Hurley and Nizzetto 2018). Microplastics are likely to be retained in soils for long periods of time due to factors such as vertical transport that draw the particles away from the surface, hindering degradation (Horton and Dixon 2017; Huerta Lwanga et al. 2017). Zubris and Richards (2005) studied fibres in soil as an indicator of the application of biosolids to land in the United States. The authors detected fibres in soil samples from field application sites up to 15 years after the application of sludge, and these data were corroborated

with biosolids application records. Additionally, vertical transport of microplastics is possible via the movement of soil organisms (Huerta Lwanga et al. 2017; Maaß et al. 2017; Rillig et al. 2017) as well as agricultural processes (e.g., tilling), which can also cause physical damage to the particles (Ng et al. 2018).

4.4 Fate in air

Although research on the fate of microplastics in air is lacking, it is understood that air is likely an important pathway for the transport of microplastic particles (Dris et al. 2016).

When released into the atmosphere, microplastics can become suspended, or further transported, due to their light weight (Horton and Dixon 2017). Suspension and dispersion of particles in the air is dependent on factors such as the size, shape and density of a particle (lighter, less dense polymers can be carried more easily), wind conditions, and precipitation, which can facilitate deposition on land or water (Dris et al. 2016; Prata 2018).

Air currents and wind can transport particles long distances. Since air currents can be multidirectional, transport in air is less limited than transport in aquatic or terrestrial environments (Horton and Dixon 2017). For example, Allen et al. (2019) observed microplastic deposition in the French Pyrenees, a remote mountain catchment. Preliminary trajectory assessments showed that the microplastics had travelled up to 95 km from their source, indicating possible long-range transport. Microplastics have also been found in the Arctic Ocean in several studies. Lusher et al. (2015a) first reported the quantity of microplastics in surface and sub-surface Arctic polar waters. Subsequently, Bergmann et al. (2017) found large quantities of microplastics in Arctic deep-sea sediments, Kanhai et al. (2018) identified the abundance, distribution and composition of microplastics in sub-surface waters of the Arctic central basin, and Peeken et al. (2018) found microplastics in Arctic sea ice cores at five different locations and analyzed their content and composition. As with persistent organic pollutants, it is speculated that long-range atmospheric input of microplastics is one of several possible transport mechanisms (with others being prevailing currents and food-webs) to the remote Arctic Ocean (AMAP 2004; Kanhai et al. 2018; Peeken et al. 2018).

A number of studies have quantified microplastic presence in the atmosphere as well as in fallout (i.e., particles that settle on a surface during the sampling period). One study, for instance, observed atmospheric fallout of microplastics at a sampling site in a dense urban environment with a daily range of 2 to 355 particles/m^2 (Dris et al. 2016). A previous study by Dris et al. (2015) measured a total atmospheric fallout of 29 to 280 particles/m^2/day in the urban Greater Paris region. In Dongguan City, China, the concentrations of microplastics in atmospheric fallout samples collected from three sites over a period of three months were 31, 33, and 43 particles/m^2/day (Cai et al. 2017). Three different polymer types were identified in the microplastic samples (PE, PP, and PS), and fibres were the predominant shape of the microplastics sampled. In the metropolitan region of Hamburg, a median microplastic fallout concentration of 136.5 to 512.0 particles/m^2/day was found across six sampling sites over a 12-week sample collection period (Klein and Fischer 2019). Of the microplastics detected, 95% were fragments, with fibres making up the remaining 5%. During periods of higher rainfall, Dris et al. (2016)

observed a higher number of fibres in atmospheric fallout; however, there were likely other temporal and mechanistic factors at play, which the authors did not identify.

In general, atmospheric concentrations of microplastics are likely to be correlated with population density, as human activities strongly influence the environmental release of microplastics. The fate and transport will depend on prevailing meteorological conditions, with long-range transport from urban source regions to remote locations highly probable.

5. Occurrence

This section reviews the available data on the occurrence of macroplastics and microplastics in aquatic and terrestrial environments and air, as well as in other matrices through which humans may be exposed to microplastics of environmental origin (namely food and drinking water). Occurrence in biota, with the exception of occurrence in food, is covered in Section 6. Where possible, Canadian occurrence data are presented. However, since Canadian occurrence data are often lacking, data from other areas around the world are also presented in many instances.

The science assessment endeavours to discuss microplastic and macroplastic occurrence in the environment; however, peer-reviewed literature generally focusses more on microplastic rather than macroplastic occurrence. Studies looking solely at macroplastic occurrence in the Canadian environment are often limited to data from litter cleanup initiatives as well as reports in the popular press. Moreover, many studies on the occurrence of macroplastics in the environment are linked to effects such as entanglement or ingestion and much of this discussion is found in Section 6.

The absence of standardized methods and analytical techniques poses a significant challenge to quantifying microplastics in the environment. As a result, it is not possible to quantitatively characterize environmental or human exposure levels at this time.

5.1 Environmental occurrence

5.1.1 Occurrence in the aquatic environment

Plastic pollution in the aquatic environment is summarized below with a focus on four compartments: shorelines (including the intertidal zone), surface waters, benthic zone (i.e., the bottom of a water body) and groundwater.

As there are limited standardized procedures for quantifying microplastics in the aquatic environment, qualitative criteria were developed to identify studies that applied practices such as the use of controls, use of appropriate and clean glassware, and application of contamination avoidance measures. Further, the criteria used for studies, specifically on the occurrence of microplastics on shorelines and in surface water, selected studies in which microplastics were identified using an analytical method, such as Fourier transform infrared spectroscopy (FTIR), Raman spectroscopy, or pyrolysis gas chromatography mass spectrometry (GCMS). These qualitative criteria draw from the knowledge presented in Koelmans et al. (2019), which uses quantitative criteria for determining the quality of studies discussing the

occurrence of microplastic in water samples. However, as acknowledged in Koelmans et al. (2019) and in Hermsen et al. (2018), these criteria are not an absolute judgement of the value of studies. Further, as illustrated in Gouin (2020) these metrics do not necessarily weigh all aspects of a study appropriately. For example, it is possible that studies that rely on visual identification of microplastics may score relatively higher when compared to studies that use analytical characterization.

The science assessment reviews the current state of science regarding plastic pollution and acknowledges that uncertainties exist and high quality information is lacking in several study areas. As such, if any studies included in this report deviated from the above criteria, the limitation is explicitly mentioned in the text.

Shoreline

In an effort to remove litter from Canada's shorelines, 21 300 cleanups have been organized by the Great Canadian Shoreline Cleanup (GCSC)[5] across the country since 1994 (GCSC 2018a). Of the top 10 most common litter types collected during the 2018 Shoreline Cleanup, seven were either plastics or items containing plastic. Plastic items included cigarette butts, tiny plastics or foam, bottle caps, plastic bags, plastic bottles, straws, and food wrappers (GCSC 2018b). A total of 0.1 kt of litter was removed from Canadian shorelines in 2018 (GCSC 2018b). Figure 4 illustrates the contribution of plastics to shoreline litter collected during historical beach cleanup surveys of the Great Lakes.

[5] The Great Canadian Shoreline Cleanup defines shorelines as anywhere land connects with water, including creeks, streams, rivers, oceans, marshes, and even storm drains (GCSC [date unknown])

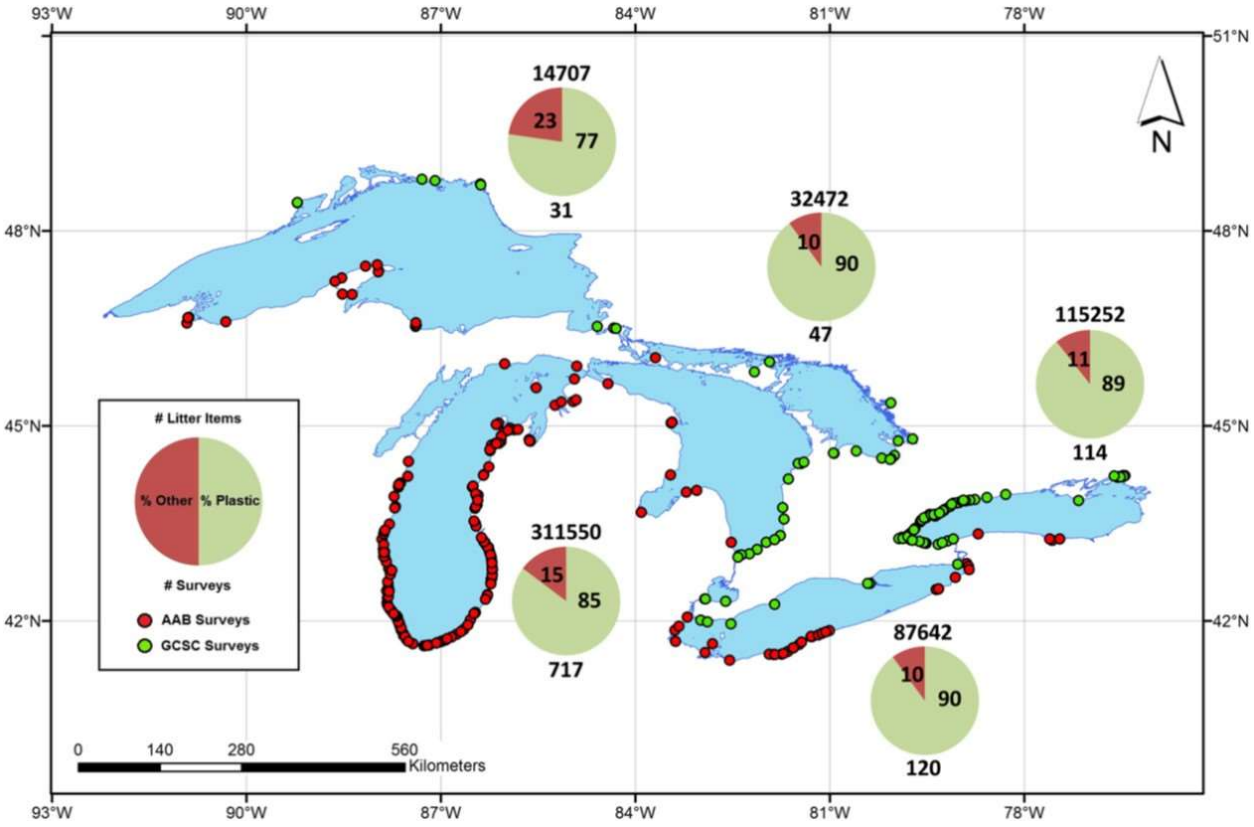

Figure 4: Great Lakes beach cleanup surveys (reproduced with permission from Figure 2 of Driedger et al. 2015)

Long Form Description: Great Lakes beach cleanup surveys. The figure shows the locations of beach cleanup surveys, the number of surveys conducted, and the percentages of anthropogenic litter comprised of plastic for each of the Great Lakes. Lake Ontario: 115252 litter items collected from 114 surveys (89% plastic); Lake Erie: 87642 litter items collected from 120 surveys (90% plastic); Lake Huron: 31472 litter items collected from 47 surveys (90% plastic); Lake Michigan: 344550 litter items collected from 717 surveys (85% plastic); Lake Superior: 14707 litter items collected from 31 surveys (77% plastic). The data used in the figure were collected by Adopt-a-Beach™ (AAB) and Great Canadian Shoreline Cleanup (GCSC) volunteers in 2012. (Reproduced with permission from Figure 2 of Driedger et al. 2015)

It is worth noting that beach cleanups generally target larger intact and mostly intact debris, resulting in underestimations of smaller plastic debris (Zbyszewski et al. 2014). Moreover, brittle plastic materials may break into smaller fragments during analysis, and broken pieces may be quantified as microplastics rather than macroplastics, thus affecting counts (Esiukova 2017). Other methods of plastic quantification may underestimate the amount of microplastics, such as surveys on rocky shorelines. McWilliams et al. (2018) highlighted the need to further develop protocols and techniques to sample microplastics on rocky shores where marine litter may be caught between rocks and crevasses, thus increasing their lifetime on the shore. In addition, waves may grind macroplastics against rocky shores, accelerating their breakdown into microplastics. The authors conducted an accumulation survey of Fogo Island beach in

Newfoundland and Labrador and found that 82% of marine litter collected from the rocky beach surface consisted of plastics, with 67% of litter being smaller than 1 cm^3.

McWilliams et al. (2018) also conducted a standing stock survey to assess the abundance of plastics at different depths of the beach on Fogo Island. This was done by first picking visible particles from the top layer of shoreline, followed by shovelling a 5 cm layer into a tray. Stratified sampling was performed to a depth of 20 cm. Across all layers, glass comprised 75.7% of litter, and plastics comprised 17.9%. More than 82% of the visually identified plastic particles sampled were macroplastics. Potential plastic particles were found throughout the different depths sampled, with the vast majority of smaller items found below the surface. Particles below the surface were found to be smaller and more abundant than particles on the surface. The surface accumulation survey in conjunction with the standing stock survey provides insight into how many particles would be missed by a survey that only considers particles on the surface.

In Lake Erie, Dean et al. (2018) collected 12 sediment samples from six beaches at the foreshore (between low- and high-water marks) and backshore (high-water mark to inland limit of beach). All samples contained microplastics, with a range of 50 to 146 particles/kg. Most of the backshore samples contained higher concentrations of microplastics than the foreshore samples. The dominant microplastic type was fibres, followed by fragments. It should be noted, that although precautions were taken against the contamination of samples from microplastics during processing, some samples were stored or sampled in PET jars or PVC liners, and sometimes new and unopened plastic containers were used in the field without prior rinsing (Dean et al. 2018).

Proximity to industrial sources may be associated with higher concentrations of plastics (Zbyszewski and Corcoran 2011; Zbyszewski et al. 2014; Corcoran et al. 2015; Driedger et al. 2015; Ballent et al. 2016). A review of existing shoreline studies by Driedger et al. (2015) indicated that higher concentrations of plastic debris in the Great Lakes region are generally found in areas with higher human and industrial activity. Zbyszewski et al. (2014) collected samples along the shorelines of Lake Erie, Lake Huron and Lake St. Clair. Macroplastics and microplastics were found along all shorelines. Additionally, Zbyszewski and Corcoran (2011) found that along the shoreline of Lake Huron, pellets represented the majority of the plastic collected. Further analysis found that most of the collected industrial pellets were PE and PP, similar to those produced by petrochemical companies (Zbyszewski et al. 2014). The relative lack of plastic debris found on the north and west shores of Lake Huron in contrast to the southeast shore suggests that the pellets followed the cyclonic flow of surface water currents away from the region of Sarnia (Zbyszewski and Corcoran 2011; Zbyszewski et al. 2014). Similarly, Corcoran et al. (2015) collected 6172 plastic pieces from Humber Bay Park West beach on the northwest shoreline of Lake Ontario and found that industrial pellets were the most common type of plastic, followed by fragments. Excluding PS, which was only quantified by mass due to the large quantity collected, the plastics concentration was 21.8 items/m^2. The majority of pellets and fragments had accumulated within organic matter along the strandline. Several nearby tributaries pass through heavily industrialized areas before draining into Lake Ontario. The researchers observed similar types of plastic pellets in sampling sites along the tributaries and at the beach, suggesting a transport pathway (Corcoran et al. 2015). Ballent et al. (2016) found that the highest concentrations of microplastics in beach sediments along Lake Ontario were in

the Greater Toronto Region, more specifically in an area made up of five watersheds that contained half of all plastic production facilities in the study region as well as 40% of the total population at the time. In this study, fragments were the predominant type of microplastic detected in beach sediment (average of 140 particles/kg dw), followed by fibres.

Munier and Bendell (2018) visually identified and collected plastic litter on the beach surface of Burrard Inlet in British Columbia. Of the 150 items collected, 144 were plastics, which were divided into seven major user groups: bags, car/bicycle parts, everyday items, food associated, packaging, functional use, and children's toys. The majority of the plastics were items related to food consumption, such as cups, straws and forks, and packaging.

Single-use plastics are one of the most common types of macroplastics found on shorelines internationally. In Canada, 17% of collected shoreline litter consisted of plastic single-use food and beverage items (GCSC 2018b). Similarly, Earthwatch Europe (2018) found that single-use plastics are a significant category of plastic pollution in European freshwater environments. The 10 most prevalent macroplastics in European freshwater environments were plastic bottles (14% of identifiable plastic pollutants), food wrappers (12%), cigarette butts (9%), food takeout containers (6%), cotton bud sticks (5%), cups (4%), sanitary items (3%), smoking-related packaging (2%), plastic straws, stirrers and cutlery (1%), and plastic bags (1%) (Earthwatch Europe 2018). Cigarette butts rank high on both European and Canadian litter lists, with cigarette butts topping Canada's 2018 Shoreline Cleanup for the highest abundance of litter and smoking-related litter making up 42.1% of the types of litter collected. In the northeast Atlantic, marine litter ranging from 2 to 30 cm was collected on beaches in the Azores, and plastic items accounted for 93% (26 321 items) of all litter. The collected litter consisted of 15.1% single-use items, 7.9% fishing-related items, and 71% fragments (Pieper et al. 2015). In the southern Caribbean, 42 585 litter items greater than 25 mm were collected at 10 locations on sandy beaches in Aruba. Of the litter collected, 89% (38 007 items) consisted of plastics. The collected litter was composed of 51% single-use plastics, of which 18% was bottle or container caps, 9% straws and 7% cigarettes. Additionally, 5% of all litter collected was fishing-related and 40% consisted of fragments of undiscernible origin (de Scisciolo et al. 2016).

In Canada, fishing-related litter made up only 1% of the litter collected in the 2018 Canadian Shoreline Cleanup (GCSC 2018b). Additionally, plastic items related to fishing activity make up a significant amount of plastic pollution found on shorelines globally (Browne et al. 2010; Chen H et al. 2019; PAME 2019). Fishing-related litter is especially important in the Arctic, where most of the marine litter analyzed in the northern parts of Norway, the Barents Sea region, and the Arctic originated from fishing-related activities (Hallanger and Gabrielsen 2018). Fishing-related litter also accounted for 48% to 100% of the mass of litter on the beaches of Svalbard, Norway (PAME 2019). An average of 1 040 plastic items/km were collected in Iceland, corresponding to an average of 104 kg/km that mostly originated from Icelandic fisheries (Kienitz 2013).

Plastics have been reported on shorelines around the world. Microplastics have been found on every Californian beach sampled by Horn et al. (2019), and fibres accounted for 95% of the microplastic items. Macroplastics have been found on beaches surveyed in Polynesia (Connors 2017) and on shorelines in

East China (Chen H et al. 2019). In China, microplastics were collected on beaches adjacent to the Bohai and Yellow Seas, where flakes were the most abundant type of plastic (Zhou et al. 2018). On surveyed South African beaches, industrial pellets were the most abundant type of plastic (Ryan et al. 2018). Typically, pellets enter the environment via accidental spills on land or at sea, and weather conditions play a factor in industrial pellet accumulation, as well as the presence of beached organic materials (e.g., wood, weeds) in which they may become entrapped (Corcoran et al. 2015; Ryan et al. 2018). Microplastics have also been collected on beaches along the southeast coast of India (Karthik et al. 2018). Finally, both macroplastics and microplastics are widely found in the Arctic, despite its distance from industrialized and highly populated areas (PAME 2019). Refer to Section B-1 of Appendix B for further data on the occurrence of plastics on shorelines.

Surface water

Several studies have looked at the occurrence of plastic pollution in Canadian marine and freshwater bodies, with a focus on the Great Lakes. According to Driedger et al. (2015), areas with greater human and industrial activity in the Great Lakes region are generally associated with a higher abundance of plastic pollution in the adjacent Great Lakes basins.

In the Government of Canada's 2015 Science Summary on Microbeads (ECCC 2015), several publications reporting on the presence of microplastics in marine and fresh waters were summarized. Building on this, a review of additional current literature on Canadian occurrence of microplastics in surface water is provided below.

Higher concentrations of plastics can be found near harbours or recreational areas and shipping routes (UNEP 2016). Hendrickson et al. (2018) studied microplastic occurrence in the surface waters of western Lake Superior. Sample sites were selected to include environments suspected to differ in microplastic distribution based on their proximity to presumed sources of microplastic pollution, such as WWTSs, urban shorelines and river outflows. On average, the estuary and harbour regions had the greatest abundance of microplastics, followed by open water sites and then nearshore sites. The average abundance for all sites was 37 000 particles/km^2. Fibres were the most abundant type of particle, followed by fragments, films, beads, and foams.

Anderson et al. (2017) found microplastics in all surface water samples collected from Lake Winnipeg at densities ranging from 66 788 to 293 161 particles/km^2. Microplastic densities in Lake Winnipeg were significantly higher than those reported for Lake Superior and Lake Huron, but were comparable to those of Lake Erie. There were no significant differences between nearshore and offshore sites. Fibres were the most common plastic type, whereas films and foams were the least common. In general, microfibres are one of the most common types of microplastic found in the aquatic environment (Anderson et al. 2017; Burns and Boxall 2018; Dean et al. 2018; Hendrickson et al. 2018; Collicutt et al. 2019; Corcoran et al. 2020). However, the distribution of microplastic type may also depend on the sample location as well as the method of quantification.

Grbić et al. (2020) quantified microplastics and other anthropogenic particles in Lake Ontario. Samples were taken from surface waters, wastewater effluent from three WWTSs, urban stormwater runoff, and

agricultural runoff. PVC, PP and PE microplastics were found in all watershed types. PET and PE were the most predominant polymer types in surface water, PE in stormwater runoff, PET in wastewater effluent, and PVC and PP in agricultural runoff.

Globally, plastics have been reported in fresh and marine surface waters, and extensive research has been done in marine surface waters. Macroplastics have been found in the Adriatic Sea, where plastic bags constitute nearly one-third of floating macroplastics (Zeri et al. 2018). Foamed PS items were the most frequently observed macroplastics surveyed in the South Pacific, North Pacific, South Atlantic, and Indian Oceans as well as around Australia (Eriksen et al. 2014). As plastic spreads throughout the world's oceans, it accumulates in subtropical gyres, such as the North Atlantic Subtropical Gyre and the North Pacific Subtropical Gyre, which is commonly referred to as the Great Pacific Garbage Patch (Eriksen et al. 2014; Poulain et al. 2019). Microplastics have also been found in Lake Michigan in the United States (Mason et al. 2016), in the Mediterranean Sea (de Haan et al. 2019) and in the Northwestern Pacific Ocean (Pan et al. 2019). Finally, macroplastics and microplastics have been found in Arctic surface water and in sea ice, and the majority of microplastics were fibres (Obbard et al. 2014; Lusher et al. 2015a; Peeken et al. 2018). Refer to Section B-2 of Appendix B for further data on plastic occurrence in surface waters.

Benthic zone

Microplastics have been reported in the benthic zone of Canadian waters. For example, in a study on the abundance and distribution of microplastics in surface sediments in Baynes Sound and Lambert Channel in British Columbia (Kazmiruk et al. 2018), particles visually identified as microplastics were found to be abundant in the sediment samples from all sampling locations. Microbeads were the most common type of microplastic found, with a maximum of 25 368 beads/kg sediment sampled at one site, followed by fibres and fragments.

Ballent et al. (2016) quantified microplastics in Lake Ontario in nearshore, tributary, and beach sediment. Fragments and fibres were the dominant type of microplastic in the size range of less than 2 mm, and fragments and industrial pellets were the primary type of microplastic in the greater than 2 mm size range. Fibres were most abundant in nearshore samples, and pellets were present in all depositional environments, but not in sediment traps. PE was the most common type of polymer in the samples analyzed. The mean microplastic abundance was 760 particles/kg sediment. The highest abundances of microplastics were found in nearshore sediments, with 980 particles/kg dw, followed by tributary and beach sediments. Lake-bottom samples were also collected from Lake Ontario, with a total of 35 pieces being found in the two sample cores (Corcoran et al. 2015). No plastics were found in samples collected from depths greater than 8 cm.

Dean et al. (2018) examined microplastic occurrence in nearshore and tributary sediment along the shoreline of Lake Erie. Benthic sediment was sampled from Lake Erie nearshore locations, from the mouth of the Grand River, and from the Detroit River. Sediment samples were also collected from two northwestern Lake Erie tributaries and two northeastern tributaries. The concentration of microplastic particles in nearshore samples ranged from 0 to 391 particles/kg sediment, and fibres were the primary

type of microplastic, followed by fragments. Tributary samples ranged from 10 to 462 particles/kg sediment with fragments dominating the samples, followed by fibres. A tributary sample from the Welland Canal, which is exposed to high shipping traffic and a sizable population, contained the largest concentration of microplastics. The passive sediment trap sample contained no microplastics, whereas the grab sample from the same location contained 390 particles/kg sediment (Dean et al. 2018).

Corcoran et al. (2020) investigated the distribution of microplastics in the Thames River in Ontario. Benthic sediment samples were collected from 34 locations along the river. All samples were found to contain microplastics, with an overall average concentration of 612 microplastics/kg dw sediment. Fibres were the most abundant (60%), followed by fragments (37%) and beads (3%). The most common polymer was PET for fibres and PE for fragments. Urban sections contained an average of 269 microplastics/kg dw, compared to rural sections which contained an average of 195 microplastics/kg dw. However, there was no significant influence of land use on the abundance of microplastics. Additionally, microplastics were found in the greatest abundance in samples with the finest sediment grain sizes and with the most organic debris.

Goodman et al. (2020) performed underwater video surveys of benthic debris in the Bay of Fundy. Surveys were conducted at 281 different locations, providing 33 hours of seafloor video footage. From the swept area, 47 items of debris were visually identified, 51% of which were categorized as plastic (71% of which were plastic bags). Abandoned, lost and discarded fishing gear, which included various plastics, comprised 28% of all debris.

Globally, plastics have been reported in marine sediment, where they are typically dominated by microplastics. Macroplastics have been found in sediment in Argentina and the United Kingdom, and the dominant type of plastic was packaging and wrappers (Browne et al. 2010; Blettler et al. 2017). Macroplastics and microplastics have also been found in sediment in Italy, with fibres being the most abundant type of microplastics (Fastelli et al. 2016). Similarly, fibres were the predominant type of microplastic found in Croatia and in the Arctic (Sundet et al. 2016; Blašković et al. 2017; Renzi et al. 2019a). Microplastics found in river sediment in Shanghai consisted primarily of spheres, and the most dominant polymer was PP, similar to the situation in Hungary and on Rameswaram Coral Island, along the southeast coast of India (Peng et al. 2018; Vidyasakar et al. 2018; Bordós et al. 2019). Plastics have been collected from the Spanish Mediterranean seafloor (García-Rivera et al. 2018), the Arctic seafloor (PAME 2019), and the Pacific Ocean's Mariana Trench (Chiba et al. 2018). Refer to Section B-3 of Appendix B for further data on plastic occurrence in the benthic zone.

Groundwater

Groundwater is likely less vulnerable to microplastic pollution than surface waters (WHO 2019), although it has recently been hypothesized that microplastics from soils may be transported to and within aquifer systems (Re 2019). Currently, there is very little empirical data on the occurrence of microplastics in groundwater.

Mintenig et al. (2019) investigated the presence of microplastics in drinking water derived from groundwater sources in the northwest region of Germany. Groundwater was supplied from wells at

least 30 m in depth and microplastics over 20 µm were detected. Despite the use of very large volumes of water (1 000 L), very low microplastic concentrations were observed in groundwater, ranging from 0 to 0.007 particles/L, with a mean of 0.0007 particles/L. All identified microplastics were found to be small fragments between the sizes of 50 and 150 µm, with the predominant polymer types being polyester, PVC, PE, PA and epoxy resin.

A South African scoping study surveyed microplastics in groundwater from four boreholes in Potchefstroom (North West), South Africa (Bouwman et al. 2018). The mean microplastics concentration reported was 0.167 particles/L. From the microplastics identified, many of the fragments were in the lower size class range, below 600 µm.

Panno et al. (2019) studied the occurrence of microplastics in springs and wells (<65 m) from two karst aquifers in the U.S. state of Illinois. Previous studies on the groundwater chemistry in these areas reported data suggesting input from septic effluent. The authors reported the presence of microplastics in 16 of the 17 water samples collected, with a median concentration of 6.4 particles/L and a maximum of 15.2 particles/L. Due to analytical limitations, the authors noted that it is possible that some particles they reported as plastic were actually fibres of natural origin.

5.1.2 Occurrence in the terrestrial environment

Litter

Information on the occurrence of plastic in litter is sparse; however, city litter audits have provided some data on the composition of litter in Canadian cities. For example, litter audits performed in Edmonton (2019), Toronto (2016) and Vancouver (2019) found that 32%, 31% and 46% of large litter, respectively, was composed of plastic (AET Group 2016; AET Group 2019; Dillon Consulting 2019). In these cities, large litter was defined as greater than 25.8 cm^2. Plastic items found in urban litter in these cities include cigarette butts, plastic films, straws and plastic packaging. Shoreline cleanups, such as the Great Canadian Shoreline Cleanup, also provide litter data, as summarised in Section 5.1.1.

Soil

The occurrence of plastics in soil is not as well studied as it is in water and sediment. Soil is an important environmental compartment in which to quantify microplastics as they may enter soils via plastic mulching or application of biosolids, among other sources. Given the lack of research on microplastic occurrence in soil, the criteria for selecting studies for this report were less stringent than for occurrence in surface waters.

Crossman et al. (2020) measured microplastics in biosolids from two suppliers and the soils from three agricultural fields in Ontario on which the biosolids were applied. One control field with no history of biosolid use was also sampled. Microplastics were found in all biosolid samples, with average concentrations ranging from 8 678 to 14 407 particles/kg dw. Overall, fragments were more predominant than fibres, comprising 63% to 73% of microplastics in biosolids. Average microplastic concentrations in soil ranged from 24 to 358 particles/kg. PE microplastics were the most abundant

plastic polymer in both biosolids and soil. All fields that were previously treated with biosolids had higher soil pre-treatment microplastic concentrations compared to the control. The field with the greatest number of previous biosolid treatments had the highest pre-treatment soil microplastic concentration, suggesting the potential accumulation of microplastics from prior biosolid applications. Microplastic concentrations in soil increased significantly immediately after biosolid application in two fields, while the third showed a reduction. Only one field demonstrated a net gain in microplastics over the course of the study. Despite the high concentrations of microplastics that were applied to soil via biosolids, greater than 99% of those microplastics were not measured during soil sampling. Furthermore, there was an increase in the proportion of fragments in soil during and shortly after biosolid application, but the proportion of fragments eventually declined with a corresponding increase in proportion of fibres. This may indicate that fragments from biosolids are preferentially transported out of the soil matrix, but fibres are retained.

In Germany, agricultural farmland was found to have 206 macroplastic pieces per hectare. The mean concentration of microplastics sized 1 to 5 mm was 0.34 particles/kg dry weight (dw) of soil (ranging from 0 to 1.25 particles/kg dw). The most common type of polymer for both macroplastics and microplastics was PE (67.9% and 62.50%, respectively) (Piehl et al. 2018).

Liu et al. (2018) found plastics in farmland soil around the suburbs of Shanghai, China. Macroplastic particles sized 5 mm to 2 cm were found at a concentration of 6.75 items/kg in shallow soil (0 to 3 cm) and 3.25 items/kg in deep soils (3 to 6 cm). Microplastic concentrations were 78.0 items/kg in shallow soil and 62.5 items/kg in deep soil. In general, Liu et al. (2018) found that topsoil contained higher concentrations of larger sizes of plastic particles. Fibres, fragments and films were the most common types of plastics and the majority of all plastics collected were PP and PE. Zhang and Liu (2018) also explored microplastic occurrence in arable land in southwestern China. The study area consisted of two cropped areas at the upstream and estuary of the Chai River, as well as a buffer zone, which was converted from cropland in 2009 to host indigenous trees. Plastic particles were found in all samples, ranging from 7 100 to 42 960 particles/kg, much higher than the concentrations measured by Liu et al. (2018). In addition, most microplastics were less than 1 mm in size and the dominant type of microplastic was fibres, constituting an average of 92% of samples. During wastewater treatment processes, microplastics can settle in sewage sludge, which can then be transferred to agricultural soils and used as fertilizer (Corradini et al. 2019). Corradini et al. (2019) sampled 30 agricultural fields in Chile with similar soil chemical and physical characteristics, but with different sludge application records over the past 10 years. The authors found high concentrations of microplastics in the soil and reported that microplastics accumulate in the soils with successive sludge applications. Scheurer and Bigalke (2018) found microplastics at concentrations up to 55.5 mg/kg (593 particles/kg) in floodplain soil samples in Switzerland, with a mean concentration of 5 mg/kg. Macroplastics sized 5 mm to 2.5 cm were also found but in much lower concentrations.

5.1.3 Occurrence in air

Indoor air

Limited data are available on exposure to microplastics in the indoor environment. Only two studies were identified in which indoor air was sampled, and three studies were identified in which fallout samples or settled dust samples (i.e., particles sampled from surfaces or vacuum cleaner bags) were collected to characterize microplastics in indoor air. Generally, particles were examined and counted microscopically and characterized by size, shape, and composition. However, collection and analysis techniques varied, and therefore comparison between studies is not possible.

In the indoor environment, microplastics are more likely to occur in settled dust than in air, as they have a higher density than air (Henry et al. 2019). This route of exposure is particularly relevant to toddlers and young children, given behaviours such as crawling and hand-to-mouth activity. However, no data have been identified on partitioning of microplastics in indoor environments, and inhalation is therefore also considered a potential exposure route.

Dris et al. (2017) looked at fibres in indoor air, indoor fallout, and settled dust in two apartments and an office in urban Paris. They found that approximately 33% of the fibres were synthetic, including PA, PP and PE. The authors reported air concentrations of 1 to 60 fibres per m^3 (median 5.4 fibres/m^3), and dust concentrations of 190 to 670 fibres per mg. The method was limited to fibres greater than 50 µm in length. However, there was an inverse relationship between the number of fibres and their size, suggesting that smaller fibres could be present in larger numbers. Fibre concentrations in indoor air were significantly higher than in outdoor air.

Vianello et al. (2019) sampled indoor air in three apartments in Denmark and found that microplastics comprised 4% of the particles identified. The average number of microplastic particles in the samples was 9.3 per m^3. Most (81%) of the microplastics were polyester; other polymers identified included PE, PP, and polyacrylonitrile (PAN). Both fibre and fragment shapes were observed, and the size limit for detection was 11 µm. Like Dris et al. (2017), Vianello et al. (2019) reported an inverse relationship between sample microplastic concentration and median of the size distribution.

Dust was examined from 39 homes in different locations across China (Liu C et al. 2019). PET was identified in all samples, and PC was found in 74% of samples. The method used could detect particles in the range of 50 to 2 000 µm; most microplastic particles were fibrous in shape. Synthetic polymers accounted for approximately 40% of the fibres collected, including polyester, PU, PA, PE, PP, and PAN. A concentration of 17 to 620 fibres per mg of dust was reported. The study authors also reported a concentration of PET in dust by mass (median of 27 µg per mg) and a concentration of PC in dust by mass (median of 0.005 µg per mg).

An earlier study (Schneider et al. 1996) looked at personal exposure to fibres at some European sites, using personal sampling pumps to collect airborne dust. The composition of fibres was not determined, but synthetic organic fibres may have included PE, PP, poly(vinyl alcohol), polyester, PA, and polytetrafluoroethylene (PTFE).

Cox et al. (2019) did a crude estimate of inhalation exposure to microplastics using the air concentrations of fibres from Dris et al. (2017) and Tunahan Kaya et al. (2018) (see outdoor air exposure section), assuming 33% of the fibres and particles were actually microplastics (Dris et al. 2017). Similarly, Prata (2018) used the data from Gasperi et al. (2015) to estimate the number of airborne microplastics that could enter the human lung each day. However, as discussed above, no quantitative estimate of exposure to microplastics from indoor air and dust was conducted for this assessment due to the limited number of studies available, the very small sample sizes, and the varying techniques and criteria applied for sample collection and particle characterization.

Outdoor air

Only a few studies have investigated microplastics in outdoor air samples. The monitoring methods employ sampling techniques in which predetermined volumes of air are passed through filters onto which particles are collected. In addition, passive techniques that depend on atmospheric fallout onto a sampling surface or filter are used. Confirmation of microplastic particles among other particles is then completed using traditional methods. There are no Canadian data available, but limited studies were conducted in Europe, Asia and the Middle East.

Dris et al. (2017) measured total fibre concentrations, including microplastic fibres, approximately 3 metres from the roof surface of an office building located roughly 10 km from the centre of Paris (four times throughout the year to account for seasonal variations). The concentration of fibres measured outdoors ranged between 0.3 and 1.5 fibres/m^3 (median of 0.9 fibres/m^3) and was significantly less than concentrations measured indoors within the office and at two residential sites in the same region. One sample collected on a rainy day in winter contained five times more fibres, suggesting that the rain caused fallout of the fibres. The methodology used in this study has a lower observation limit of 50 µm. However, the results revealed a more elevated number of particles in the smaller size fraction, suggesting that microplastics smaller than 50 µm could be present in greater numbers.

The concentration of suspended atmospheric microplastics (SAMPs) measured in Shanghai ranged from 0 to 4.18 SAMPs/m^3 (mean of 1.42 SAMPs/m^3) (Liu K et al. 2019a). Microplastic fibres comprised 67% of the SAMPs, followed by fragments and granules (30% and 3%, respectively). The size and concentration of microplastics was shown to vary with altitude. The concentration of SAMPs was highest closer to the ground (1.7 metres), and lower at higher altitudes (33 and 80 metres). Larger sized particles (>5 000 µm) were also detected near ground level and not at higher altitudes. SAMPs were shown to represent 54% of the total particles collected and were comprised mostly of PET, PE, polyester and PAN. Poly(N-methyl acrylamide) (PAA) was the predominant SAMP at the highest altitude. It should be noted that rayon was included in the definition of SAMPs. Because this material is synthesized using cellulose, it is not always grouped with microplastics. This study estimates that the average adult in Shanghai inhales 21 microplastic particles per day.

The limited data on microplastics in outdoor air, measured in France and China, collectively identify an exposure level of approximately 1 microplastic particle per m^3 of air. The primary exposure form is through microplastic fibres. However, there is significant uncertainty with regards to exposure to

smaller microplastic particles, particularly those below 50 µm. In outdoor air, it is anticipated that most human inhalation exposures would occur near ground level and that concentrations would depend on many factors, including geographical proximity to outdoor microplastic sources, wind, temperature and precipitation (Prata 2018). Since people spend less time in outdoor or transit environments, they would be exposed to fewer microplastics outdoors than indoors.

A few studies have investigated the contribution of tire wear emissions to ambient levels of $PM_{2.5}$ and PM_{10} (Panko et al. 2013, 2019; Kole et al. 2017; Kreider et al. 2019). In general, tire wear pollution data are sparse, available for a few locations, and estimates are indirectly calculated based on limited observations. Notwithstanding study limitations, a recent analysis by Panko et al. (2019) suggests that tire wear emissions contribute less than 1% to ambient levels of $PM_{2.5}$ and less than 3% to ambient levels of PM_{10}.

5.2 Occurrence in food and drinking water

5.2.1 Occurrence in food

Current knowledge of the occurrence of microplastics in food is limited. The point sources of confirmed microplastics in food are currently unknown, although microplastics likely enter the food chain through plastic waste breaking down in environmental matrices, such as water and air. For example, animal species consumed by humans may ingest microplastics from aquatic environments or become exposed via trophic transfer of microplastics from prey to predator (EFSA 2016; Toussaint et al. 2019). It is also possible for ambient microplastics in the air to settle on food items (Catarino et al. 2018; Li et al. 2018a). In a number of microplastics occurrence studies, contamination of laboratory control blanks and test samples by background and/or ambient air microplastics was reported as a methodological challenge (Mathalon and Hill 2014; Lachenmeier et al. 2015).

Some research has suggested that food manufacturing, processing, and handling, as well as food packaging materials, may be potential point sources of microplastics in food (Karami et al. 2018; Oßmann et al. 2018; Schymanski et al. 2018). However, to date, there is no conclusive scientific evidence that food packaging materials, when used as intended (i.e., under normal conditions of use), are a source of microplastics in food or bottled water. Further studies are needed to determine whether food manufacturing, processing and/or handling, as well as food packaging materials, may contribute to microplastic concentrations in food.

The majority of available data on findings of microplastics in foods pertain to analyses conducted internationally and, unless otherwise stated, are not Canadian specific data. Most studies have focused on investigating microplastic content in seafood, specifically fish and shellfish harvested from non-Canadian marine environments (EFSA 2016; FAO 2017; Lusher et al. 2017; Barboza et al. 2018; Toussaint et al. 2019).

The available data for other animal species that may be consumed as part of the diet of Indigenous Peoples is summarized in Section 6. It is noted that the available research is limited to identifying macroplastics and microplastics from the perspective of animal health. It is not anticipated that

consumption of gastrointestinal (GI) tracts, which would likely have the greatest amount of microplastics for marine mammals, fish and seabirds, is a major source of country food consumption. For example, according to the Inuit Health Survey, the most commonly consumed country foods include the flesh and organs of various mammals, birds and fish, but the survey does not indicate that the GI tract is usually consumed (Egeland and CINE 2010a,b,c). There is a need for research to quantify the presence of microplastics in the animal tissues and organs that are typically consumed.

While some peer-reviewed studies report the presence of microplastics in certain foods, the particles were not confirmed as plastic, as the methodology employed relied on visual inspection or crude staining (Mathalon and Hill 2014; Desforges et al. 2015; Lachenmeiser et al. 2015; Liebezeit and Liebezeit 2013, 2014, 2015; Rochman et al. 2015; Wójcik-Fudalewska et al. 2016; Karlsson et al. 2017; Kosuth et al. 2018; Renzi et al. 2018). Given the lack of certainty that the particles reported in these studies are, in fact, microplastics, these results are not considered further in this report.

The available data on the occurrence of microplastics in food, including bottled water, are summarized below, with further details available in Appendix C.

Fish and shellfish

The presence of microplastics in the GI tract of over 150 fish species is well-documented, with microplastic content ranging in number from 0 to 20 microplastics per fish and ranging in size from 130 µm to 5 mm (Lusher et al. 2013; Campbell et al. 2017; EFSA 2016; FAO 2017; Barboza et al. 2018; Hantoro et al. 2019; Liboiron et al. 2018, 2019; Slootmaekers et al. 2019; Toussaint et al. 2019). There is significantly less information available on microplastic occurrence in fish muscle, which is the tissue of bony fish that is typically consumed (Karami et al. 2017a; Abbasi et al. 2018; Akhbarizadeh et al. 2018). The existing information indicates that microplastic concentrations in muscle tissue are lower than what has been reported in the GI tract of bony fish. The majority of whole fish samples (including fish muscle tissue and viscera) purchased from local markets in Malaysia did not contain any microplastics. Of the small proportion of samples that did contain microplastics, concentrations ranged from 1 to 3 microplastics per fish (Karami et al. 2017a). Conversely, microplastics were detected in all analyzed fresh fish samples from the Persian Gulf, at concentrations ranging from 3.1 to 4.6 microplastics per fish (Abbasi et al. 2018) or 0.57 to 1.85 microplastics per gram of fish muscle tissue (Akhbarizadeh et al. 2018). Most microplastics were larger than 100 µm, with fragments and fibres being the predominant particle shapes in fish muscle tissue (Abbasi et al. 2018; Akhbarizadeh et al. 2018).

Microplastics have been detected in a number of edible species of molluscs, including mussels, clams, oysters, scallops, and snails (Barboza et al. 2018; Toussaint et al. 2019). The most commonly investigated species of molluscs is the blue mussel, which was found to contain 0 to 10 microplastics per individual mussel or 0.2 to 2.9 microplastics per gram of meat (De Witte et al. 2014; Van Cauwenberghe and Janssen 2014; Li et al. 2015, 2018a; Van Cauwenberghe et al. 2015; Catarino et al. 2018; Toussaint et al. 2019). Similar concentrations of microplastics have been reported in clams, oysters, scallops, and snails (Van Cauwenberghe and Janssen 2014; Li et al. 2015; Naji et al. 2018; Su et al. 2018; Hantoro et al.

2019). Fibres and fragments were the most commonly detected shape, ranging in size from 5 µm to up to 4.7 mm (EFSA 2016; FAO 2017; Catarino et al. 2018; Li et al. 2018a; Naji et al. 2018; Su et al. 2018). The concentration of microplastics detected in mussels varies; with higher concentrations of microplastics observed in the tissue of mussels harvested from waters with higher environmental concentrations of microplastics (EFSA 2016; Li WC et al. 2016; FAO 2017; Hantoro et al. 2019).

Occurrence data on the presence of microplastics in crustaceans is extremely limited. The average microplastic content in green tiger prawns sampled from the Persian Gulf was 7.8 microplastics per individual (muscle tissue and exoskeleton combined), with filamentous fragments measuring 100 to 250 µm identified as the most abundant type of microplastic (Abbasi et al. 2018). Conversely, microplastics were observed in the digestive tract, head, and gills of whole brown shrimp, but not in the abdominal muscle tissue of peeled brown shrimp, sampled from the Clyde Sea (Devriese et al. 2015). Microplastics have also been found in the guts of lobsters at concentrations of up to 0.80 mg per individual, with fibres being the most frequently observed shape (Murray and Cowie 2011; Welden and Cowie 2016).

Other foods

The occurrence of microplastics has also been reported in a very small number of other foods, including honey, sugar, beer, and salt (EFSA 2016; Peixoto et al. 2019; Toussaint et al. 2019). One study reported that the majority of fibres in honey samples were naturally occurring cellulose fibres, with only a small portion of fibres confirmed to be PET by spectroscopy, but the number of PET fibres was not reported (Mühlschlegel et al. 2017). The remaining honey studies and all sugar and beer studies used a non-specific staining method to identify particles in the food items and thus, none of these particles could be confirmed as plastic (Liebezeit and Liebezeit 2013, 2014, 2015; Lachenmeier et al. 2015; Kosuth et al. 2018).

A recent review of microplastics in salt reported that their presence in commercial salts was common, although microplastic concentrations varied considerably depending on the origin and type of salt (Peixoto et al. 2019). Sea salts contained the highest concentrations of microplastics, ranging from 0 to 19 800 microplastics per kg of salt (Yang et al. 2015; Iñiguez et al. 2017; Karami et al. 2017b; Gündoğdu 2018; Kim et al. 2018; Renzi and Blašković 2018; Seth and Shriwastav 2018). Concentrations in lake and rock/well salts were much lower, ranging from 0 to 800 microplastics per kg of salt and 0 to 204 microplastics per kg of salt, respectively (Yang et al. 2015; Iñiguez et al. 2017; Karami et al. 2017b; Gündoğdu 2018; Kim et al. 2018). In most studies of salt, microplastics less than 500 µm accounted for the largest proportion of detected microplastics, with fragments and fibres being the most abundant microplastic shape, regardless of salt type (Yang et al. 2015; Iñiguez et al. 2017; Karami et al. 2017b; Gündoğdu 2018; Kim et al. 2018; Renzi and Blašković 2018; Seth and Shriwastav 2018).

Bottled water

A few studies have evaluated the occurrence of microplastics in bottled water (Wiesheu et al. 2016; Kosuth et al. 2018; Mason et al. 2018; Oßmann et al. 2018; Schymanski et al. 2018; Zuccarello et al. 2019). In one study, microplastics were detected in 93% of bottled water samples purchased from 19 locations in nine countries outside of Canada, with an average concentration of 10.4 microplastics greater than or equal to 100 µm/L (Mason et al. 2018). The number of particles in the 6.5 to 100 µm size range were reported. However, spectroscopic analyses were not performed at this size range, and thus the particles could not be confirmed as plastic (Mason et al. 2018).

Microplastic concentrations are reported to vary across bottle type (i.e., plastic, glass or beverage carton) and intended use conditions (i.e., single-use versus multi-use bottles) (Oßmann et al. 2018; Schymanski et al. 2018). The highest concentrations of microplastics were reported in water from older multi-use plastic bottles, followed by glass bottles, newer multi-use plastic bottles, single-use plastic bottles, and beverage cartons (Oßmann et al. 2018; Schymanski et al. 2018). Approximately 78% to 98% of the microplastics detected in bottled water samples were between 1 and 5 µm, with less than 7% of microplastics greater than 10 µm (Oßmann et al. 2018). The point source of microplastics in bottled water is still unknown, and the variation in the reported microplastic concentrations does not seem to correlate with bottle type alone. This suggests that the origin of reported findings of some microplastics in bottled water may be environmental (i.e., from the source water and air as a result of secondary microplastics forming in the environment).

5.2.2 Occurrence in drinking water

A limited number of studies have measured microplastics in tap water, and even fewer are considered reliable due to concerns with quality assurance measures (WHO 2019). Average microplastic particle concentrations in tap water have been reported to range from 0.0007 to 628 particles/L (WHO 2019), and microplastics as small as 1 µm in size have been measured in drinking water (Pivokonsky et al. 2018). Due to the limitations of existing detection techniques, no information is available on the occurrence of particles below 1 µm in size. The most predominant polymer types detected were PET and PP in the form of fibres and fragments (WHO 2019).

In a WHO-commissioned review, Koelmans et al. (2019) reviewed 50 studies on microplastics in tap water, bottled water and freshwater. The majority of the studies were missing at least one of nine critical aspects of quality assurance (Koelmans et al. 2019). Specifically, the authors noted uncertainties with the concentrations reported in many of these studies and concluded that any information presented on the presence of microplastics in water must be interpreted with this knowledge. Relevant studies on microplastics in tap water and freshwater are summarized below. See section 5.2.1 for a review of relevant bottled water studies.

Pivokonsky et al. (2018) examined raw surface water and treated drinking water for microplastics from three drinking water treatment plants (DWTPs) in urban areas of the Czech Republic. Drinking water samples were analyzed by scanning electron microscopy (SEM), FTIR and Raman spectroscopy. The

results from this quantitative analysis indicated average concentrations of 338, 443 and 628 particles/L for drinking water at each of the respective DWTPs, with microplastics smaller than 10 µm accounting for up to 95% of particles retained. Although 12 different materials were identified, PET and PP were found to be the prevailing microplastics in treated water collected at two of the DWTPs, while PP and PE were most abundant in treated water collected at the third DWTP. Some of the limitations that may affect the overall quality and reliability of this dataset include the use of small sampling volumes and failure to take sufficient measures to control background contamination (i.e., wiping down surfaces and working under clean air conditions) (Koelmans et al. 2019).

Strand et al. (2018) did not find significant concentrations of microplastics in tap water sampled from 17 different locations across Denmark sourced by groundwater. Samples were visually examined by stereomicroscopy for all particles greater than 100 µm displaying microplastic-like characteristics. Only a single sample concentration was reported above the level of detection (LoD) of 0.58 particles/L, at 0.6 particles/L. Chemical analysis by FTIR revealed that of the particles exhibiting microplastic properties, only 3% were confirmed to be microplastics, with the remainder identified as cellulose-like material (76%), as having poor spectra (10%), as having an unknown spectra (7%), or as protein-like material (4%). Polymer types were reported as PP, PS and PET. Given the very low level of each type of plastic polymer identified in the tap water samples, the authors caution against drawing conclusions on the origin of the plastic contamination. Additional tap water samples were collected to investigate the occurrence of smaller microplastics 10 to 100 µm, and chemical analyses were performed by FTIR. Only a single concentration of 0.8 particles/L was reported above the LoD of 0.3 particles/L, in the form of fragments comprised of PP, PET, acrylonitrile butadiene and PU. Despite the small sample volumes used in this study, the data presented was found to be among the most reliable studies on the occurrence of microplastics in drinking water (WHO 2019).

In a study on tap water derived from the purification of groundwater in northwestern Germany, Mintenig et al. (2019) investigated the abundance of microplastics at different locations within the drinking water supply chain. Particles were characterized using FTIR imaging, and microplastics down to a size of 20 µm were identified. Results indicated a low level of microplastic contamination of tap water derived from groundwater, with concentrations in both raw and drinking water ranging from 0 to 7 x 10^3 particles/L and a reported mean of 0.7 x 10^{-3} particles/L. Microplastic particles identified were small fragments between 50 and 150 µm in size, with the predominant polymer types identified as polyester, PVC, PE, PA and epoxy resin. Although this study lacks some aspects of quality assurance, such as the use of clean air conditions and absence of positive controls (Koelmans et al. 2019), when assessed on key quality control criteria, it was found to score the highest of all tap water studies by the WHO (WHO 2019).

Two studies (Uhl et al. 2018; Kosuth et al. 2018) were identified but not considered reliable due to uncertainty about whether the methods used could accurately identify particles as plastic. In one study, no particles were observed in treated or distributed water in 24 DWTPs in Norway (Uhl et al. 2018). In another study, Kosuth et al. (2018) evaluated synthetic particles in tap water from 14 countries across five continents and found particles in 81% of samples, with the most abundant type being fibres. Concentrations ranged from 0 to 61 particles/L, with an overall mean of 5.45 particles/L.

Finally, the possibility exists that microplastic contamination could occur at some point in the water supply chain as a result of abrasion of water pipes containing plastic materials, or from membrane filters made of polymers (Novotna et al. 2019). Further research is required to investigate this possibility.

5.2.3 Drinking water treatment

DWTPs provide a barrier against the introduction of waterborne microplastics in drinking water. The current literature, while limited, shows that drinking-water treatment can be effective at removing microplastics. However, given the lack of standardized methods for quantifying microplastics in water, further research is required in this area (Novotna et al. 2019).

Drinking water treatment typically occurs via clarification or membrane processes. Clarification processes are the most commonly used methods for removing particles from drinking water and involve techniques such as coagulation, flocculation, flotation, and/or filtration (Novotna et al. 2019). Membrane processes involve the use of diffusion membranes (e.g., reverse osmosis) or porous membranes (e.g., microfiltration, ultrafiltration). Diffusion membranes allow only dissolved substances (such as ions and specific dissolved substances) to pass through, whereas porous membranes allow only particles of a certain size to pass (Crittenden et al. 2012). As most observed microplastics are above the membrane size thresholds for porous membranes (i.e., 0.1, 0.01 and 0.001 µm for micro-, ultra- and nano-filtration respectively), porous membranes have the potential to be very effective at removing microplastics (Crittenden et al. 2012). For example, a laboratory study by Ma et al. (2018) found complete rejection of PE microplastics by an ultrafiltration membrane. The type of drinking water treatment process may affect the efficiency of DWTPs in removing microplastics. However, further research is required to inform drinking water treatment optimization for microplastics. Pivokonsky et al. (2018) observed microplastic removal rates of between 70% and 82% for three DWTPs employing conventional coagulation, clarification, and filtration. In a study using groundwater, Mintenig et al. (2019) found no significant difference between source water and treated water, although microplastic concentrations were very low in both source and treated water, varying from 0 to 0.007 particles/L (Mintenig et al. 2019).

Microplastic properties (e.g., size, shape, and surface properties), as well as water properties (e.g., pH and organic matter content) may also impact the efficiency of microplastic removal during different treatment processes. As microplastics are hydrophobic, adsorption of organic materials to the particles can occur, which can prevent their aggregation and thus make separation more difficult (Napper et al. 2015; Koelmans et al. 2016). Hydraulic forces can also break down large aggregates or particles themselves, creating smaller particles that may not be removed as easily during the clarification process (Jarvis et al. 2005). Ma et al. (2018) found that while pH and turbidity of the water had little effect on the microplastics removal efficiency, the microplastics themselves can actually influence the turbidity of water at sufficient concentrations.

6. Impacts on environmental health

This section reviews data on the effects of both macroplastics and microplastics on environmental receptors. Each subsection begins with a discussion of occurrence in biota, followed by an overview of their effects.

6.1 Macroplastic

Plastic pollution can have various effects on organisms and their habitats, depending on the size and type of plastic, and the level of biological organization (Werner et al. 2016). In 2016, the Secretariat of the Convention on Biological Diversity (CBD) reported that a total of 817 marine species had been affected in some way by marine litter (CBD 2016), up 23% from the same assessment performed four years earlier. It also found that over 80% of this marine litter was plastic (CBD 2012, 2016). A literature review of 340 publications involving 693 species found that, globally, 92% of reported interactions between litter and species were related to plastic pollution (Gall and Thompson 2015).

Rochman et al. (2016) conducted an extensive literature review of primary publications (283 papers) on marine litter (including macro- and micro-sized plastic pollution) published through to 2013. The authors compiled the perceived and demonstrated effects of litter and sorted them by levels of biological organization: suborganism, organism, population and assemblage. Micro-sized litter (defined as <1 mm in this study) accounted for 71% of the demonstrated impacts, while macro-sized litter (defined as >1 mm in this study) accounted for 29%. A further breakdown of these effects by level of biological organization shows that of the demonstrated impacts from macro-sized litter, the majority were classified as suborganismal, with the most common effects being seen in tissues (e.g., inflammation or lacerations) and organ systems (e.g., poor functioning). Of the demonstrated impacts at the suborganismal level, 78% were due to micro-sized litter, 74% of which were caused solely by plastics. Other demonstrated effects include effects on cells (e.g., necrosis, viability), in organs (e.g., change in size, lesions) and macromolecules (e.g., protein, DNA damage). All of these demonstrated impacts of macro-sized litter were found to be from plastic pollution. The remaining demonstrated effects were divided between the organismal level and the ecological level. At the organismal level, the main effect observed was death to an individual, whereas at the ecological level, the main effect was on assemblages (i.e., change in abundance or diversity of biota). The most common items reported to cause an effect were lost and abandoned fishing gear or other plastic items, such as rope, bags, straws and degraded fragments.

The adverse effects of macroplastic pollution include entanglement, ingestion, and impacts on habitat integrity (Gall and Thompson 2015; Rochman et al. 2016; Werner et al. 2016).

6.1.1 Entanglement

Entanglement from macroplastics can occur from ropes, nets, cable ties, plastic bags, packaging bands and rings (such as for cans in bulk), and other string-like items (Werner et al. 2016). Observations of entanglement are reported more frequently than other impact pathways, likely due to its very visible

nature (Werner et al. 2016). For example, Gall and Thompson (2015) found reported occurrences of entanglement for 30 896 individuals from 243 species. Of these reported cases, 79% were linked to direct harm or mortality, and the majority of these incidents involved plastic rope and netting. As well, Rochman et al. (2016) found that 29% of demonstrated impacts at the organismal level were caused by entanglement. The species most commonly impacted by entanglement events were marine invertebrates (75 species), seabirds (49 species), fish (27 species), and marine mammals (10 species).

Entanglement in the marine environment is often due to "ghost fishing," which occurs when lost, abandoned, or discarded fishing gear continues to catch fish in the ocean or on the seafloor (Hallanger and Gabrielsen 2018; PAME 2019). In the Arctic, old fishing-related products were found entangled with dead seabirds, dead and living Svalbard reindeers (*Rangifer tarandus platyrhynchus*), and seals (Hallanger and Gabrielsen 2018). In addition, Page et al. (2004) found the entanglement rates of Australian sea lions (*Neophoca cinerea*) and New Zealand fur seals (*Arctocephalus forsteri*) to be 1.3% and 0.9%, respectively, in 2002. These are some of the highest reported entanglement rates for all seal species. The authors estimated that 1478 seals die from entanglement events each year in Australia. Good et al. (2007) recovered 494 derelict fishing nets from Puget Sound and the Northwest Straits, along the coast of Washington, USA. Overall, 74% of the 7 539 organisms that were entangled in the derelict nets were dead, including marine invertebrates and vertebrates, of which 71% and 96% were recovered dead, respectively. All of the 123 birds and 16 mammals, including Harbor seals California sea lions and a Harbor porpoise, were recovered dead. In addition, a review of global data by Ryan (2018) reported that a total of 265 bird species were reported to be entangled in discarded plastics or other synthetic materials. Fishing gear was determined to be the cause of entanglement in 83% of species.

Votier et al. (2011) examined the use of macroplastics as nesting material by northern gannets (*Morus bassanus*) in Grassholm, Wales and assessed the associated entanglement events. Nests contained an average of 469.9 g dw of plastic and the preferred material used was synthetic rope. The authors estimate that, on average, 65.6 birds are entangled each year, with the majority being full-grown nestlings.

Large plastics such as bags, sheets, and films can also cover plants, sponges, and corals, affecting gas exchange and their photosynthetic capacities (Werner et al. 2016). This phenomenon, known as "smothering," can lead to mortality of affected vegetation (Kühn et al. 2015). Rochman et al. (2016) found that 8% of deaths at the organismal level were due to smothering when examining demonstrated effects. Smothering by plastic pollution can also lead to sublethal effects in these organisms. To study the effects of smothering on cold-water corals (*Lophelia pertusa*), Chapron et al. (2018) used 10 x 10 cm pieces of LDPE to represent fragments of plastic bags, which have been seen covering polyps in the field. They observed a decrease in growth rates from 3.59 mm/year in control aquaria conditions to 2.51 mm/year in the test group exposed to macroplastics. The plastics may have acted as physical barriers to feeding, leading to impaired energy acquisition and slower growth rate. In addition, activity was 11% lower in coral exposed to macroplastics in comparison to control conditions after 7 days. However, activity was enhanced after 20 days, which the authors hypothesized to be a compensatory physiological response to enhance capture efficiency or a mechanism to cope with long-term low

oxygen supply (Chapron et al. 2018). Macroplastic exposure also led to a noticeable decrease in feeding rates throughout the duration of the experiments.

Similarly, Qi et al. (2018) found that exposing soil to plastic films (1% w/w) had weak effects on the growth of wheat (*Triticum aestivum*). Plastic mulch films, comprised of 37.1% Pullulan (a polysaccharide), 44.6% PET and 18.3% polybutylene terephthalate (PBT), had stronger negative effects on wheat growth compared to the PE mulch. The authors note that this might be related to the presence of PET and PBT in the mulch, which have been shown in previous studies to have stronger negative effects on soil-plant systems than LDPE (Qi et al. 2018). However, exposure to both types of films inhibited wheat growth with respect to plant height at day 40 and shoot biomass at 2 months. The plants in both plastic mulch treatments also displayed fewer leaves, decreased leaf surface areas, and thinner stems.

6.1.2 Ingestion

Ingestion of plastics is another pathway that can lead to potential adverse effects. Ingestion of plastic can be intentional (e.g., where an organism eats the plastic, mistaking it for food), or unintentional (e.g., where predators feed on prey that have ingested plastics). Filter-feeding or detritus-feeding species are especially prone to unintentional plastic ingestion (Werner et al. 2016).

In the Mediterranean Sea, PE macroplastics were found in the gastrovascular cavity of 2 of 20 sampled jellyfish (*Pelagia noctiluca*) (Macali et al. 2018). Bernardini et al. (2018) also sampled 139 blue sharks (*Prionace glauca*) from the Mediterranean Sea. Blue sharks in the Mediterranean basin are categorized as a "Critically Endangered" species by the International Union for Conservation of Nature. Of the 95 adult blue sharks that were examined and had full stomachs, 24 contained plastic pollutants. Juveniles were also found to have a greater frequency of ingested plastics. In addition, macroplastics accounted for more than 70% of all plastic pieces. The majority of ingested plastic items were sheet-like (72.38%), followed by fragments (18.10%) and threadlike plastic items (5.71%), with the most common polymer found being PE.

The GI tracts of Atlantic cod (*Gadus morhua*), European flounder (*Platichthys flesus*), common dab (*Limanda limanda*), Atlantic herring (*Clupea harengus*), and Atlantic mackerel (*Scomber scombrus*) caught from the North Sea and Baltic Sea were sampled for plastics by Rummel et al. (2016). Of the 290 investigated fish, 16 contained plastics (approximately 74% microplastics and 26% macroplastics). Macroplastics and microplastics were found in the GI tracts of 47.7% of the coastal fish and 2.4% of the offshore fish collected from Scottish marine waters by Murphy et al. (2017), or 29.7% (n=63) of all fish sampled. The mean number of plastic pieces found per fish was 1.8, with PA being the most common polymer. Choy and Drazen (2013) also found plastics in the stomachs of seven different species of pelagic fish from the central North Pacific Subtropical Gyre, many of which were macroplastics.

Schuyler et al. (2014) conducted a global analysis of plastic ingestion in various sea turtle species and found that the most commonly ingested anthropogenic pollutants were plastics. Plot and Georges (2010) reported a field observation of an adult leatherback turtle that expulsed 2.6 kg of plastic

pollutants, consisting primarily of plastic bags and plastic fragments. Plastics have also been found in green sea turtles (*Chelonia mydas*) (Özdilek et al. 2006; Stamper et al. 2009).

Lusher et al. (2015b) studied two adult and one juvenile True's beaked whales (*Mesoplodon mirus*) that were found stranded on the coast of Ireland. Analysis of the contents of their stomachs and intestines revealed that both adults appeared to have ingested macroplastics, but it could not be determined whether the whales died as a direct consequence of plastic ingestion (Lusher et al. 2015b). Marine litter was also found in the stomachs and intestines of 26 out of the 175 (approximately 15%) dead Magellanic penguins (*Spheniscus magellanicus*) collected from the Brazilian coastal zone, roughly 58% of which was plastics (Brandão et al. 2011).

Gall and Thompson (2015) reported occurrences of marine litter ingestion for 13 110 individuals of 208 species, and Kühn et al. (2015) reported that the number of species known to ingest plastics increased by approximately 87% from 1997 to 2015 (177 to 331 species) and that marine litter ingestion has been recorded in 50.4% of marine mammal species, 40.4% of seabird species, and 100% of turtle species. However, cases of plastic ingestion leading to direct harm or death is less frequent in comparison to entanglement. Gall and Thompson (2015) found that only 4% of reported cases of ingestion resulted in direct harm or death. In contrast, Rochman et al. (2016) found that 63% of deaths were due to ingestion of marine litter. Specifically, demonstrated impacts from ingestion were observed in marine mammals (two species), sea turtles (one species), seabirds (one species), and marine invertebrates (two species).

Ingestion of plastics by organisms has been shown to have consequences from several pathways. Current literature shows that the most clear adverse effects from plastic ingestion is the blockage of intestinal systems, preventing feeding and thus leading to possible starvation. For example, a common dolphinfish (*Coryphaena hippurus*) caught in the Western Equatorial Atlantic had a large plastic bowl measuring 99.57 cm^2 in its stomach (Menezes et al. 2019). Researchers suggested that the bowl was likely blocking its digestive tract, leading to starvation. A study by Pierce et al. (2004) reported plastic ingestion by a male northern gannet (*Morus bassanus*) and a female greater shearwater (*Puffinus gravis*). Both birds had blockages of the pylorus, which prevented feeding, leading to starvation and death. Ulcerations near the pylorus were also seen in the northern gannet, which matched up exactly with the shape of the bottle cap found in its esophagus that was thought to have been dislodged from the gizzard.

Ingested plastics can also damage organs and intestinal systems. Brandão et al. (2011) observed a dead Magellanic penguin (*Spheniscus magellanicus*) whose stomach had been perforated by a plastic straw. Jacobsen et al. (2010) studied two sperm whales (*Physeter macrocephalus*) post-mortem, both of which had netting, fishing line, and plastic pollutants such as bags in their stomachs. The cause of death in both whales was suspected to be gastric impaction, as one whale had a ruptured stomach and the other was emaciated. Stamper et al. (2009) observed an emaciated green sea turtle (*Chelonia mydas*) floating off the coast of a Florida beach. The turtle displayed signs of cachexia, lethargy, increased buoyancy, obstipation, and anorexia. Radiographs confirmed the presence of plastics in the GI tract, hindering regular function. After the removal of 74 foreign objects (including latex balloons, string, nylon rope, and soft and hard plastics) via enemas, the turtle showed improvements in its health, appetite, and

behaviour. The authors note that this demonstrates a cause-and-effect relationship between plastic ingestion and morbidity in organisms (Stamper et al. 2009).

6.1.3 Habitat integrity and rafting (organism transport)

The presence of plastic pollution in water bodies can also pose potential problems for ecosystem function, biodiversity, and habitat integrity (Werner et al. 2016). An increasing amount of plastic pollution in surface waters has the potential to act as a stressor on ecosystem dynamics and habitat integrity (CBD 2012).

Plastics can be effective transport mediums due to their potential for surface adhesion and to the low density of certain types of plastic and can potentially accentuate transport of organisms or other organic matter, a phenomenon known as “rafting” (Werner et al. 2016). This process can also occur with naturally occurring material such as wood, but the increasing prevalence of plastic pollution in surface waters increases the likelihood for organisms to be transported, which can pose a threat to the receiving environment. Gall and Thompson (2015) identified 34 reports of organisms rafting on marine litter, including packaging, fragments, and intact items (plastic or otherwise). Of the 259 total species described in these reports, six were listed as being invasive (i.e., non-native). However, the authors note that this is likely an underrepresentation (CBD 2012; Gall and Thompson 2015). The transport of non-native species is a particular concern, as they have the potential to negatively impact the structure of other well-established ecosystems by becoming predators to native species and/or outcompeting them for resources, leading to a loss of biodiversity (Werner et al. 2016). Non-native species could also transport diseases to which native species have not previously been exposed and could alter the genetic diversity in the ecosystem. Furthermore, plastic pollutants can also act as an artificial habitat for the colonization and growth of microorganisms that can affect species assemblage (Werner et al. 2016).

Katsanevakis et al. (2007) studied the impacts of marine litter on the abundance and community structure of epibenthic megafauna in the Aegean Sea. They demonstrated that an increase in marine litter caused a marked and gradual increase in both the total abundance and number of species, changing the structure of the megafaunal community. This was attributed to the fact that the litter was able to provide refuge for mobile species and to act as a colonization site for hard-substratum sessile species. This change in dynamics can have significant long-term effects on the ecosystem, such as altered predator-prey dynamics.

6.2 Microplastic

There are no standardized methods for testing the effects of microplastics. Currently, concentrations of microplastics used in effect studies are much higher than those measured in the environment (Burns and Boxall 2018). Furthermore, effects studies focus on particle sizes much smaller than those currently sampled for in the environment (SAPEA 2019). Particle concentration can also influence toxicity, as higher concentrations are expected to overwhelm biological clearance mechanisms and cause responses that are not otherwise observed at lower doses (WHO 2019). Results from Pikuda et al. (2019) indicate that preservatives in commercial plastic formulations, rather than the plastic particle itself, may be

responsible for the observed acute toxicity to test organisms. However, the washing of test particles is not currently standard practice and therefore this was not considered in the above criteria.

For the purposes of this report, the following criteria were used to select the studies: the study reported details of the analytical techniques, the study reported the type of plastic used (i.e., polymer type, size, shape, virgin vs. aged), and the study monitored and reported measured concentrations that were similar to the nominal (i.e., theoretical) concentrations. Similar to the environmental occurrence section, these qualitative criteria draws upon the quantitative criteria presented by Hermsen et al. (2018) for the determination of study quality in papers examining the ingestion of microplastics by biota. However, as acknowledged in this paper and in Koelmans et al. (2019), these criteria are not an absolute judgment of the value of studies, because not all aspects of studies could be captured in the scoring systems. As such, if any studies included in this report deviated from the above criteria, the limitation is explicitly mentioned in the text. Furthermore, studies in this section were selected in order to cover a variety of organism types and effects.

6.2.1 Uptake, ingestion, and egestion

Microplastics have been found in many species, including invertebrates, fish, turtles, mammals, and birds. Given the lack of standardized methods for quantifying occurrence in biota as well as the limited data on occurrence in Canadian species, criteria for selecting reliable studies (e.g., studies that used an analytical method to identify microplastics) were identified but many studies did not meet these standards. Moving forward, it is recommended that a standardized method for quantifying microplastics in biota be developed.

A review by Provencher et al. (2017) showed that the literature on global macroplastic and microplastic ingestion in marine vertebrates is dominated by seabirds and that there is an increasing number of reports in fish, turtles and mammals each year. Fibres and fragments are the most common microplastic types found in organisms (Burns and Boxall 2018). For example, Beer et al. (2018) visually identified microplastics in 20% of the 814 fish they studied in the Baltic Sea, with 93% of these being fibres. Collicutt et al. (2019) determined by light microscopy that over 90% of the microplastics they found in juvenile Chinook salmon (*Oncorhynchus tsawytscha*) were fibres.

As in the case of macroplastics, several factors can affect the intake and ingestion of microplastics by organisms. In laboratory studies, Scherer et al. (2017) demonstrated that co-exposure of microplastics with algae significantly decreased ingestion of microplastics by *Daphnia magna*. Weber et al. (2018) demonstrated that exposure concentration and age of the freshwater amphipod *Gammarus pulex* affected its microplastic body burden. Feeding selectivity of biota is also thought to be a driving factor for microplastic ingestion: non-selective filter feeders are more prone to direct microplastic uptake, whereas more specialized feeders will uptake microplastics indirectly through ingestion by their prey (Scherer et al. 2018). Uptake of microplastics via prey ingestion is discussed further below. Select reported ingestion events are outlined below, with Canadian and global examples.

Liboiron et al. (2019) studied the GI tracts of three fish species commonly used for human consumption on the island of Newfoundland: Atlantic cod (*Gadus morhua*), Atlantic salmon (*Salmo salar*) and capelin (*Mallotus villosus*). The frequency of occurrence of macroplastic and microplastic ingestion by Atlantic salmon and capelin was 0% for specimens collected between 2015 and 2016 (a total of 419 fish). In Atlantic cod examined during the same period, the frequency of occurrence of plastic ingestion was 1.68%. These results are consistent with a previous study by Liboiron et al. (2018), in which 134 silver hake (*Merluccius bilinearis*) from the south coast of Newfoundland were studied and found to have a 0% frequency of occurrence of plastic ingestion.

In a study of microplastics in the GI tract of juvenile Chinook salmon on the east coast of Vancouver Island, Collicutt et al. (2019) found that 59% of the specimens examined contained at least one plastic particle, with an average of 1.15 microplastic pieces per individual. It should be noted that plastic identification was not confirmed using an analytical method other than visual identification using light microscopy.

In a study of microplastics in fish from a prairie creek downstream from a WWTS in Regina, Saskatchewan (Campbell et al. 2017), five species of fish were collected: fathead minnow (*Pimephales promelas*), northern pike (*Esox lucius*), white sucker (*Catostomus commersoni*), emerald shiner (*Notropis atherinoides*), and five-spine stickleback (*Eucalia inconstans*). Of the 181 fish sampled, 73.5% had between 1 and 20 microplastics in their GI tracts. The number of microplastics varied significantly between the five species sampled. This inter-species variation is hypothesized by the authors to be attributable to differences in feeding habits. The northern pike, an apex predator, had the highest proportion of sampled fish with microplastics present in their GI tracts at 83.3%, while the fathead minnow had the lowest, at 50.0%. The authors acknowledge that characterization of plastics using spectroscopic identification methods was not performed in this study. However, a hot needle was used to test whether the suspected plastic particles melted, to confirm that the particle was plastic (Campbell et al. 2017). It should be noted that some types of plastic will not melt under these conditions (i.e., thermosets).

O'Hara et al. (2019) conducted a study on the seasonal variability of exposure of Cassin's auklets (*Ptychoramphus aleuticus*) to microplastic pollution. Following a series of storm events, 707 carcasses were found on the beaches of Vancouver Island and Haida Gwaii in British Columbia. A total of 85 carcasses were collected for examination, and plastics were found in the stomachs of 40% of the birds. Macroplastic and microplastic pieces in the stomachs of the birds were visually identified and separated. The average number of plastic pieces ingested per bird was 1.6, with an average mass of 0.0085 g, and one outlier ingested 61 pieces of plastic. Furthermore, ingested plastics were predominantly microplastics (86.6%). There was no significant difference between the number of pieces ingested by age, sex, or health condition of the bird (O'Hara et al. 2019). Similarly, Poon et al. (2017) studied plastic ingestion by Northern fulmars (*Fulmarus glacialis*) in the Canadian high Arctic. None of the stomachs of Northern fulmars sampled in 2013 contained more than 0.1 g of visually identified plastics. Provencher et al. (2018a) demonstrated that Northern fulmars excreted microplastics via their guano and found that the number of pieces of plastic in the gut was positively related to the number of microplastics in the guano.

Plastics have been identified in organisms from several regions of the world. Representative studies are presented below to demonstrate that microplastic ingestion by biota occurs globally. An exhaustive review has been conducted by Provencher et al. (2018b).

Microplastics have been found in gudgeons (*Gobio gobio*) in Flemish rivers in Belgium (Slootmaekers et al. 2019). Gudgeons from 15 rivers at 17 locations were sampled to study the occurrence of microplastics in their intestines. Microplastic contamination was found in four of the rivers studied. Of the 78 fish examined, 9% contained microplastic particles in their intestines, and only one fish had ingested more than one particle. A total of 16 suspected plastic particles were extracted from all sampled fish; however, only eight particles were identified to be plastic following μ-FTIR analysis. Overall, seven different polymers were identified: ethylene-vinyl acetate copolymer, PP, PET, PVC, cellophane, polyvinyl acetate and PA (Slootmaekers et al. 2019).

In the heavily industrialized city of Tuticorin, India, Kumar et al. (2018) investigated the occurrence of microplastics in Indian mackerel (*Rastrilliger kanagurta*) and honeycomb grouper (*Epinephalus merra*) on the southeastern coast. Of the 40 fish sampled, 12 had plastic particles in their intestines. FTIR analysis revealed that the particles were PE and PP. Fibres constituted 80% of particles, whereas fragments constituted the remaining 20%.

While the ingestion of microplastics has been widely demonstrated, egestion has also been shown to be possible in some organisms. For example, Grigorakis et al. (2017) found that goldfish (*Carassius auratus*) have efficient gut clearance of microbeads and microfibres: the time required for 90% clearance was 33.4 hours. Mazurais et al. (2015) found complete egestion of PE microbeads from European seabass (*Dicentrarchus labrax*) larvae after 48 hours. In invertebrates, significant microplastic egestion was seen in studies by Chua et al. (2014), Blarer and Burkhardt-Holm (2016), Frydkjær et al. (2017), and Hämer et al. (2014). In *Hyalella azteca*, an amphipod crustacean, microplastic fibres were found to be more slowly egested than microbeads during acute exposure; however, both were able to be completely egested (Au et al. 2015).

6.2.2 Ecotoxicological effects

Despite the ability of some organisms to egest plastic particles, microplastics have been shown in the current literature to have adverse effects on organisms. In their respective literature reviews, Rochman et al. (2016) and the Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) (2016) reported that, in the laboratory, the effects from micro-sized litter (consisting primarily of plastic) were overwhelmingly seen at the suborganismal level. The predominant observed effects at this level were in macromolecules, cells, and tissues and can include inflammation and changes in gene expression (Rochman et al. 2016; GESAMP 2016). The remaining demonstrated effects were at the organismal level, primarily due to individual deaths (Rochman et al. 2016). In addition, Foley et al. (2018) conducted a meta-analysis of 43 papers published before October 2016 and observed that while effects from microplastic exposure were highly variable across taxa, the most consistently reported effect across both marine and freshwater taxa was a reduction in the consumption of natural prey.

In the Government of Canada's 2015 science summary on microbeads (ECCC 2015), 130 publications on the fate and effects of microplastics were examined and reviewed. Several key studies were summarized in the assessment report. The report noted a scarcity of information on long-term and multigenerational effects of microbeads; however, short-term and direct effects are well described. Physical effects were identified as the primary driver for effects to organisms. Some examples of effects in organisms from microbead exposure that have been described in literature include: decreased survival and fecundity (Lee et al. 2013), decreased reproduction from impedance of feeding behaviour (Cole et al. 2015), liver stress (Rochman et al. 2013), altered gene expression (Rochman et al. 2014), and possible genotoxicity in the form of DNA damage (Avio et al. 2015). Au et al. (2015) found that acute exposure to microfibres produced greater toxicity (due to physical effects) to *Hyalella azteca* than spherical beads, with 10-day median lethal concentration (LC_{50}) values of 71.43 microfibres/mL and 4.64 x 10^4 microbeads/mL, respectively. Hämer et al. (2014) observed no impact on survival, growth, and intermolt duration in isopods (*Idotea emarginata*) following chronic exposure to microplastic particles of multiple forms. More detailed summaries of these studies can be found in ECCC (2015).

In its proposal for a restriction on intentionally added microplastics, the European Chemicals Agency (ECHA) reviewed and summarized 25 influential scientific papers on the ecotoxicological effects of microplastics (ECHA 2019). The papers include data that overlap with those from studies cited in ECCC (2015). Experimental data cited by the ECHA in its proposal that were not discussed in ECCC (2015) are summarized briefly below. For more detailed summaries of these studies, please refer to ECHA (2019).

- Earthworms (*Lumbricus terrestris*) showed a decrease in growth rate with exposure to high concentrations of LDPE particles (<150 µm; 28, 45, 60% dw), but reproduction was not affected (Huerta Lwanga et al. 2016).
- Zebrafish (*Danio rerio*) exposed to PS microspheres (5 µm) exhibited inflammation, lipid accumulation in liver, oxidative stress, and altered metabolomics profiles (Lu et al. 2016).
- European sea bass (*Dicentrarchus labrax*) feeding on PVC pellets (<0.3 mm; 1.4% bw) had significant structural damage to the intestine (Pedà et al. 2016).
- *Daphnia magna* that ingested PE particles (1 µm; 12.5 to 200 mg/L) experienced immobilization that increased with concentration and time following 96-hour exposure (Rehse et al. 2016).
- Pacific oysters (*Crassostrea gigas*) exposed to PS spheres (2 and 6 µm; 0.023 mg/L) had significant reductions in oocyte number, oocyte diameter, sperm velocity, and larval development of offspring following two-month exposure (Sussarellu et al. 2016).
- Mussels (*Mytilus edulis*) and lugworms (*Arenicola marina*) exposed to PS microspheres (10 µm, 30 µm, 90 µm; 110 particles/mL seawater for mussels, 110 particles/g for lugworms) showed increased metabolism, but no adverse effects on energy allocation (Van Cauwenberghe et al. 2015).
- Common shore crabs (*Carcinus maenas*) feeding on PP microfibres (1 to 5 mm in length; 1% plastic) showed a decrease in food consumption rates over time and a drastic reduction in energy available for growth, with minimal lasting consequences (Watts et al. 2015).
- Marine worms (*Arenicola marina*) had reduced feeding activity and reduced available energy reserves from ingestion of unplasticized PVC treatments (130 µm mean diameter; 5% by weight) (Wright et al. 2013).

A review of additional current literature on ecotoxicological effects of microplastics is provided below for each environmental compartment of interest. Relevant studies are outlined in the sections below, with more detailed summaries provided in Appendix D, including information on the size, concentration and polymer type of the particles. Due to physicochemical similarities, information on primary microplastics was used as surrogate information where information on secondary microplastics was not readily available.

Water

The aquatic environment and marine organisms in particular, have been the focus of much of the ecotoxicological research on plastics (SAPEA 2019). In freshwater studies, invertebrates have been the focus of research on sensitivity to microplastic exposure (Adam et al. 2019).

Studies on the effects of plastics on organisms in both freshwater and marine environments are presented below, by level of biological organization.

Vertebrates

Yin et al. (2018) exposed the fish species *Sebastes schlegelii* to PS spheres and observed a reduction in foraging time and swimming speed, an increase in shoaling behaviour, and a feeding time of almost twice that of the control. Goldfish (*Carassius auratus*) exposed to ethylene vinyl acetate fibres, PS fragments, and polyethylene acrylate pellets also exhibited sublethal effects such as weight loss, histological changes to the GI tract and intestines, inflammation of the liver, and physical damage to the jaw, including incisions from chewing fragmented particles, but no mortality (Jabeen et al. 2018).

Similarly, in zebrafish (*Danio rerio*) exposed to PA, PE, PP, and PVC microplastics, no significant difference in lethality was observed; however, microplastics caused intestinal damage such as cracking of villi and splitting of enterocytes (Lei et al. 2018a). A study by Qiao et al. (2019a) reported similar findings when using PS microplastics, such that zebrafish exposed to virgin PS beads had significant intestinal damage, inflammation, oxidative stress, and altered gut microbiomes.

At the molecular level, Qiang and Cheng (2019) found that exposure to PS microplastics induced upregulated expression of inflammation and oxidative stress-regulated genes in zebrafish larvae. *S. schlegelii* showed a significant reduction in crude protein and lipid contents and had black bile in their gallbladders, indicating GI function disorder resulting from accumulation of PS spheres in their intestinal tract (Yin et al. 2018).

Conversely, several current studies report no significant effects on vertebrates for any of the endpoints measured. De Felice et al. (2018) exposed tadpoles of African clawed frog (*Xenopus laevis*) to PS microplastics and found no significant effects on mortality, body growth, or swimming activity during their early life stages, despite observing microplastics in the digestive tracts of all exposed tadpoles. Further, Ašmonaitė et al. (2018) observed no significant histological effects or inflammatory responses in rainbow trout (*Oncorhynchus mykiss*) exposed to PS microplastics, and Jacob et al. (2019) observed no effects on foraging or predation avoidance in coral-reef fish (*Acanthurus triostegus)* exposed to PS

microbeads. Dietary exposure to PVC, PA, PE, and PS microplastics also did not affect stress responses, growth rate, or induce pathology changes in seabream (*Sparus aurata*) (Jovanović et al. 2018).

Invertebrates

In cladocerans (*Daphnia* spp.), increased microplastic concentrations led to increased mortality (Aljaibachi and Callaghan 2018; Martins and Guilhermino 2018; Pacheco et al. 2018). However, Jaikumar et al. (2018) suggest that mortality might also be temperature-dependent. Martins and Guilhermino (2018) further observed that microplastic exposure could have transgenerational effects in *D. magna*. Females descending from groups exposed to microspheres showed reduced growth, reproduction, and population growth rates up to the F_3 generation, indicating that complete recovery from chronic exposure may take several generations for this species. Tang et al. (2019) found no mortality in *Daphnia* exposed to PS particles, but observed a reduction in body growth rate and increased transcription of arginine kinase and permease (enzymes involved in oxidative defence and energy production).

Freshwater crabs (*Eriocheir sinensis*) exposed to PS microspheres similarly showed a decrease in weight gain, reduced hepatosomatic index, and several biochemical effects, such as an increase in transcription of genes involved in the oxidative stress response and anti-inflammation pathways (Yu et al. 2018).

Similarly, Jeong et al. (2017) found that exposing the marine copepod *Paracyclopina nana* to PS microbeads increased antioxidant enzyme activity in a size-dependent manner. A previous study by Jeong et al. (2016) found similar results when studying the monogonont rotifer *Brachionus koreanus*: several antioxidant enzymes showed increased activity in rotifers exposed to PS microbeads, indicating a defence mechanism against oxidative stress.

Beiras et al. (2018) studied rotifers as well as the crustacean *Tigriopus fulvus* and determined lowest observed effect concentrations (LOECs) of 0.01 mg/L for rotifer immobility and 1.0 mg/L for rotifer and crustacean mortality, using PE particles.

For the freshwater invertebrate *Gammarus pulex,* Weber et al. (2018) found no significant effects on juvenile survival, development (molting), metabolism, or feeding activity following chronic exposure to PET. Another study by Redondo-Hasselerharm et al. (2018) exposed *G. pulex* to PS microplastics. While the survival of *G. pulex* was not affected, the organisms experienced a significant reduction in growth, with a 28-day EC_{10} (10% effect concentration) of 1.07% plastic weight in sediment dw.

Studies have also been conducted on coral species. Chapron et al. (2018) found that marine corals (*Lophelia pertusa*) exposed to LDPE microbeads had significantly lower prey capture rates and reduced skeletal growth rates and calcification compared to the controls. Hankins et al. (2018) found no significant effects on calcification in either the large polyp coral *Montastraea cavernosa* or the small polyp coral *Orbicella faveolata* despite active ingestion of PE microbeads.

Microplastic exposure has also been studied at early developmental stages for invertebrates. Lo and Chan (2018) found that larval and juvenile sea snails (*Crepidula onyx*) were not affected by exposure to environmentally-relevant concentrations of PS particles. At higher concentrations, the larvae grew

slower and settled at a smaller size compared to control conditions. In addition, individuals exposed to microplastics only in their larval stages displayed slower growth rates even after the removal of the microparticles, indicating a possible legacy effect (Lo and Chan 2018). Similarly, blue mussel (*Mytilus edulis*) larvae with PS bead exposure experienced no changes in growth rate; however, there was an increase in the amount of abnormally developed larva (Rist et al. 2019). Beiras et al. (2018) found no significant effect on mussel embryonic development under static conditions from virgin PE microplastics.

Primary producers

Green algae (*Chlorella pyrenoidosa*) exposed to PS beads displayed inhibited growth rates that corresponded to increasing plastic concentration (Mao et al. 2018). Reduced photosynthetic activity and damaged cell membranes were also evident; however, a recovery of algal biomass and photosynthetic activity was seen during the later phases of growth, which may be linked to detoxification mechanisms. Additionally, Gambardella et al. (2018) found that green microalga (*Dunaliella tertiolecta*) exposed to PS microbeads experienced a dose-dependent inhibition of growth: inhibition reached 40% at the highest concentration.

Current studies also show an absence of significant effects on primary producers for endpoints tested. Sjollema et al. (2016) exposed both freshwater and marine microalgal species to uncharged virgin PS microbeads and negatively-charged beads and found an absence of significant effects on photosynthesis from exposure to all treatments. Further, Garrido et al. (2019) found no effect on the daily growth rate of the microalgae *Isochrysis galbana* exposed to PE particles at any of the tested concentrations.

Soil

Experimental studies involving biota in the soil compartment are limited, but the studies that do exist show that microplastic exposure can negatively impact organism health and behaviour.

Ju et al. (2019) showed that exposing soil springtails (*Folsomia candida*) to PE microplastics for 28 days led to an increase in avoidance behaviours and an inhibition of reproduction rate by up to 70.2% at the highest exposure concentration. Additionally, the exposed springtails had significantly decreased bacterial diversity in their guts. Similarly, Kim and An (2019) found that microplastic infiltration into soil system bio-pores caused movement inhibition in the invertebrate *Lobella sokamensis*.

PS microplastics also caused toxicity to the soil invertebrate *Caenorhabditis elegans* following a three-day exposure period (Lei et al. 2018b). Nematodes exposed to 1.0 µm PS particles had lower survival rates, shorter average lifespans, decreased average body lengths, and significant damage to GABAergic neurons in comparison to the other microplastic sizes tested.

Sediment

Although the sediment compartment has also been less studied than the water compartment, the current literature indicates that microplastics may have adverse effects on sediment-dwelling organisms.

Ziajahromi et al. (2018) exposed sediment-dwelling midge (*Chironomus tepperi*) larvae to four different size ranges of virgin PE microplastics to assess development. They concluded that midge survival was size-dependent; organisms exposed to microplastics that were similar in size to their normal food sources (10 to 27 µm) had a survival rate of 57% compared to 92% in the negative control group, as well as significantly smaller body sizes and head capsule lengths. Further, Leung and Chan (2018) found that PS microplastics significantly increased mortality and decreased body part regeneration in polychaetes (*Perinereis aibuhitensis*) after a four-week exposure period in a size-dependent manner. In addition, sediment-dwelling bivalves (*Ennucula tenuis, Abra nitida*) exposed to fragmented PE microplastics in three size classes displayed a dose-dependent decrease in energy reserves; however, no significant mortality was observed (Bour et al. 2018). The exposed *E. tenuis* also had significantly lower lipid content for only one condition, while lower protein content was observed in *A. nitida* from exposure to the largest particles at all concentrations.

Nematodes (*Caenorhabditis elegans*) exposed to PA, PE, PP, and PVC microplastics had decreased survival rates, body length, and reproduction, as well as reduced calcium levels and increased expression of enzymes, indicating oxidative stress and intestinal damage (Lei et al. 2018a).

In contrast to the above-summarized research, the current literature also contains studies that show an absence of adverse effects on organisms exposed to microplastics in sediment. Redondo-Hasselerharm et al. (2018) observed no significant effects on survival or growth of the freshwater benthic macroinvertebrates *Hyalella azteca, Asellus aquaticus, Sphaerium corneum,* and *Tubifex* spp. from exposure to PS microplastics. Further, they observed no effects on the reproduction of the freshwater worm *Lumbriculus variegatus*.

6.2.3 Trophic transfer

There is limited information on the ability of microplastics to travel through different trophic levels, as seen in a food chain. Very few studies have looked at trophic transfer, and even fewer have studied the importance of bioconcentration, biomagnification, and bioaccumulation (Provencher et al. 2018b). Hammer et al. (2016) conducted one of the few studies that demonstrate vertical transfer of plastic particles within a food web. In that study, plastics found in the guts of great skuas (*Stercorarius skua*) from the Faroe Islands corresponded to the plastic contents of their prey (surface-feeding seabirds), implying indirect consumption.

Additionally, Cuthbert et al. (2019) demonstrated transference of microplastics in predatory midge larvae (*Chaoborus flavicans*) that consumed mosquito (*Culex pipiens)* larvae exposed to 2 µm PS microplastics. They found that the amount of microplastics transferred correlated with feeding rates towards mosquito larvae.

To study transfer along a natural food chain, Batel et al. (2016) exposed nauplii of the brine shrimp *Artemia* to microplastics ranging from 1 to 5 µm or from 10 to 20 µm, then fed the nauplii to zebrafish (*Danio rerio*). They observed that, while the zebrafish were able to uptake the microplastic particles, no significant accumulation or further retention was observed within their intestinal tract, and no transfer

to other organs was observed. Similarly, Welden et al. (2018) found by examination of stomach contents that trophic transfer of microplastics occurred between sand eels (*Ammodytes tobianus*) and their predator, plaice (*Pleuronectes platessa*) from the Celtic Sea. However, the microplastics were egested in the plaice.

Some studies suggest that unintentional ingestion, rather than trophic transfer, is the primary means by which microplastics are ingested. Chagnon et al. (2018) found no accumulation of microplastics in stomachs of yellowfin tuna (*Thunnus albacares*), a large predatory fish from Easter Island, despite plastics being found in the guts of its prey. Hipfner et al. (2018) also concluded that two fish species from the northeastern Pacific Ocean, the Pacific sand lance (*Ammodytes personatus*) and the Pacific herring (*Clupea pallasii*), do not act as significant conduits for the vertical transfer of microfibres to marine piscivores along the coast of British Columbia.

6.2.4 Translocation

While mechanisms of translocation from an organism's gut to other parts of its body are not well studied to date, the current literature has shown that translocation is usually size-dependent. For example, Lu et al. (2016) found that particles less than 5 µm can translocate to fish liver from the gut, while 20 µm particles cannot (Jovanović 2017). Smaller particles have the potential to more easily enter the circulatory system, but can also be egested more easily than larger microplastic particles (Jovanović 2017; Burns and Boxall 2018).

Current studies show that translocation occurs in some organisms and organs, while other studies contradict these findings. For example, translocation of 0.5 µm PS spheres to the haemolymph, gills, and ovary was observed in crabs (*Carcinus maenas*) (Farrell and Nelson 2013). In zebrafish (*Danio rerio*), Lu et al. (2016) found 5 µm PS particles in the gills, liver, and gut, while 20 µm particles were only found in the gills and gut. In bivalves, tissue translocation of 3.0 or 9.6 µm PS spheres from the digestive tract to the circulatory system was seen in mussels (*Mytilus edulis*) by Browne et al. (2008). However, a study by Sussarellu et al. (2016) using Pacific oysters (*Crassostrea gigas*) showed no evidence of PS sphere (2 and 6 µm) translocation. Limited information in fish also shows very small amounts of microplastics in fish muscle (Karami et al. 2017a; Abbasi et al. 2018; Akhbarizadeh et al. 2018).

The conflicting results observed in these studies may be attributable to species-specific differences and/or false positive results that may occur as a result of leaching of fluorescent dye, which is often used to track particle ingestion. Schür et al. (2019) tested this theory and found that fluorescent droplets did not always co-localize with the plastic PS beads ingested by *Daphnia magna*. Using confocal laser scanning microscopy, 1 µm beads did not co-localize with the fluorescent dye in the gut and there was a rapid loss of fluorescence upon investigation. Fluorescence was also observed in lipid droplets outside of the digestive tract, but plastic particles were not detected in these same lipid droplets. Therefore, given that false positives may occur in uptake studies where precautions were not taken to avoid potential artifacts by ensuring the stability of dyes, controlling for dye leaching (e.g., by pre-washing the particles), or using microscopic imaging to confirm plastic presence, results should be interpreted with caution.

7. Impacts on human health

7.1 Macroplastic

While people regularly observe and interact with macroplastics, human exposure to macroplastic pollution is not anticipated to be a concern. The effects of macroplastic pollution on human health are therefore not considered in this report.

7.2 Microplastic

Humans may be exposed to microplastics through the ingestion of food and drinking water (see Section 5.2) and the inhalation of indoor and outdoor air (see Section 5.1.3). The toxicity of microplastics via the ingestion and inhalation routes of exposure is reviewed below. Where possible, inferences are made from epidemiological studies on microplastics in humans and experimental studies on microplastics in animal models. A comprehensive review of in vitro studies on microplastics was not conducted as their relevance to human health is unclear. The effects of biofilms on human health are also discussed.

Upon ingestion or inhalation, microplastics may exert effects due either to their physical presence in the gut or lung or to the chemical composition of the plastic polymers themselves or their monomers, additives or sorbed substances. The World Health Organization (WHO) recently carried out an assessment of human exposure to microplastics in drinking water using conservative worst-case estimates of the levels of additives and sorbed chemicals on microplastics (WHO 2019). The Food and Agriculture Organization of the United Nations (FAO) and the European Food Safety Authority (EFSA) conducted a similar assessment of exposure to microplastics in seafood (EFSA 2016; FAO 2017). These evaluations concluded that exposure to microplastics and/or chemicals associated with microplastics are considered to be a low concern to human health (EFSA 2016; FAO 2017; WHO 2019). The reader is referred to those reports for further information on the exposure and risk assessments conducted therein.

7.2.1 Effects from oral exposure

Physicochemical properties affecting uptake and toxicity

Following ingestion, microplastic uptake and translocation are strongly dependent on the physicochemical properties of the ingested particles (FAO 2017; Wright and Kelly 2017; WHO 2019). Particle size is an important determinant of absorption through the intestinal epithelium. Smaller particles have larger surface-area-to-volume ratios, which can increase their ability to translocate to internal organs and increase bioreactivity (WHO 2019). A higher surface-area-to-volume ratio may also increase the sorption capacity of microplastics for environmental contaminants. Smaller particles may also be more susceptible to fragmentation, and while degradation of microplastics to smaller polymers has been demonstrated in the GI tract of Antarctic krill (Dawson et al. 2018), it is uncertain whether this occurs within the human GI tract (WHO 2019). Particle concentration can also influence toxicity, as

higher concentrations are expected to overwhelm biological clearance mechanisms and cause responses that are otherwise not observed at lower doses (WHO 2019). At present, it is unclear how other properties, such as shape and surface chemistry, may affect the uptake, retention, and/or toxicity of ingested microplastics (Stock et al. 2019; WHO 2019).

Toxicokinetics

There are limited data regarding the fate of orally ingested microplastics in mammalian species. Available literature suggests that following oral ingestion, microplastics may remain confined to the GI tract, translocate from the GI tract into organs or tissues, and/or be excreted (EFSA 2016; FAO 2017).

Several uptake mechanisms have been proposed for microplastics, including endocytosis via microfold cells (M cells) of the intestinal Peyer's patches and paracellular persorption (see EFSA 2016, FAO 2017, and Wright and Kelly 2017 for an extensive review of the toxicokinetics of microplastics). Based on limited data, it is expected that the largest fraction of orally ingested microplastics (>90%) will be excreted in the feces (EFSA 2016; FAO 2017). Microplastics greater than 150 µm are also expected to remain confined to the gut lumen and be excreted, while only limited uptake is expected for smaller particles (EFSA 2016; FAO 2017; WHO 2019). Various types of microparticles have been shown to translocate across the mammalian GI tract into the lymphatic system at sizes ranging from 0.1 to 150 µm (Hussain et al. 2001; EFSA 2016; FAO 2017). For example, in one study, PVC microplastics (5 to 110 µm) were detected in the portal veins of dogs (Volkheimer 1975). Given these findings, it is possible that microplastics less than or equal to 150 µm may end up in the lymphatic system and result in systemic exposure, although absorption is expected to be low (≤0.3%; EFSA 2016; FAO 2017). Only very small microplastics (<1.5 µm) are expected to enter into capillaries and penetrate deeply into tissues (Yoo et al. 2011; EFSA 2016). This is consistent with a recent 28-day study in which mice were administered high concentrations of a mixture of PS microplastics of various sizes by oral gavage three times per week (Stock et al. 2019). Only a few microplastics were detected in the intestinal walls (no quantitative analysis completed), representing a very low uptake by the GI tissue, and no microplastics were found in the liver, spleen or kidney. Conversely, another study reported significant translocation of 5 µm and 20 µm PS microplastics to the liver and kidney in mice (Deng et al. 2017), although these data are of questionable quality due to notable limitations in study design, data reporting, and biological plausibility of results (Tang 2017; Böhmert et al. 2019; Braeuning 2019). Based on a single human ex vivo placental perfusion model, fluorescently-labelled PS beads less than 240 nm may be taken up by the placenta (Wick et al. 2010).

Studies in humans

No epidemiological or controlled dose studies that evaluated the effects of orally ingested microplastics in humans were identified.

Studies in experimental animals

A small number of animal studies have evaluated the potential adverse effects of orally ingested microplastics (Merski et al. 2008; Mahler et al. 2012; Deng et al. 2017, 2018; Lu et al. 2018; Rafiee et al.

2018; Jin et al. 2019; Stock et al. 2019). Studies were limited to a few types of virgin microplastics and tested either unknown or high concentrations of microplastics that were not necessarily reflective of anticipated human exposure. Test concentrations in toxicity studies are orders of magnitude higher than would be anticipated for humans. Therefore, it was not possible to adequately evaluate the health risk of orally ingested microplastics with the currently available animal data (EFSA 2016; FAO 2017; Wright and Kelly 2017; WHO 2019). The WHO conducted the most recent review of the toxicological data on microplastics ingestion. Consistent with previous reviews by the EFSA (2016) and FAO (2017), the WHO concluded that there were insufficient data to allow for a robust evaluation of the potential human health risks of ingested microplastics, although there was no information to suggest it represented a potential human health concern (WHO 2019). Relevant toxicological studies are briefly summarized below, with more detailed descriptions, including test concentrations, provided in Table E-1 in Appendix E.

In a 90-day study that was compliant with Organisation for Economic Co-operation and Development (OECD) test methods, rats fed a daily diet that contained up to 5% milled PE and PET fabric exhibited no treatment-related adverse effects on blood parameters, organ weights, or histopathology (Merski et al. 2008). Based on the absence of observed toxicity, the highest test dose was considered to be the no observed effect level (NOEL), equivalent to approximately 2 500 mg/kg body weight (bw)/day (WHO 2019). Fibre concentrations were not reported.

Other studies have reported adverse health effects in mice following the administration of very high oral doses of microplastics, several orders of magnitude above expected microplastic concentrations in food and drinking water (Deng et al. 2017, 2018; Lu et al. 2018; Jin et al. 2019). These studies have been extensively criticized for their lack of reliability and relevance (Böhmert et al. 2019; Braeuning 2019; Tang 2017; WHO 2019; Stock et al. 2019). Exposure to high concentrations of PS microplastics in drinking water was associated with alterations in lipid metabolism, gut microbiota composition, amino acid and bile acid metabolism, mucus secretion, and reduced intestinal barrier function in mice (Jin et al. 2019; Lu et al. 2018). Inflammation and lipid droplets were reported in the livers of mice administered high concentrations of PS microplastics by gavage (Deng et al. 2017), but the presence of these effects cannot be determined due to poor quality histological images (Braeuning 2019). Deng et al. (2017) also reported changes in metabolic profiles suggestive of disturbances in energy and lipid metabolism, oxidative stress, and neurotoxic responses. However, the relevance of these metabolic endpoints in assessing the potential human health effects of microplastics is difficult to interpret (Tang 2017; Braeuning 2019; WHO 2019).

More recently, a 28-day mouse study evaluated the potential adverse effects of a mixture of various sizes of PS microplastics (1, 4 and 10 µm) administered via oral gavage three times per week using male heme oxygenase-1 reporter mice, a transgenic mouse model used to evaluate oxidative stress and inflammatory responses (Stock et al. 2019). In contrast to previous studies (Deng et al. 2017, 2018), the authors reported no evidence of oxidative stress or inflammation. While very high microplastic doses were selected for purposes of consistency with other rodent oral toxicity studies, the selected treatment scheme involved dosing the animals three times per week, which was intended to be more representative of a realistic human exposure scenario. However, given the high level of uncertainty

surrounding human exposure to microplastics, it is unclear whether this experimental dosing regime (i.e., three times per week) was in fact more representative of human exposure than daily dosing regimes.

7.2.2 Effects from inhalation

There may be hazards associated with the inhalation of microplastic particles due to their physical presence in the lung that are independent of chemical-related hazards. The scientific literature demonstrating the specific effects of microplastics on the lung is emerging, but their potential to cause effects in the respiratory tract or to translocate to other tissues remains uncertain. Still, inferences can be made from concepts of particle toxicology. Overall, toxicity related to the physical hazard of particles can include oxidative stress, cytotoxicity, inflammation, translocation to other tissues and, in some exceptionally elevated exposure circumstances, particle overload (elevated alveolar burden of particles that can impair clearance) (Prata 2018). Poorly soluble particles that are not inherently toxic, such as carbon black and TiO_2, have been shown to cause inflammation and tumours in rodents, albeit at very elevated levels of exposure (Borm and Driscoll 2019). Inhalation of fine particles is also associated with adverse respiratory and cardiovascular effects, although it is not possible to draw any conclusions regarding particle-driven effects of microplastics exposure at this time.

The potential toxicity of particles will largely depend on particle size and shape, which will influence their deposition in the respiratory tract, their interaction with biological matrices, their potential to translocate, and the efficiency of particle clearance mechanisms. In general, inhalable particles larger than 10 µm in aerodynamic equivalent size will deposit mostly in the extrathoracic region, whereas particles below 10 µm can reach the tracheobronchial regions of the lung (US EPA 2009). It is expected that the majority of these particles will be removed from the airways by means of mucociliary clearance (i.e., trapping of the particles in mucus and coughing), though such clearance can result in ingestion of the particles and subsequent GI exposure (Gasperi et al. 2018). In theory, small particles below 2.5 µm in size can reach the alveolar region of the lung. These particles are removed through phagocytosis by alveolar macrophages, although there is some conflicting evidence demonstrating that very small particles in the nano-size range can evade alveolar clearance mechanisms and potentially accumulate in the lung, eventually reaching the interstitium (Li N et al. 2016).

In the case of fibres, deposition patterns are more difficult to predict. Given their length, most microplastic fibres are expected to be deposited either in the extrathoracic region or in the upper airways and removed via mucociliary clearance (Gasperi et al. 2018). In general, longer plastic fibres, although flexible, are more likely to be associated with evasion of clearance mechanisms (Prata 2018). The area in which deposition occurs and residency time in the lung will greatly influence physical hazards associated with microfibres. Although there are insufficient data specific to microplastics, the observation of plastic microfibres in lung tissue biopsies of workers from a synthetic textile industry, as well as in healthy and neoplastic lung tissues from lung cancer patients, substantiates the plausibility of pulmonary microplastic retention through inhalation (Pauly et al. 1998; Wright and Kelly 2017).

There are few studies that evaluate microplastic particle translocation from the lung following inhalation. It is possible that microplastics can translocate from the lung to systemic circulation or to the lymphatic system, potentially reaching other tissues. One study has examined translocation following intratracheal instillation in pregnant rats (equivalent to 2.4×10^{13} particles) and revealed systemic translocation to placenta, whole pup, fetal liver, heart and spleen (Fournier et al. 2018). Rats intratracheally instilled with radiolabelled PS particles of 56.4 and 202 nm in size exhibited only a small fraction (<2.5%) of particle translocation into systemic circulation in healthy rats, which increased to 4.7% for the smaller particles in the presence of lipopolysaccharide-induced lung inflammation (Chen et al. 2006). The likelihood of translocation is expected to increase with decreasing particle size and increased residency time as well as in individuals with compromised lung function and during events of inflammation (i.e., due to increased cellular permeability) (Galloway 2015). The alveolar region of the lung is a site of potential concern, in part because smaller particles can penetrate this region of the lung (and because they are, by nature, more reactive due to their high surface area), and in part because of the exchanges with systemic circulation that occur at this site. In the upper airway, particles may diffuse through mucus and reach underlying epithelium, where translocation may occur. However, diffusion through mucus is not expected to occur for insoluble particles such as microplastics. It should be noted that, in rats, ultrafine particles have been shown to reach brain tissue via translocation from the nasal cavity through the olfactory nerve (Oberdörster et al. 2004).

There is a paucity of information on the physical hazards related to inhalation of microplastics. Future studies should focus on confirming and exploring the toxicological mechanisms of the physical hazards associated with microplastics, including their effects on the lung and cardiovascular system and their capacity to translocate to extra-pulmonary tissues.

Studies in humans

In the only controlled dosing studies of microplastics in humans, participants were exposed to printer toner, which was not considered relevant for this evaluation. Epidemiology studies of microplastics in indoor or ambient air could not be found for the general population. Studies on the health effects of microplastics are limited to several occupational epidemiology studies and a lung biopsy study; these studies are summarized below.

Two reviews (Wright and Kelly 2017; Prata 2018) summarized the outcomes of occupational epidemiology studies in individuals who worked with synthetic textiles, nylon flock, and PVC. The studies identified associations between work in these industries and increases in adverse respiratory effects, including airway lesions and fibrosis, decreased pulmonary function, wheezing, dyspnea, inspiratory crackles, chronic cough, chronic mucous production, eye and throat irritation, increased bronchial responsiveness, bronchitis, bronchiolitis, emphysema, asthma, pneumoconiosis, interstitial lung disease, foreign body granulomas, and acute respiratory failure (Wright and Kelly 2017; Prata 2018). Several studies also found associations between work in these industries and cancers of the digestive system and respiratory tract, but not all studies investigating these effects identified the association. Despite the associations between exposures to plastic particulates or fibres and adverse health effects, no firm

conclusions on human health effects can be made owing to confounding variables such as co-exposures with other workplace hazards that could contribute to respiratory effects.

A third review discussed the epidemiological evidence of health effects in women working in plastics manufacturing and processing industries, but did not specifically address microplastics (DeMatteo et al. 2012). Epidemiology studies identified associations between work in plastics industries and breast cancer, spontaneous abortion, and infertility. As exposures to microplastics were not specifically discussed in these studies, it is unclear whether associations with these health effects are related to inhalation of plastic particulates and fibres or exposure to other substances used in the production of plastic.

Epidemiology studies have been developed for other occupations with exposure to microplastics. However, most studies limited exposure categorization to occupation, and therefore adverse outcomes from exposure to microplastics were not specifically investigated. A small subset of the epidemiology studies included analyses specifically related to exposure to plastic particulates or fibres; only these studies are discussed here. No increases in lung or respiratory tract cancer were associated with exposure to PU dust in polyurethane foam (PUF) workers (Sorahan and Pope 1993; Mikoczy et al. 2004; Pinkerton et al. 2016). In pattern and model makers, an increase in lymphocytopenia was significantly associated with exposure to plastic dusts, but no exposure–response relationship was observed (Demers et al. 1994).

The relevance of occupational data on airborne microplastics to the general population is unknown, as extrapolation from high-dose occupational exposures to lower doses, as would be expected for the general population, is difficult in the absence of health effect data at lower concentrations. A further limitation of the dataset is that most studies did not investigate the impact of dose-response on the health outcomes. Additionally, workers in the studies might have had co-exposures to other chemicals associated with adverse health effects, such as monomers, catalysts, additives, and other compounds used in the workplace.

Studies in experimental animals

Studies of inhaled microplastics were identified for rats (Laskin et al. 1972; Thyssen et al. 1978; Hesterberg et al. 1992; Warheit et al. 2003; Ma-Hock et al. 2012), hamsters (Laskin et al. 1972), and guinea pigs (Pimentel et al. 1975). The microplastic constituents in the studies included PP fibres (Hesterberg et al. 1992), PU particulate (Laskin et al. 1972; Thyssen et al. 1978), nylon fibres or particulate (Pimentel et al. 1975; Warheit et al. 2003), PAN particulate (Pimentel et al. 1975), and acrylic ester copolymer (Ma-Hock et al. 2012). Exposure duration varied, with one longer-duration study of 325 days, two studies of subchronic duration (12 to 13 weeks), and three studies of subacute duration (5 to 30 exposure days). Detailed descriptions of these studies, including test concentrations and results, are presented in Table E-2 in Appendix E.

Effects observed in inhalation studies tended to be consistent and independent of duration, type of plastic, and species. Observations consistent with foreign body reactions were common in the studies. This included an increase in activity or number of inflammatory cells, which contained fibres or particles

(primarily in lung tissues and bronchoalveolar lavage fluid [BALF], but also in the lymphatic system) and which were often accompanied by granulomas. In areas of lungs associated with particle deposition, hyperplasia, emphysema, and edema were observed. Studies in which animals were euthanized at various timepoints post-exposure tended to indicate a reversibility of effects, suggesting that effects are adaptive rather than adverse responses. No dose-related effects were observed in mortality, survival time, behaviour, clinical observations, tumour incidence, or fibrosis. LOECs adjusted to reflect intermittent exposure ranged from 0.48 to 2.3 mg/m^3. One exception was for the shortest duration study, in which no treatment-related changes in BALF or histology were observed up to the adjusted no observed effect concentration (NOEC) of 2.7 mg/m^3 in rats exposed for 5 days and followed up to 24 days post-exposure (Ma-Hock et al. 2012). However, most studies were not performed according to OECD test guideline methods. Moreover, the human relevance of these animal studies is unclear, as exposures in the studies are much higher than would be expected in humans under typical exposure scenarios.

Inhalation studies are also supported by observations in intratracheal instillation studies in rats. Exposures in the studies were to PVC particulate (Agarwal et al. 1978; Pigott and Ishmael 1979; Xu et al. 2004), nylon fibres or particulates (Porter et al. 1999), PS particulate (Brown et al. 2001; Fournier et al. 2018), or PU particulate (Stemmer et al. 1975). Most of the studies incorporated only one exposure level, and contained a single intratracheal instillation, except for one group in Fournier et al. 2018 (every second day). The rats in the various studies were followed from 1 day to 24 months post-instillation. In general, the foreign body reactions observed in inhalation studies were also observed in the intratracheal studies. One study demonstrated that effects from washed PVC particulates were equal to or greater than those from unwashed PVC particulates, suggesting that adverse effects were from the plastic particulate itself rather than from adsorbed additives (Xu et al. 2004). Additional pulmonary effect observations are outlined in Table E-2 in Appendix E. A developmental study also observed an increase in fetal reabsorption sites and evidence of particle translocation from the lungs (placenta, whole pup, fetal liver and heart, and maternal heart and spleen) (Fournier et al. 2018). Although results from the intratracheal studies corroborate effects observed in the inhalation studies, they should not be primarily relied upon for quantitative dose-response assessments because the route of exposure does not accurately represent deposition patterns and dosing that would be observed from inhalation.

A review of the toxicology of *p*-aramid (an aromatic PA commonly known as Kevlar) fibrils was also identified (Donaldson 2009). Studies of rat lungs identified effects at high exposure levels, such as inflammation, increased cell proliferation, fibrosis, and development of cystic keratinizing squamous cell carcinoma (a tumour stated to be of questionable relevance to humans due to an absence of a human homologue).

7.2.3 Effects of biofilms

Microplastics provide a unique and extensive surface for microorganisms to attach to and colonize in water environments, forming biofilms (Zettler et al. 2013; De Tender et al. 2015; McCormick et al. 2016; Oberbeckmann et al. 2018; Kettner 2018; Arias-Andres et al. 2018, 2019). However, very few studies have analyzed microplastic-associated biofilms.

Biofilms consist of accumulations of microorganisms, typically encased in a self-secreted matrix of extracellular polymeric substances, containing both organic and inorganic matter (Liu et al. 2016; Prest et al. 2016; WRF 2017). The structure of the extracellular polymeric substances provides protection from stressors (e.g., predators, disinfectants), and aids in uptake and utilization of nutrients (Flemming and Wingender 2010; Prest et al. 2016). Biofilms are ubiquitous in the environment (Hall-Stoodley et al. 2004; Yadav 2017) and in drinking water distribution systems (Liu et al. 2016; Prest et al. 2016; WRF 2017), where they provide a habitat for the survival and growth of microorganisms, including potential pathogens (US EPA 2002; Batté et al. 2003; Berry et al. 2006; Liu et al. 2016).

The higher surface-to-volume ratio of microplastics facilitates the absorption of organic matter, which serves as nutrients for microorganisms, thereby promoting biofilm formation. The transport of microplastics over long distances and through the water column (Peng et al. 2017) affords opportunities for attachment of microbial "hitchhikers" and biofilm formation (Kirstein et al. 2016; Zalasiewicz et al. 2016; Keswani et al. 2016). These plastic-associated biofilm communities are sometimes referred to as "plastispheres" (Zettler et al. 2013) and tend to differ from microorganisms in surrounding water or on natural aggregates/particles (Zettler et al. 2013; Hoellein et al. 2014; McCormick et al. 2016; Oberbeckmann et al. 2016; Kettner et al. 2017; Arias-Andres et al. 2018, 2019). Gene sequencing studies have demonstrated that microbial communities on microplastics are less diverse than those on non-plastic substrates (Zettler et al. 2013; Harrison et al. 2014; McCormick et al. 2014, 2016; Ogonowski et al. 2018a), suggesting that microplastics may select for specific microbial colonizers. In other words, the physicochemical properties of microplastics influence the composition and structure of the associated biofilm community (Bhardwaj et al. 2013; Zettler et al. 2013; Harrison et al. 2014; McCormick et al. 2014, 2016). It is unclear what impact this has, but some have hypothesized that it may result in reduced competition and predation, leading to the emergence of potential pathogens (Amalfitano et al. 2014; Keswani et al. 2016; Andrady 2017). Other factors, including environmental conditions (e.g., salinity, temperature), can also influence biofilm formation on microplastics (Harrison et al. 2018; Oberbeckmann et al. 2018; WHO 2019). In addition, microorganism features, such as the hydrophobicity of their cell walls and cell surface charge, can impact attachment to microplastics (Rummel et al. 2017).

Biofilm constituents commonly found on microplastics include various non-pathogenic microorganisms, comprising species of *Pseudomonas*, *Arcobacter*, *Erythrobacter*, *Streptococcus*, *Staphylococcus*, *Aspergillus*, *Penicillium* and *Phanerochaete* (Bhardwaj et al. 2013; McCormick et al. 2014). Pathogenic bacterial sequences, primarily those of *Vibrio*, have been detected in microplastic-associated biofilms (Zettler et al. 2013; De Tender et al. 2015; Kirstein et al. 2016). However, aside from one study (Kirstein et al. 2016), species identification was not possible, and it is therefore unknown whether the organisms were of human health concern. In the study, *Vibrio* spp. of potential human health significance were identified, namely *V. parahaemolyticus*, *V. fluvialis*, and *V. alginolyticus*.

The increased cell density and proximity, improved nutrient availability, and protection afforded by an extracellular polymeric substances matrix make biofilms an ideal environment for interactions between microorganisms, including those on microplastics. Among these interactions is conjugation, the transfer of genetic material through direct cell-to-cell contact (Cook et al. 2011; Stalder and Top 2016). Conjugation is a method of horizontal gene transfer (HGT), the primary mechanism for the spread of

antibiotic resistance, whereby a mobile genetic element (MGE), such as a plasmid, containing antibiotic resistance genes (ARGs), is transferred from a donor to a recipient cell (Von Wintersdorff et al. 2016). A few studies have shown that ARGs are more frequently transferred between microplastic-associated biofilm members than free-living bacteria or biofilms associated with natural aggregates (Arias-Andres et al. 2018; Eckert et al. 2018a,b; Imran et al. 2019; Laganà et al. 2019). Transfer also occurred between a broader (i.e., more distantly related) group of microorganisms on the microplastics than in the natural environment. These findings suggest that microplastic-associated biofilms provide a favourable environment (i.e., "hot spot") for HGT events and may select for antibiotic resistant microorganisms and ARGs, which may then be transported to different habitats. Transfer of ARGs via microplastics has been observed between wastewaters and the aquatic environment (Eckert et al. 2018a,b). Transfer events on microplastics may be further amplified through exposure to metals, as metal resistance genes are present on the same plasmid as antibiotic resistance genes (Baker-Austin et al. 2006; Wright et al. 2006; Seiler and Berendonk 2012; Zhang et al. 2018; Imran et al. 2019).

Although research in this area is very limited, studies suggest that plastic-associated biofilms in water may harbour potential human pathogens and ARGs. Given that microplastics can travel long distances (see Section 4, WHO 2019), there is a possibility that these organisms and/or ARGs may be dispersed across waters and enter drinking water sources. Despite this, there is no indication of how prevalent these organisms are or of how long they persist and/or remain infectious while in a plastisphere. Moreover, conventional drinking water treatment is expected to significantly reduce microplastics and inactivate associated biofilm organisms (see Section 4.1.3, WHO 2019). Thus, there is currently no evidence to suggest that microplastic-associated biofilms in drinking water pose a risk to human health.

Microorganisms might also adhere to the surface of airborne microplastics, but data are limited. Microorganisms have been measured in airborne particulates (Noble et al. 1963; Brodie et al. 2007), although no data exist specifically for plastic particulates. Adherence and growth of microorganisms on airborne microplastics might be limited because they could be dependent on the contact of microorganisms and microplastics in the environment. However, if contact does occur, the plastic particulates might protect and shield adhered microorganisms (Prata 2018). While no data could be found on the characterization of microbial communities potentially colonizing airborne microplastics, lung infections could theoretically occur if pathogenic species were adhered to microplastics and inhaled (Prata 2018).

8. Transport of chemicals

In addition to the physical hazards presented by plastic particles themselves, it is possible that effects could occur as a result of exposure to residual monomers, chemical additives, and sorbed environmental contaminants (e.g., persistent organic pollutants [POPs] and metals) that may leach from microplastic particles (Munier and Bendell 2018; SAPEA 2019). Although there is potential for environmental or human exposure to these compounds, these chemicals are considered to be under the purview of various programs at ECCC and Health Canada and will continue to be managed in accordance with those programs.

Any effects observed from the transport of chemicals are highly context dependent. For example, the type of plastic and the physicochemical properties of the sorbed chemical are known to have an effect on sorption ability. In general, PE shows a greater ability to sorb contaminants, while PET and PVC have a lower sorption capacity (Alimi et al. 2018). Plastics with high surface-area-to-volume ratios (i.e., small, elongated, or have an irregular shape) tend to have higher sorption capacities (Rochman 2015). For instance, PVC was shown to have significantly greater absorption of copper than PS, which could be due to its greater surface area and polarity (Brennecke et al. 2016; Munier and Bendell 2018). Sorption can also be affected by factors such as age, shape, molecular weight and porosity of the particle, temperature, salinity and pH of the environment (increased salinity and particle age tend to increase sorption, and alkaline environments favour sorption of cations), and the concentration of metals and other contaminants in the surrounding waters (Rochman 2015; Alimi et al. 2018; Munier and Bendell 2018; Guo and Wang 2019a). Di and Wang (2018) sampled surface waters and sediments from China's Three Gorges Reservoir and found that several contaminants were adsorbed to the surface of the recovered microplastics, including organic solvents and pharmaceutical intermediates.

The properties of the receiving environment can also affect contaminant transfer. Mohamed Nor and Koelmans (2019) found that the transfer of polychlorinated biphenyls (PCBs) from microplastics in simulated gut fluid is biphasic and fully reversible. More specifically, the effect of microplastics in the gut depends on the contents of the gut system. Ingested plastics acted as a source of hydrophobic organic compounds (HOCs) in clean gut systems, whereas in contaminated gut systems, clean microplastics rapidly extracted PCBs from food or other organic matter (Mohamed Nor and Koelmans 2019). The authors concluded that chemical contamination and cleaning can occur simultaneously when microplastics are ingested.

Although many of the compounds associated with plastic have short biological half-lives and are not persistent, plastic particles within the body could present a long-term source of exposure to the chemicals (Engler 2012). While recent reviews indicate that there is a low health concern for human exposure to chemicals from ingestion of microplastics from food or drinking water (EFSA 2016; FAO 2017; WHO 2019), further research would be required before a human health risk assessment on microplastics is possible. No data could be found on the transfer of these compounds in the human respiratory or GI tract.

Sorbed chemicals

Provencher et al. (2018c) found no significant correlations between concentrations of various PCB congeners in Northern fulmars (*Fulmarus glacialis*) and the amount of ingested plastics when using a toxic equivalency factor (TEF) approach. They found that plastics did not contribute to the PCB concentrations in the birds and that the PCB congener profile between ingested plastics and the liver differed (Provencher et al. 2018c). This could be the result of the ability of Northern fulmars to metabolize or bio-transform contaminants such as PCBs (Letcher et al. 2010; Provencher et al. 2018c). In a study using goldfish (*Carassius auratus*), Grigorakis and Drouillard (2018) observed lower dietary assimilation efficiencies (13.4%) for PCBs sorbed to microplastics compared to efficiencies (51.6%) for PCBs associated with food. The authors concluded that the lower bioavailability of PCBs associated with

microplastics indicates that microplastic presence is unlikely to increase PCB bioaccumulation in fish. In a study by Devriese et al. (2017), Norway lobsters (*Nephrops norvegicus*) exposed to PCB-loaded PE or PS microplastics showed no significant bioaccumulation of the chemicals, with uptake of the PCBs being limited. Furthermore, Gerdes et al. (2019) found a positive correlation between the elimination rate of PCBs in *Daphnia magna* and the presence of microplastics. More specifically, the presence of microplastics together with PCBs was able to increase the elimination rate of high-molecular-weight PCB congeners in *D. magna* fourfold.

Diepens and Koelmans (2018) introduced a theoretical model simulating the transfer of microplastics and HOCs in aquatic Arctic food webs. Simulated scenarios showed that PCBs biomagnify to a lesser extent with higher levels of microplastic ingestion, which supports the evidence previously described. Conversely, the same model also indicated that polycyclic aromatic hydrocarbons (PAHs) biomagnify more with elevated levels of microplastic ingestion. Under different conditions, Magara et al. (2018) found that the uptake and accumulation of fluoranthene (a PAH) in blue mussels *(Mytilus edulis*) were not affected by incubation with microplastics and that incubation with microplastics reduced the bioavailability of fluoranthene. In a study modelling the transfer of POPs from PVC and PE to benthic invertebrates, fish, and seabirds, Bakir et al. (2016) found that food and water were the main pathways of exposure for all organisms, and input from microplastic particles was negligible.

Tanaka et al. (2013) studied the occurrence of polybrominated diphenyl ethers (PBDEs) in tissues of short-tailed shearwater (*Puffinus tenuirostris*) seabirds, in their natural prey, and in plastics in the stomachs of the seabirds. In three of the 12 short-tailed shearwaters examined, they detected higher-brominated congeners of PBDEs that were not present in their prey (i.e., lanternfish and squid), which were also sampled from the same area as the seabirds. However, they did detect these PBDEs on the plastics found in the stomachs of the three birds, which suggests that plastic-derived chemicals were transferred from the ingested plastic to the seabird tissue.

Hydrophobic POPs of potential human health concern (such as PCBs, PAHs, and organochlorine pesticides) can readily sorb to plastics. For that reason, plastic compounds such as PE and PU are used as passive samplers in environmental monitoring (WHO 2019).

Studies on microplastic-associated sorbed pollutants in drinking water could not be identified, but increased POPs in microplastics have been measured in marine environments and shorelines near urban environments (Wang et al. 2017; Pellet Watch 2019).

Limited data exist on the sorption of chemicals to microplastic particulates in outdoor air, indoor air, or indoor dust. Adsorption of organic pollutants in air to plastic particulates could theoretically occur, but would be dependent on the duration of microplastic suspension in air (Prata 2018). One study reported that no significant adsorption of PCBs, dichlorodiphenyldichloroethylene (DDE) or nonylphenol occurred on virgin PP pellets released to the atmosphere for six days (Mato et al. 2001). Therefore, the contribution of microplastics to inhalation of sorbed chemicals is unknown but potentially limited, although it is anticipated to be dependent on environment (e.g., urban versus rural environments, proximity to point sources). Overall, current research shows that, while microplastics are able to

transport POPs, the evidence suggests that the impact of this exposure pathway is minimal (Burns and Boxall 2018).

Monomers

Plastics are manufactured through the polymerization of monomers, which vary in toxicity. Some of the more hazardous compounds include acrylonitrile, acrylamide, 1,3-butadiene, ethylene oxide, and vinyl chloride (Lithner et al. 2011). Depending on the polymerization process, the plastic material can contain a range of concentrations of residual monomers (from negligible amounts to up to 4%) due to incomplete polymerization (Araújo et al. 2002; Lithner et al. 2011). Plastics can also be degraded (through biological processes and weathering) into monomers and oligomers, but few data exist on the contribution of these processes to human exposures to monomers (WHO 2019).

Additives

As discussed in Section 2, plastic additives can include polymer stabilizers, flame retardants, lubricants, plasticizers, and colourants. Compounds with potential human health effects that are additives of plastics include phthalates, PBDEs, lead, and cadmium (WHO 2019), among others. Plastic additives are mostly not co-polymerized, resulting in increased likelihood of being leached into the environment (Wright and Kelly 2017; Hahladakis et al. 2018). Molecular weight of additives and age of plastics are factors that can influence the rate of migration of additives from plastics to the surrounding environment (Hansen et al. 2013; Suhrhoff and Scholz-Böttcher 2016; Jahnke et al. 2017). Limited data exist on the contribution of microplastics to concentrations of plastic additive compounds in the environment, but there is evidence of potential migration pathways for the compounds in sources of human relevance, such as food (Helmroth et al. 2002; Muncke 2011), water (WHO 2019) and indoor dust (Rauert et al. 2014).

9. Knowledge gaps and considerations for future research

Several knowledge gaps were identified during the writing of this report and are outlined below with the objective of encouraging further research. Addressing these knowledge gaps will contribute to the understanding of the environmental and human health risks of plastic pollution and will inform science-based policy and regulatory decisions related to plastic pollution.

9.1 Occurrence

While the approach to observing macroplastics is relatively obvious given their size, there is a general lack of consistency and reliability in the methods used to sample and quantify microplastics in the environment and other media (e.g., drinking water and food). Many studies rely only on visual identification to determine if a particle is plastic. This can lead to a high false positive rate (especially at sizes smaller than 1 mm) and does not allow for proper characterization of plastics. For instance, when fibres visually identified as microplastics from the GI tracts of eelpout (*Zoarces viviparus*) were analyzed with µATR-FTIR by Wesch et al. (2016), none of the fibres were determined to be of synthetic origin.

Given these findings, the authors question whether visual identification alone is sufficient to determine if microfibres are microplastics and call for standardized approaches for identifying and monitoring microplastics. Non-specific fluorescence staining methods have been suggested as a potential rapid-screening approach for detecting and quantifying microplastics in various media (Erni-Cassola et al. 2017; Maes et al. 2017; Prata et al. 2019). However, a major drawback of these staining methods is the possible introduction of false positives through the staining of biological organisms, such as marine algae or organic matter.

Spectroscopic techniques, such as FTIR, Raman spectroscopy and pyrolysis GCMS, are currently the preferred methods for plastic characterization and are often used following separation of suspected plastic particles from sample media and visual identification using a microscope. Although they increase the accuracy of the identification of microplastics, spectroscopic analyses have limitations that can lead to the underestimation of microplastics in samples. With Raman spectroscopy, the generation of fluorescence can overpower the Raman spectrum produced, which can hinder the identification of potential plastics (Rezania et al. 2018). Furthermore, the signal can be heavily influenced by dyes, as well as by microbiological, organic and inorganic substances (Nguyen et al. 2019). With infrared (IR) spectroscopy, black or dark particles are not detected because they have a high absorption rate (Rezania et al. 2018), and particles below 20 µm may not yield enough absorbance interpretable spectra (Li et al. 2018b). Pyrolysis GCMS lacks reproducibility, as results are highly dependent on sample preparation and pyrolysis type. Thermal desorption GCMS is best used for samples of high mass (up to 100 mg) but lacks the sensitivity of pyrolysis GCMS (Nguyen et al. 2019). Microplastic counts can also be overestimated. Using SEM with energy dispersive X-ray spectroscopy (EDS), Anderson et al. (2017) found that, on average, 23% of the particles that were visually identified as plastics were not plastic. Burns and Boxall (2018) highlight that the error rate for visually identifying particles as plastic ranges from 33% to 70%. While analytical methods may help to confirm the synthetic nature of microplastics sampled in the environment, the inconsistencies in sampling methods (e.g., size of subsamples and sampling strategies) can limit the comparability of such analyses.

Studies investigating the occurrence of plastics in the environment and other media often use different units to report plastic abundance (e.g., plastics per area vs. plastics per unit volume), thereby limiting comparisons between studies and the generalizability of results. Standardized reporting metrics are required to ensure reporting consistency and study comparability (Burns and Boxall 2018). Another major gap in the analytical process is that there are no inter-laboratory studies, which are useful for method validation. Furthermore, due to variability and difficulty in quantifying microplastics, large standard deviations have been reported for the occurrence of microplastics in the environment and, in some instances, the standard deviation value exceeds the reported measurement.

In water, microplastics are sampled at size ranges that are compatible with available sampling apparatus (e.g., trawl nets, which have a mesh size of 300 to 350 µm). This means that microplastics smaller than 300 µm can often go undetected. This is an issue for microfibres in particular given their narrow size (Covernton et al. 2019). Sampling methods therefore need to be developed to support the characterization of the smaller size fractions of plastics in the environment. Further, sampling depths vary across studies and are not standardized (i.e., trawl nets would be biased to less dense plastics that

are present near the surface of surface waters, and studies conducted at a greater depth would be biased against denser plastics).

A limited number of published studies report on the environmental monitoring and effects of microplastics in freshwater and terrestrial environments (Burns and Boxall 2018; Provencher et al. 2018b). There is a need to expand work to include monitoring studies to other ecosystems, particularly terrestrial ecosystems.

In terrestrial matrices, studies of microplastic occurrence are scarce, possibly due to difficulty in translating research ideas in a marine context to a terrestrial context (Rillig 2012; da Costa et al. 2019). For example, there are no parallels for the accumulation of microplastics along shorelines in a terrestrial setting. In addition, it is often more difficult to isolate and characterize microplastics from a soil matrix; soil can contain varying levels of organic matter, which can distort signals and present problems when using FTIR and Raman spectroscopy for plastic characterization (Bläsing and Amelung 2018). Furthermore, there is a lack of standardized protocols for soil sampling and analysis in various soil types (da Costa et al. 2019). It has been suggested that a standard step-by-step approach be employed for terrestrial samples, involving removal of adherent fragments, mineral phase, and organic matter, followed by microplastic identification and quantification (da Costa et al. 2019).

There is also a lack of appropriate quantitative data for microplastic presence in drinking water and in water discharged after wastewater treatment, and limited information is available on the fate of microplastics during the wastewater treatment process, including particle breakdown, particle composition, removal efficiency, and subsequent release of these microplastics to other environmental compartments.

Occurrence data for microplastics in food is also scarce, with little to no Canadian-specific data. Data that do exist are focused on wild marine fish and shellfish, with limited occurrence data for freshwater and farmed species or other foods. In addition, occurrence data are needed for the tissues and organs of animals that are consumed by humans. Data are lacking on the potential effects of cooking or food processing (e.g., fresh versus frozen food) on microplastic concentrations, the impact of the food matrix on microplastic bioavailability (e.g., water-based versus solid/dry foods), and the potential point source(s) of exposure to microplastics in food. Further studies are needed to determine whether food manufacturing, processing and/or handling as well as food packaging materials may contribute to microplastic concentrations in food.

There are currently no validated or recognized methods for the collection or analysis of microplastic samples in air, and little information is available on the partitioning of microplastic particles between air and dust. In order to accurately assess microplastic exposure from air, there is a need to develop and validate accurate, precise and reproducible methods for the sampling, extraction, characterization, and quantification of airborne microplastics and microplastics in settled dust and air, including robust quality assurance and control protocols. As Canadians spend approximately 90% of their time indoors, data on both indoor and outdoor microplastic exposures are needed to determine personal exposures, to understand their sources, pathways, fate, and distribution, and to identify and prioritize specific

microplastic categories or mixtures for future research. There is also a need to explore the relationship between airborne microplastic particles and particulate matter. For example, knowing what proportion of particulate matter is composed of plastic polymers and knowing whether airborne plastic particles behave similarly to other airborne particulates would be useful in determining whether inferences can be made from the wealth of knowledge that exists on particulate matter.

Several researchers have identified the need for standardized protocols and stricter quality assurance in literature to ensure the availability of more high quality occurrence and exposure data in all media (Burns and Boxall 2018; Hermsen et al. 2018; Gouin et al. 2019; Koelmans et al. 2019). For sampling methods, this would include collection media, equipment, and handling procedures, as well as laboratory analysis practices. Due to the ubiquity of plastics, additional care must be taken throughout the entire process, from sample collection to laboratory analysis, to prevent sample contamination.

The importance of protocol development can be demonstrated by Provencher et al. (2017, 2019), who developed the only standardized protocols for monitoring and studying ingested plastics in seabirds. They include standardized field and lab techniques, as well as reporting guidelines for data (Provencher et al. 2017, 2019). The use of these standardized techniques by the international seabird community has led to spatial and temporal tracking of trends in plastics in the marine environment.

It has been recommended that standardized quality criteria be developed that can be used to evaluate the appropriateness of studies on microplastic occurrence and effects. Hermsen et al. (2018) proposed several areas that should be evaluated when scoring the quality of microplastic ingestion studies: sampling method and strategy, sample size, sample processing and storage, laboratory preparation, clean air conditions, negative and positive controls, target component, sample treatment, and polymer identification. When reviewing current studies on microplastic ingestion by biota, they identified negative controls, polymer identification, laboratory preparation, and sample treatment as areas that were particularly lacking in quality and available information. Koelmans et al. (2019) evaluated 50 microplastic studies in freshwater surface water and drinking water using the same method identified by Hermsen et al. (2018). Only four studies scored positively on all proposed quality criteria; 92% of the reviewed studies were not considered complete or reliable on at least one criterion. It should be noted that Hermsen et al. (2018) and Koelmans et al. (2019) acknowledge that their criteria are not an absolute judgement of the value of studies since not all aspects of studies could be captured in their scoring system. Moving forward, the use of standardized quality criteria will ensure that only data of acceptable quality are being used to inform scientists and policy makers and that the data are both reproducible and directly comparable.

There is also a paucity of data on the common or important sources of microplastics in the environment and other media, such that identifying source contributions of microplastics is difficult. There is a need to develop libraries that can be used to link samples to their sources using their chemical composition (polymer and additive chemicals) and other physical properties. Furthermore, establishing a taxonomy of microplastics based on morphology may also be informative in determining sources (Helm 2017).

Lastly, data on the occurrence and effects of nanoplastics are still emerging and poorly understood. It is unclear whether and how nanoplastics may form in the environment (e.g., whether they are formed by processes such as the weathering of macroplastics or microplastics). There is a lack of appropriate analytical methodologies for nanoscale materials in all media, making accurate measurements of environmental occurrence and behaviour of nanoplastics difficult to evaluate (SAPEA 2019). As nanoplastics are inherently more difficult to test and measure, the importance of plastics fragmenting to the nanoscale remains unclear at this time (Koelmans et al. 2015).

9.2 Environmental effects

The size ranges and concentrations of microplastics used in ecotoxicological research do not reflect the concentrations or sizes of microplastics collected in the environment using current sampling techniques. Microplastic effects studies are often performed using concentrations that are much higher than those currently reported in the environment or very small microplastics for which limited occurrence data exists (SAPEA 2019). Researchers studying effects should use plastics of similar size, shape, and composition to those found in the environment. Additionally, there is a need to further investigate the relationship between microplastics and natural particles that exist in the environment that induce similar effects in biota. Currently, experimental designs do not differentiate plastic-specific effects from those caused by other particles, such as clay or cellulose (Ogonowski et al. 2018b). Furthermore, effects studies are largely conducted with PS microplastic spheres, which are not representative of plastics found in the environment. More frequently detected microplastics (i.e., PP, polyester, and PA, among others) are underrepresented in effect studies (SAPEA 2019).

There is therefore a need to develop standard methods for testing the potential for adverse effects associated with exposure to plastic. For example, there is a need to evaluate the relationship between the properties of plastic (e.g., particle size, polymer type, shape and particle number) and toxicity. There is a corresponding lack of consistency in reporting test concentrations in studies; some studies report weight/volume, while others report particle number/volume.

Often, microplastics used in toxicity testing are purchased. These microplastics would not be ideal models for microplastics that would be encountered in the environment as they can contain additives such as surfactants. For example, Pikuda et al. (2019) found that the acute toxicity to *D. magna* was associated with sodium azide, a surfactant, and not the plastic PS particles themselves. When the sodium azide was removed from the plastics, the PS particles no longer caused mortality. Thus, plastic particles used in toxicological studies should be washed to remove any additives that may cause effects that can be confused with effects caused by the particles themselves. Currently, this is not standard practice, and was not considered when selecting studies for this report.

Burns and Boxall (2018) suggested that research in this field should move to the use of environmental degradation studies. Follow-up ecotoxicological studies should then be conducted using the resulting materials identified in the degradation studies. There is a need to develop certified standard reference materials that are environmentally relevant and meet the needs for risk assessment. This would help to characterize the effects of environmentally relevant plastics. Experiments that consider chronic effects

(including effects of long-term retention within organisms) using consistent endpoints should also be completed. Provencher et al. (2018b) highlighted a need for studies that examine plastic transfer between predator and prey, as well as the biomagnification, bioaccumulation, and bioconcentration of these transferred plastics. Further research is needed on the mechanisms of absorption, distribution, metabolism, and excretion of microplastics and on the feasibility of a read-across approach from particle translocation studies. There is also a need to develop a better understanding of the sublethal, interactive and cumulative effects of plastics with other factors. For example, although a recent study has shown that there may be sublethal effects related to plastic ingestion on the blood chemistry of flesh-footed shearwaters in the southern hemisphere, the authors are unable to make definitive links at this time (Lavers et al. 2019). Further, while studies such as those by Lavers and Bond (2016) on ingested plastics as a route for the transport of trace metals have indicated that concentrations of certain metals were positively related to plastic mass, generalizations about the transfer of trace elements from ingested plastics are not yet possible as the mechanisms underlying this process are unknown. In addition, some studies on microplastic ingestion have only examined a portion of an organism's digestive tract, which may lead to an underestimation of ingestion rates, since other components of the GI tract may also contain microplastic particles. To accurately estimate all ingested microplastic, it is recommended that the entire GI tract, from esophagus to vent, of fish and the entire body for smaller species (e.g., bivalves) be examined (Hermsen et al. 2018).

There is a lack of studies on microplastics in soil, and further research is needed to fully understand the interactive effects that plastic pollution will have on soil fauna and potential uptake into food crops. Finally, while some sources and occurrences of microfibres have been identified, further work is needed to fully understand their distribution and fate in the environment, as well as the effects this type of plastic pollutant presents.

Recent research has begun to explore links between plastic pollution and climate change. For example, Royer et al. (2018) showed that commonly used plastics produce greenhouse gases when exposed to ambient solar radiation, and virgin plastics had higher emissions of hydrocarbon gases than environmentally aged plastic pellets. This suggests that plastic pollution may be contributing to climate change. There is also evidence to suggest that climate change could contribute to increased wildlife exposure to plastic pollution. For example, Drever et al. (2018) reported that, under conditions of unusually warm ocean temperatures, red phalaropes (*Phalaropus fulicarius*) were found feeding closer to shore. The authors indicated that distribution shifts of the birds closer to shore resulted in increased exposure to plastic pollution.

In addition to the uncertainties inherent in microplastic toxicity testing described above, there is a need to conduct toxicity tests on nanoplastics; however, these may also be confounded by the suspension matrix used (Pikuda et al. 2019). Toxicity results for studies using commercially formulated nanoplastics, which are likely to contain preservatives, antimicrobials, or surfactants, must therefore be carefully considered (Pikuda et al. 2019).

9.3 Human health effects

In order to better understand the potential human health effects of microplastics resulting from both oral and inhalation exposures, an improved understanding of the extent and nature of human exposure and potential toxicological hazards is required.

With respect to the potential human health impacts of microplastic ingestion (e.g., from drinking water and/or food) and inhalation (e.g., from indoor and ambient air), more research is needed on the uptake and fate of microplastics in the GI and respiratory tracts and on the bioavailability of chemical substances associated with microplastics. In addition, from an inhalation perspective, there is a need to better characterize microplastics exposure for particles of aerodynamic diameter in the micron scale (<1 mm), with a focus on inhalable particles (<10 µm) and especially respirable particles (<2.5 µm) that can penetrate deep into the lungs. There is also a need to understand the physical characteristics of microplastics (e.g., length, diameter, polymer type and surface chemistry) that may determine their bioavailability, tissue distribution, and potential relevance to human health.

Toxicological research using appropriate cell models and experimental animals is needed to better inform human health risk assessment, including identifying target tissues, threshold doses, and mode of action. Epidemiological studies in the general population would also help to inform the human health impacts of microplastics. More research is also required to improve the understanding of whether the characteristics (e.g., size, shape, composition) of microplastics influence their potential adverse effects. In addition, as information on the health-relevant properties of microplastics emerges, standardized reporting metrics are needed to ensure that those features are adequately characterized in scientific reports.

There is also a need to understand the extent to which microplastics may act as a vector for transporting other chemicals (e.g., chemicals additives, adsorbed environmental contaminants) and to determine whether they have an impact on human health. While recent reviews indicate that there is a low health concern for human exposure to chemicals from ingestion of microplastics from food or drinking water (EFSA 2016; FAO 2017; WHO 2019), further research would be required before a human health risk assessment on microplastics is possible. Further research investigating the toxicity of nanoplastics is also required, as described above.

Lastly, there is also a need for improved characterization of microplastic-associated biofilms in drinking water, drinking water sources and air. Gaining increased knowledge in areas such as the factors shaping biofilm composition, the taxonomy of biofilm communities, and biofilm activity and interactions (e.g., transfer of ARGs) would contribute to the understanding of the importance of biofilms on human health.

10. Findings

Plastic pollution, in the form of macroplastics and microplastics, is ubiquitous in the environment. It is estimated that in 2016, 1% of all plastic waste in Canada, or 29 kt, was discharged to the environment as

pollution. Since plastics degrade very slowly and are persistent in the environment, the frequency of occurrence of plastic pollution in the environment is expected to increase.

Macroplastics have been demonstrated to cause physical harm to environmental receptors on an individual level and to have the potential to adversely affect habitat integrity. Organisms have been shown to ingest macroplastics and to become entangled in macroplastics, which can result in direct harm and in many cases, mortality.

The evidence for potential effects of microplastic pollution on environmental receptors is less clear and sometimes contradictory, and further research is required. For example, although there are reports indicating that exposure of environmental receptors to microplastics can lead to mortality, developmental and reproductive effects, effects on feeding and energy production, and biochemical or molecular-level effects, a similar number of reports have found no effects.

The current literature on the human health effects of microplastics is limited, although a concern for human health has not been identified at this time. Potential exposure pathways include air, water and food. While some occupational epidemiology and experimental animal studies show the potential for effects at high exposure concentrations, they are of questionable reliability and relevance, and further research on the potential for microplastics to impact human health is required.

In order to advance the understanding of the impacts of plastic pollution on the environment and human health, it is recommended that research be conducted to address key knowledge gaps identified in this report. This includes studies to improve the understanding of both exposure to and potential toxicity of plastics. More specifically, research is recommended in the following areas:

- Developing standardized methods for sampling, quantifying, characterizing and evaluating the effects of macroplastics and microplastics;
- Furthering understanding of human exposure to microplastics;
- Furthering understanding of the ecotoxicological effects of microplastics;
- Furthering understanding of the effects of microplastics on human health; and
- Expanding and developing consistent monitoring efforts to include lesser characterized environmental compartments such as soil.

Given the increasing amounts of plastic pollution in the environment and the demonstrated ability of macroplastics to harm biota, it is anticipated that the frequency of occurrence of physical effects on individual environmental receptors will continue to increase if current trends continue without mitigation measures.

In accordance with the precautionary principle, action is needed to reduce macroplastics and microplastics that end up in the environment.

11. References

Abbasi S, Soltani N, Keshavarzi B, Moore F, Turner A, Hassanaghaei M. 2018. Microplastics in different tissues of fish and prawn from the Musa Estuary, Persian Gulf. Chemosphere. 205:80-87.

Adam V, Yang T, Nowack B. 2019. Toward an ecotoxicological risk assessment of microplastics: Comparison of available hazard and exposure data in freshwaters. Environ Toxicol Chem. 38(2):436-447.

AET Group (Kitchener, ON). 2016. 2016 Toronto Litter Audit. Summary Report. Toronto (ON): City of Toronto. 90 p.

AET Group (Kitchener, ON). 2019. 2019 City of Edmonton Litter Audit Report. Final Report. Edmonton (AB): City of Edmonton. 72 p.

Agarwal DK, Kaw JL, Srivastava SP, Seth PK. 1978. Some biochemical and histopathological changes induced by polyvinyl chloride dust in rat lung. Environ Res. 16(1-3):333-341.

Akhbarizadeh R, Moore F, Keshavarzi B. 2018. Investigating a probable relationship between microplastics and potentially toxic elements in fish muscles from northeast of Persian Gulf. Environ Pollut. 232:154-163.

Akhbarizadeh R, Moore F, Keshavarzi B. 2019. Investigating microplastics bioaccumulation and biomagnification in seafood from the Persian Gulf: a threat to human health? Food Addit Contam Part A Chem Anal Control Expo Risk Assess. 36(11):1696-1708.

Alimi OS, Farner Budarz J, Hernandez LM, Tufenkji N. 2018. Microplastics and nanoplastics in aquatic environments: Aggregation, deposition, and enhanced contaminant transport. Environ Sci Technol. 52(4):1704-1724.

Aljaibachi R, Callaghan A. 2018. Impact of polystyrene microplastics on *Daphnia magna* mortality and reproduction in relation to food availability. PeerJ. 6:e4601.

Allen S, Allen D, Phoenix VR, Le Roux G, Durántez Jiménez P, Simonneau A, Binet S, Galop D. 2019. Atmospheric transport and deposition of microplastics in a remote mountain catchment. Nat Geosci. 12:339-344.

Amalfitano S, Coci M, Corno G, Luna, GM. 2014. A microbial perspective on biological invasions in aquatic ecosystems. Hydrobiologia. 746(1):13-22.

[AMAP] Arctic Monitoring and Assessment Programme. 2004. AMAP Assessment 2002: Persistent Organic Pollutants in the Arctic. Oslo (NO): AMAP. 310 p.

Anderson JC, Park BJ, Palace VP. 2016. Microplastics in aquatic environments: implications for Canadian ecosystems. Environ Pollut. 218:269-280.

Anderson PJ, Warrack S, Langen V, Challis JK, Hanson ML, Rennie MD. 2017. Microplastic contamination in Lake Winnipeg, Canada. Environ Pollut. 225:223-231.

Andrady AL. 2015. Persistence of plastic litter in the oceans. In: Bergmann M, Gutow L, Klages M, editors. Marine anthropogenic litter. Cham (CH): Springer. Chapter 3. p. 57-72.

Andrady AL. 2017. The plastic in microplastics: A review. Mar Pollut Bull. 119(1):12-22.

Arias-Andres M, Kettner MT, Miki T, Grossart HP. 2018. Microplastics: New substrates for heterotrophic activity contribute to altering organic matter cycles in aquatic ecosystems. Sci Total Environ. 635:1152-1159.

Arias-Andres M, Rojas-Jimenez K, Grossart H. 2019. Collateral effects of microplastic pollution on aquatic microorganisms: An ecological perspective. Trends Analyt Chem. 112:234-240.

Araújo PHH, Sayer C, Giudici R, Poço JGR. 2002. Technique for reducing residual monomer content in polymers: a review. Polym Eng Sci. 42(7):1442-1468. [cited in WHO 2019].

Ašmonaitė G, Sundh H, Asker N, Carney Almroth B. 2018. Rainbow trout maintain intestinal transport and barrier functions following exposure to polystyrene microplastics. Environ Sci Technol. 52(24):14392-14401.

Au SY, Bruce TF, Bridges WC, Klaine SJ. 2015. Responses of *Hyalella azteca* to acute and chronic microplastic exposures. Environ Toxicol Chem. 34(11):2564-2572.

Avio CG, Gorbi S, Milan M, Benedetti M, Fattorini D, d'Errico G, Pauletto M, Bargelloni L, Regoli F. 2015. Pollutants bioavailability and toxicological risk from microplastics to marine mussels. Environ Pollut. 198:211-222.

Baker-Austin C, Wright MS, Stepanauskas R, McArthur JV. 2006. Co-selection of antibiotic and metal resistance. Trends Microbiol. 14(4):176-182.

Bakir A, O'Connor IA, Rowland SJ, Hendriks AJ, Thompson RC. 2016. Relative importance of microplastics as a pathway for the transfer of hydrophobic organic chemicals to marine life. Environ Pollut. 219:56-65.

Baldwin AK, Corsi SR, Mason SA. 2016. Plastic debris in 29 Great Lakes tributaries: Relations to watershed attributes and hydrology. Environ Sci Technol. 50(19):10377-10385.

Balestri E, Menicagli V, Ligorini V, Fulignati S, Raspolli Galletti AM, Lardicci C. 2019. Phytotoxicity assessment of conventional and biodegradable plastic bags using seed germination test. Ecol Indicators. 102:569-580.

Ball H, Cross R, Grove E, Horton A, Johnson A, Jürgens M, Read D, Svendsen C. 2019. Sink to river - river to tap: A review of potential risks from nanoparticles and microplastics. London (GB): UK Water Industry Research Limited. Report No.:19/EQ/01/18.

Ballent A, Corcoran PL, Madden O, Helm PA, Longstaffe FJ. 2016. Sources and sinks of microplastics in Canadian Lake Ontario nearshore, tributary and beach sediments. Mar Pollut Bull. 110(1):383-395.

Barboza LGA, Dick Vethaak A, Lavorante BRBO, Lundebye AK, Guilhermino L. 2018. Marine microplastic debris: An emerging issue for food security, food safety and human health. Mar Pollut Bull. 133:336-348.

Barboza LGA, Cunha SC, Monteiro C, Fernandes JO, Guilhermino L. 2020. Bisphenol A and its analogs in muscle and liver of fish from the North East Atlantic Ocean in relation to microplastic contamination. Exposure and risk to human consumption. J Hazard Mater. 393:122419.

Batel A, Linti F, Scherer M, Erdinger L, Braunbeck T. 2016. Transfer of benzo[a]pyrene from microplastics to *Artemia nauplii* and further to zebrafish via a trophic food web experiment: CYP1A induction and visual tracking of persistent organic pollutants. Environ Toxicol Chem. 35(7):1656-1666.

Batté M, Appenzeller BMR, Grandjean D, Fass S, Gauthier V, Jorand F, Mathieu L, Boualam M, Saby S, Block JC. 2003. Biofilms in drinking water distribution systems. Rev Environ Sci Biotechnol. 2(2):147-168.

Beer S, Garm A, Huwer B, Dierking J, Nielsen TG. 2018. No increase in marine microplastic concentration over the last three decades—A case study from the Baltic Sea. Sci Total Environ. 621:1272-1279.

Beiras R, Bellas J, Cachot J, Cormier B, Cousin X, Engwall M, Gambardella C, Garaventa F, Keiter S, Le Bihanic F, et al. 2018. Ingestion and contact with polyethylene microplastics does not cause acute toxicity on marine zooplankton. J Hazard Mater. 360:452-460.

Bergmann M, Wirzberger B, Krumpen T, Lorenz C, Primpke S, Tekman MB, Gerdts G. 2017. High quantities of microplastic in Arctic deep-sea sediments from the HAUSGARTEN Observatory. Environ Sci Technol. 51:11000-11010.

Bernardini I, Garibaldi F, Canesi L, Fossi MC, Baini M. 2018. First data on plastic ingestion by blue sharks (*Prionace glauca*) from the Ligurian Sea (North-Western Mediterranean Sea). Mar Pollut Bull. 135:303-310.

Berry D, Xi C, Raskin L. 2006. Microbial ecology of drinking water distribution systems. Curr Opin Biotechnol. 17(3):297-302.

Besseling E, Redondo-Hasselerharm P, Foekema EM, Koelmans AA. 2019. Quantifying ecological risks of aquatic micro- and nanoplastic. Crit Rev Environ Sci Technol. 49:32-80.

Bhardwaj H, Gupta R, Tiwari A. 2013. Communities of microbial enzymes associated with biodegradation of plastics. J Polym Environ. 21(2):575-579.

Biber NFA, Foggo A, Thompson RC. 2019. Characterising the deterioration of different plastics in air and seawater. Mar Pollut Bull. 141:595-602.

Bishop DP. 1995. Physical and chemical effects of domestic laundering processes. In: Carr CM, editor. Chemistry of the Textiles Industry. Glasgow (GB): Springer, Dordrecht. p. 125-172.

Blašković A, Fastelli P, Čižmek H, Guerranti C, Renzi M. 2017. Plastic litter in sediments from the Croatian marine protected area of the natural park of Telaščica bay (Adriatic Sea). Mar Pollut Bull. 114(1):583-586.

Blarer P, Burkhardt-Holm P. 2016. Microplastics affect assimilation efficiency in the freshwater amphipod *Gammarus fossarum*. Environ Sci Pollut Res. 23(23):23522-23532.

Bläsing M, Amelung W. 2018. Plastics in soil: Analytical methods and possible sources. Sci Total Environ. 612:422-435.

Blettler MCM, Ulla MA, Rabuffetti AP, Garello N. 2017. Plastic pollution in freshwater ecosystems: macro-, meso-, and microplastic debris in a floodplain lake. Environ Monit Assess. 189(11):581.

Böhmert L, Stock V, Braeuning A. 2019. Plausibility of microplastic uptake in a paper by Deng et al., Scientific Reports 7: 46687, 2017. Arch Toxicol. 93(1):217-218.

Boots B, Russell CW, Green DS. 2019. Effects of microplastics in soil ecosystems: above and below ground. Environ Sci Technol. 53:11496-11506.

Bordós G, Urbányi B, Micsinai A, Kriszt B, Polatai Z, Szabó I, Hantosi Z, Szoboszlay S. 2019. Identification of microplastics in fish ponds and natural freshwater environments of the Carpathian basin, Europe. Chemosphere. 216:110-116.

Borm PA, Driscoll KE. 2019. The hazards and risks of inhaled poorly soluble particles - where do we stand after 30 years of research? Part Fibre Toxicol. 16(1):11.

Boucher J, Friot D. 2017. Primary microplastics in the oceans: A global evaluation of sources. Gland, (CH): [IUCN] International Union for Conservation of Nature. 43 p.

Bour A, Haarr A, Keiter S, Hylland K. 2018. Environmentally relevant microplastic exposure affects sediment-dwelling bivalves. Environ Pollut. 236:652-660.

Bourdages MPT, Provencher JF, Sudlovenick E, Ferguson SH, Young BG, Pelletier N, Murphy MJJ, D'Addario A, Vermaire JC. 2020. No plastics detected in seal (Phocidae) stomachs harvested in the eastern Canadian Arctic. Mar Pollut Bull. 150:110772.

Bouwman H, Minnaar K, Bezuidenhout C, Verster C. 2018. Microplastics in freshwater environments – A Scoping Study. Technical Report. Republic of South Africa: Water Research Commission. 62 p. Report No. 2610/1/18.

Braeuning A. 2019. Uptake of microplastics and related health effects: a critical discussion of Deng et al., Scientific reports 7:46687, 2017. Arch Toxicol. 93(1):219-220.

Brandão ML, Braga KM, Luque JL. 2011. Marine debris ingestion by Magellenic penguins, *Spheniscus magellanicus*, from the Brazilian coastal zone. Mar Pollut Bull. 62(10):2246-2249.

Brennecke D, Duarte B, Paiva F, Caçador I, Canning-Clode J. 2016. Microplastics as vector for heavy metal contamination from the marine environment. Estuar Coast Shelf Sci. 178:189-195.

Brodie EL, DeSantis TZ, Parker JP, Zubietta IX, Piceno YM, Andersen GL. 2007. Urban aerosols harbor diverse and dynamic bacterial populations. Proc Natl Acad Sci U S A. 104(1):299-304. [cited in Prata 2018].

Brown DM, Wilson MR, MacNee W, Stone V, Donaldson K. 2001. Size-dependent proinflammatory effects of ultrafine polystyrene particles: a role for surface area and oxidative stress in the enhanced activity of ultrafines. Toxicol Appl Pharmacol. 175(3):191-199.

Browne MA, Dissanayake A, Galloway TS, Lowe DM, Thompson RC. 2008. Ingested microscopic plastic translocates to the circulatory system of the mussel, *Mytilus edulis* (L.). Environ Sci Technol. 42(13):5026-5031.

Browne MA, Galloway TS, Thompson RC. 2010. Spatial Patterns of Plastic Debris along Estuarine Shorelines. Environ Sci Technol. 44(9):3404-3409.

Burns EE, Boxall ABA. 2018. Microplastics in the aquatic environment: Evidence for or against adverse impacts and major knowledge gaps. Environ Toxicol Chem. 37(11):2776-2796.

Cai L, Wang J, Peng J, Tan Z, Zhan Z, Tan X, Chen Q. 2017. Characteristic of microplastics in the atmospheric fallout from Dongguan City, China: Preliminary research and first evidence. Environ Sci Pollut Res Int. 24(32):24928-24935.

Campbell SH, Williamson PR, Hall BD. 2017. Microplastics in the gastrointestinal tracts of fish and the water from an urban prairie creek. Facets (Ott). 2:395-409.

Carney Almroth BM, Åström L, Roslund S, Petersson H, Johansson M, Persson NK. 2018. Quantifying shedding of synthetic fibres from textiles; a source of microplastics released into the environment. Environ Sci Pollut Res Int. 25(2):1191-1199.

Catarino AI, Macchia V, Sanderson WG, Thompson RC, Henry TB. 2018. Low levels of microplastics (MP) in wild mussels indicate that MP ingestion by humans is minimal compared to exposure via household fibres fallout during a meal. Environ Pollut. 237:675-684.

Cau A, Avio CG, Dessì C, Follesa MC, Moccia D, Regoli F, Pusceddu A. 2019. Microplastics in the crustaceans *Nephrops norvegicus* and *Aristeus antennatus*: Flagship species for deep-sea environments? Environ Pollut. 255(Pt 1):113107.

[CBD] Secretariat of the Convention on Biological Diversity. 2012. Impacts of Marine Debris on Biodiversity: Current Status and Potential Solutions. Technical Series No. 67. Montreal (QC): CBD. 61 p.

[CBD] Secretariat of the Convention on Biological Diversity. 2016. Marine debris: Understanding, preventing, and mitigating the significant adverse impacts on marine and coastal biodiversity. Technical Series No. 83. Montreal (QC): CBD. 78 p.

[CCME] Canadian Council of Ministers of the Environment. 2018. Strategy on Zero Plastic Waste. Winnipeg (MB): CCME. 14 p. Report No. PN 1583.

Cesa FS, Turra A, Checon HH, Leonardi B, Baruque-Ramos J. 2020. Laundering and textile parameters influence fibers release in household washings. Environ Pollut. 257:113553.

Chae Y, Hong SH, An Y-J. 2020. Photosynthesis enhancement in four marine microalgal species exposed to expanded polystyrene leachate. Ecotoxicol Environ Saf. 189:109936.

Chagnon C, Thiel M, Antunes J, Ferreira JL, Sobral P, Ory NC. 2018. Plastic ingestion and trophic transfer between Easter Island flying fish (*Cheilopogon rapanouiensis*) and yellowfin tuna (*Thunnus albacares*) from Rapa Nui (Easter Island). Environ Pollut. 243(Pt A):127-133.

Chapron L, Peru E, Engler A, Ghiglione JF, Meistertzheim AL, Pruski AM, Purser A, Vétion G, Galand PE, Lartaud F. 2018. Macro- and microplastics affect cold-water corals growth, feeding, and behaviour. Sci Rep. 8(1):15299.

Chen H, Wang S, Guo H, Lin H, Zhang Y, Long Z, Huang H. 2019. Study of marine debris around a tourist city in East China: Implication for waste management. Sci Total Environ. 676:278-289.

Chen H, Wang Y, Sun X, Peng Y, Xiao L. 2020. Mixing effect of polylactic acid microplastic and straw residue on soil property and ecological function. Chemosphere. 243:125271.

Chen J, Tan M, Nemmar A, Song W, Dong M, Zhang G, Li Y. 2006. Quantification of extrapulmonary translocation of intratracheal-instilled particles in vivo in rats: effect of lipopolysaccharide. Toxicology. 222(3):195-201.

Chen Q, Allgeier A, Yin D, Hollert H. 2019. Leaching of endocrine disrupting chemicals from marine microplastics and mesoplastics under common life stress conditions. Environ Int. 130:104938.

Chen Y, Liu X, Leng Y, Wang J. 2020. Defense responses in earthworms (*Eisenia fetida*) exposed to low-density polyethylene microplastics in soils. Ecotoxicol Environ Saf. 187:109788.

Choy CA, Drazen JC. 2013. Plastic for dinner? Observations of frequent debris ingestion by pelagic predatory fishes from the central North Pacific. Mar Ecol Prog Ser. 485:155-163.

Chiba S, Saito H, Fletcher R, Yogi T, Kayo M, Miyagi S, Ogido M, Fujikura K. 2018. Human footprint in the abyss: 30 year records of deep-sea plastic debris. Mar Policy. 96:204-212.

Chua EM, Shimeta J, Nugegoda D, Morrison PD, Clarke BO. 2014. Assimilation of polybrominated diphenyl ethers from microplastics by the marine amphipod, *Allorchestes compressa*. Environ Sci Technol. 48(14):8127-8134.

Cole M, Lindeque PK, Fileman E, Halsband C, Galloway TS. 2015. The impact of polystyrene microplastics on feeding, function and fecundity in the marine copepod *Calaus helgolandicus*. Environ Sci Technol. 49(2):1130-1137.

Cole M, Lindeque PK, Fileman E, Clark J, Lewis C, Halsband C, Galloway TS. 2016. Microplastics alter the properties and sinking rates of zooplankton faecal pellets. Environ Sci Technol. 50:3239-3246.

Collicutt B, Juanes F, Dudas SE. 2019. Microplastics in juvenile Chinook salmon and their nearshore environments on the east coast of Vancouver Island. Environ Pollut. 244:135-142.

Cong Y, Jin F, Tian M, Wang J, Shi H, Wang Y, Mu J. 2019. Ingestion, egestion and post-exposure effects of polystyrene microspheres on marine medaka (*Oryzias melastigma*). Chemosphere. 228:93-100.

Connors EJ. 2017. Distribution and biological implications of plastic pollution on the fringing reef of Mo'orea, French Polynesia. PeerJ. 5:e3733.

Cook L, Chatterjee A, Barnes A, Yarwood J, Hu WS, Dunny G. 2011. Biofilm growth alters regulation of conjugation by a bacterial pheromone. Mol Microbiol. 81(6):1499-1510.

Corcoran PL, Norris T, Ceccanese T, Walzak MJ, Helm PA, Marvin CH. 2015. Hidden plastics of Lake Ontario, Canada and their potential preservation in the sediment record. Environ Pollut. 204:17-25.

Corcoran PL, Belontz SL, Ryan K, Walzak MJ. 2020. Factors controlling the distribution of microplastic particles in benthic sediment of the Thames River, Canada. Environ Sci Technol. 54:818-825.

Corradini F, Meza P, Eguiluz R, Casado F, Huerta-Lwanga E, Geissen V. 2019. Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. Sci Total Environ. 671:411-420.

Cosgrove L, McGeechan PL, Robson GD, Handley PS. 2007. Fungal communities associated with degradation of polyester polyurethane in soil. Appl Environ Microbiol. 73(18):5817-8524.

Covernton GA, Pearce CM, Gurney-Smith HJ, Chastain SG, Ross PS, Dower JF, Dudas SE. 2019. Size and shape matter: A preliminary analysis of microplastic sampling technique in seawater studies with implications for ecological risk assessment. Sci Total Environ. 667:124-132.

Cox KD, Covernton GA, Davies HL, Dower JF, Juanes F, Dudas SE. 2019. Human consumption of microplastics. Environ Sci Technol. 53(12):7068-7074.

Crittenden JC, Rhodes Trussell R, Hand DW, Howe KJ, Tchobanoglous G. 2012. MWH's Water Treatment Principles and Design. Third Edition. Hoboken (NJ): Wiley & Sons Inc. 1920 p.

Crossman J, Hurley RR, Futter M, Nizzetto L. 2020. Transfer and transport of microplastics from biosolids to agricultural soils and the wider environment. Sci Total Environ. 724:138334.

Curren E, Leong SCY. 2019. Profiles of bacterial assemblages from microplastics of tropical coastal environments. Sci Total Environ. 655:313-320.

Cuthbert RN, Al-Jaibachi R, Dalu T, Dick JTA, Callaghan A. 2019. The influence of microplastics on trophic interaction strengths and oviposition preferences of dipterans. Sci Total Environ. 651(Pt 2):2420-2423.

da Costa JP, Paço A, Santos PSM, Duarte AC, Rocha-Santos T. 2019. Microplastics in soils: assessment, analytics and risks. Environ Chem. 16(1):18-30.

da Costa Araújo AP, Gomes A, Malafaia G. 2020. Hepatotoxicity of pristine polyethylene microplastics in neotropical physalaemus cuvieri tadpoles (Fitzinger, 1826). J Hazard Mater. 386:121992.

Dai Z, Zhang H, Zhou Q, Tian Y, Chen T, Tu C, Fu C, Luo Y. 2018. Occurrence of microplastics in the water column and sediment in an inland sea affected by intensive anthropogenic activities. Environ Poll. 242(Pt B):1557-1565.

Dawson AL, Kawaguchi S, King CK, Townsend KA, King R, Huston WM, Bengtson Nash SM. 2018. Turning microplastics into nanoplastics through digestive fragmentation by Antarctic krill. Nat Commun. 9:1001.

de Haan WP, Sanchez-Vidal A, Canals M, NUREIEV1 Shipboard Scientific Party. 2019. Floating microplastics and aggregate formation in the Western Mediterranean Sea. Mar Pollut Bull. 140:523-535.

De Falco F, Gullo MP, Gentile G, Di Pace E, Cocca M, Gelabert L, Brouta-Agnésa M, Rovira A, Escudero R, Villalba R, et al. 2018. Evaluation of microplastic release caused by textile washing processes of synthetic fabrics. Environ Pollut. 236:916-925.

De Falco F, Cocca M, Avella M, Thompson RC. 2020. Microfiber release to water, via laundering, and to air, via everyday use: A comparison between polyester clothing with differing textile parameters. Environ Sci Technol. 54(6):3288-3296.

De Felice B, Bacchetta R, Santo N, Tremolada P, Parolini M. 2018. Polystyrene microplastics did not affect body growth and swimming activity in *Xenopus laevis* tadpoles. Environ Sci Pollut Res Int. 25(34):34644-34651.

De Felice B, Sabatini V, Antenucci S, Gattoni G, Santo N, Bacchetta R, Ortenzi MA, Parolini M. 2019. Polystyrene microplastics ingestion induced behavioral effects to the cladoceran *Daphnia magna*. Chemosphere. 231:423-431.

de Scisciolo T, Mijts EN, Becker T, Eppinga MB. 2016. Beach debris on Aruba, Southern Caribbean: Attribution to local land-based and distal marine-based sources. Mar Pollut Bull. 106(1-2):49-57.

de Souza Machado AA, Lau CW, Kloas W, Bergmann J, Bachelier JB, Faltin E, Becker R, Görlich AS, Rillig MC. 2019. Microplastics can change soil properties and affect plant performance. Environ Sci Technol. 53(10):6044-6052.

De Tender CA, Devriese LI, Haegeman A, Maes S, Ruttink T, Dawyndt P. 2015. Bacterial community profiling of plastic litter in the Belgian part of the North Sea. Environ Sci Technol. 49(16):9629-9638.

de Vries AN, Govoni D, Árnason SH, Carlsson P. 2020. Microplastic ingestion by fish: Body size, condition factor and gut fullness are not related to the amount of plastics consumed. Mar Pollut Bull. 151:110827.

De Witte B, Devriese L, Bekaert K, Hoffman S, Vandermeersch G, Cooreman K, Robbens J. 2014. Quality assessment of the blue mussel (*Mytilus edulis*): comparison between commercial and wild types. Mar Pollut Bull. 85(1):146-155.

Dean BY, Corcoran PL, Helm PA. 2018. Factors influencing microplastic abundances in nearshore, tributary and beach sediments along the Ontario shoreline of Lake Erie. J Great Lakes Res. 44(5):1002-1009.

Delacuvellerie A, Cyriaque V, Gobert S, Benali S, Wattiez R. 2019. The plastisphere in marine ecosystem hosts potential specific microbial degraders including *Alcanivorax borkumensis* as a key player for the low-density polyethylene degradation. J Hazard Mater. 380:120899.

DeMatteo R, Keith MM, Brophy JT, Wordsworth A, Watterson AE, Beck M, Ford AR, Gilbertson M, Pharityal J, Rootham M, et al. 2012. Chemical exposures of women workers in the plastics industry with particular reference to breast cancer and reproductive hazards. New Solut. 22(4):427-448.

Demers PA, Schade WJ, Demers RY. 1994. Lymphocytopenia and occupational exposures among pattern and model makers. Scand J Work Environ Health. 20(2):107-112.

Deng Y, Zhang Y, Lemos B, Ren H. 2017. Tissue accumulation of microplastics in mice and biomarker responses suggest widespread health risks of exposure. Sci Rep. 7:46687.

Deng Y, Zhang Y, Qiao R, Bonilla MM, Yang X, Ren H, Lemos B. 2018. Evidence that microplastics aggravate the toxicity of organophosphorus flame retardants in mice (*Mus musculus*). J Hazard Mater. 357:348-354.

Desforges J-PW, Galbraith M, Ross PS. 2015. Ingestion of microplastics by zooplankton in the northeast Pacific Ocean. Archi Environ Contam Toxicol. 69(3):320-330.

Devriese LI, van der Meulen MD, Maes T, Bekaert K, Paul-Pont I, Frère L, Robbens J, Vethaak AD. 2015. Microplastic contamination in brown shrimp (*Crangon crangon*, Linnaeus 1758) from coastal waters of the Southern North Sea and Channel area. Mar Pollut Bull. 98(1-2):179-187.

Devriese LI, De Witte B, Vethaak AD, Hostens K, Leslie HA. 2017. Bioaccumulation of PCBs from microplastics in Norway lobster (*Nephrops norvegicus*): An experimental study. Chemosphere. 186:10-16.

Di M, Wang J. 2018. Microplastics in surface waters and sediments of the three gorges reservoir, China. Sci Total Environ. 616-617:1620-1627.

Diepens NJ, Koelmans AA. 2018. Accumulation of plastic debris and associated contaminants in aquatic food webs. Environ Sci Technol. 52(15):8510-8520.

Dillon Consulting. 2019. Street Litter Audits – 2019 Results. Final Report. Vancouver (BC): City of Vancouver. 71 p.

Donaldson K. 2009. The inhalation toxicology of *p*-aramid fibrils. Crit Rev Toxicol. 39(6):487-500.

Doyle D, Frias J, Nash R, Gammell M. 2020. Current environmental microplastic levels do not alter emergence behaviour in the intertidal gastropod *Littorina littorea*. Mar Pollut Bull. 151:110859.

Drever MC, Provencher JF, O'Hara PD, Wilson L, Bowes V, Bergman CM. 2018. Are ocean conditions and plastic debris resulting in a "double whammy" for marine birds? Mar Pollut Bull. 133:684-692.

Driedger AGJ, Dürr HH, Mitchell K, Van Cappellen P. 2015. Plastic debris in the Laurentian Great Lakes: A review. J Great Lakes Res. 41(1):9-19.

Dris R, Gasperi J, Rocher V, Saad M, Renault N, Tassin B. 2015. Microplastic contamination in an urban area: A case study in Greater Paris. Environ Chem. 12(5):592-599.

Dris R, Gasperi J, Saad M, Mirande C, Tassin B. 2016. Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? Mar Pollut Bull. 104(1-2):290-293.

Dris R, Gasperi J, Mirande C, Mandin C, Guerrouache M, Langlois V, Tassin B. 2017. A first overview of textile fibers, including microplastics, in indoor and outdoor environments. Environ Pollut. 221:453-458.

Duncan EM, Arrowsmith JA, Bain CE, Bowdery H, Broderick AC, Chalmers T, Fuller WJ, Galloway TS, Lee JH, Lindeque PK, et al. 2019. Diet-related selectivity of macroplastic ingestion in green turtles (*Chelonia mydas*) in the eastern Mediterranean. Sci Rep. 9(1):11581.

Earthwatch Europe. 2018. Plastic Rivers reducing the plastic pollution on our doorstep [PDF]. Oxford (GB): Earthwatch Institute. 20 p. [accessed 2019 May 15].

[ECCC] Environment and Climate Change Canada. 2015. Microbeads – A Science Summary. Ottawa (ON): Government of Canada. 34 p.

[ECCC] Environment and Climate Change Canada. 2019b. Canada's Plastics Science Agenda. Ottawa (ON): Government of Canada. 23 p.

[ECCC] Environment and Climate Change Canada. 2019a. Economic study of the Canadian plastic industry, markets and waste. Summary Report. Ottawa (ON): Government of Canada. 43 p.

[ECCC] Environment and Climate Change Canada. 2019c. Economic study of the Canadian plastic industry, markets and waste. Final Report - Task 1. Ottawa (ON): Government of Canada. 186 p. Available from: ec.pddemandes-pd-requests.ec@canada.ca.

[ECCC] Environment and Climate Change Canada. 2019d. Economic study of the Canadian plastic industry, markets and waste. Final Report - Task 2. Ottawa (ON): Government of Canada. 121 p. Available from: ec.pddemandes-pd-requests.ec@canada.ca.

[ECCC] Environment and Climate Change Canada. 2019e. Economic study of the Canadian plastic industry, markets and waste. Final Report - Task 5. Ottawa (ON): Government of Canada. 62 p. Available from: ec.pddemandes-pd-requests.ec@canada.ca.

[ECHA] European Chemicals Agency. 2019. ANNEX to the ANNEX XV Restriction Report - Proposal for a restriction on intentionally added microplastics. Helsinki (FI): ECHA. 299 p.

Eckert EM, Di Cesare A, Kettner MT, Arias-Andres M, Fontaneto D, Grossart HP, Corno G. 2018a. Microplastics increase impact of treated wastewater on freshwater microbial community. Environ Pollut. 234:495-502.

Eckert EM, Di Cesare, A, Coci M, Corno G. 2018b. Persistence of antibiotic resistance genes in large subalpine lakes: The role of anthropogenic pollution and ecological interactions. Hydrobiologia. 824(1):93-108.

[EFSA] European Food Safety Authority. 2016. Presence of microplastics and nanoplastics in food, with particular focus on seafood. EFSA J. 14(6):e04501.

Egeland GM and [CINE] Centre for Indigenous Peoples' Nutrition and Environment. 2010a. Inuit Health Survey 2007-2008 - Inuvialuit Settlement Region. [PDF] Ste-Anne-de-Bellevue (QC): CINE. 31 p.

Egeland GM and [CINE] Centre for Indigenous Peoples' Nutrition and Environment. 2010b. Inuit Health Survey 2007-2008 - Nunatsiavut. [PDF] Ste-Anne-de-Bellevue (QC): CINE. 35 p.

Egeland GM and [CINE] Centre for Indigenous Peoples' Nutrition and Environment. 2010c. Inuit Health Survey 2007-2008 - Nunavut. [PDF] Ste-Anne-de-Bellevue (QC): CINE. 51 p.

Ekebafe LO, Ogbeifun DE, Okieimen FE. 2011. Polymer applications in agriculture. Biokemistri. 23(2):81-89.

Engler RE. 2012. The complex interaction between marine debris and toxic chemicals in the ocean. Environ Sci Technol. 46(22):12302-12315. [cited in Galloway 2015].

Eriksen M, Lebreton LCM, Carson HS, Thiel M, Moore CJ, Borerro JC, Galgani F, Ryan PG, Reisser J. 2014. Plastic pollution in the world's oceans: More than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. PLoS One. 9(12):e111913.

Erni-Cassola G, Gibson MI, Thompson RC, Christie-Oleza JA. 2017. Lost, but found with Nile Red: A novel method for detecting and quantifying small microplastics (1 mm to 20 µm) in environmental samples. Environ Sci Technol. 51(23):13641-13648.

Esiukova E. 2017. Plastic pollution on the Baltic beaches of Kaliningrad region, Russia. Mar Pollut Bull. 114(2):1072-1080.

Espinosa C, Esteban MÁ, Cuesta A. 2019. Dietary administration of PVC and PE microplastics produces histological damage, oxidative stress and immunoregulation in European sea bass. Fish Shellfish Immunol. 95:574-583.

Eubeler JP, Bernhard M, Knepper TP. 2010. Environmental biodegradation of synthetic polymers II. Biodegradation of different polymer groups. Trends Anal Chem. 29:84-100.

European Commission. 2018. Report from the Commission to the European Parliament and the Council on the impact of the use of oxo-degradable plastic, including oxo-degradable plastic carrier bags, on the environment. Final Report. Brussels (BE): European Commission. 8 p.

European Commission. 2019. Environmental and Health Risks of Microplastic Pollution. Independent Expert Report. Brussels (BE): European Commission. 60 p.

Farrell P, Nelson K. 2013. Trophic level transfer of microplastic: *Mytilus edulis* (L.) to *Carcinus maenas* (L.). Environ Pollut. 177:1-3.

[FAO] Food and Agriculture Organization of the United Nations. 2017. Microplastics in fisheries and aquaculture: Status of knowledge on their occurrence and implications for aquatic organisms and food safety. Technical Report. Rome (IT): FAO. 126 p. Report No. 615.

Fastelli P, Blašković A, Benardi G, Romeo T, Čižmek H, Andaloro F, Russo GF, Guerranti C, Renzi M. 2016. Plastic litter in sediments from a marine area likely to become protected (Aeolian Archipelago's islands, Tyrrhenian sea). Mar Pollut Bull. 113(1-2):526-529.

Feng Z, Zhang T, Li Y, He X, Wang R, Xu J, Gao G. 2019. The accumulation of microplastics in fish from an important fish farm and mariculture area, Haizhou Bay, China. Sci Total Environ. 696:133948.

Fischer M, Goßmann I, Scholz-Böttcher M. 2019. Fleur de Sel - An interregional monitor for microplastics mass load and composition in European coastal waters? J Anal Appl Pyrolysis. 144:104711.

Flemming HC, Wingender J. 2010. The biofilm matrix. Nat Rev Microbiol. 8(9):623-633.

Foley CJ, Feiner ZS, Malinich TD, Höök TO. 2018. A meta-analysis of the effects of exposure to microplastics on fish and aquatic invertebrates. Sci Total Environ. 631-632:550-559.

Fournier S, D'Errico J, Stapleton P. 2018. Translocation of engineered nanomaterials from the maternal lungs to the fetal compartment after instillation. FASEB J. 32(Suppl. 1). Abstract No. 830.5.

Frydkjær CK, Iversen N, Roslev P. 2017. Ingestion and egestion of microplastics by the cladoceran *Daphnia magna*: Effects of regular and irregular shaped plastic and sorbed phenanthrene. Bull Environ Contam Toxicol. 99:655-661.

Gago J, Portela S, Filgueiras AV, Pauly Salinas M, Macías D. 2020. Ingestion of plastic debris (macro and micro) by longnose lancetfish (*Alepisaurus ferox*) in the North Atlantic Ocean. Reg Stud Mar Sci. 33:100977.

Gall SC, Thompson RC. 2015. The impact of debris on marine life. Mar Pollut Bull. 92(1-2):170-179.

Galloway TS. 2015. Micro- and nano-plastics and human health. In: Bergmann M, Gutow L, Klages M, editors. Marine Anthropogenic Litter. Cham (CH): Springer. Chapter 13. p. 343-366.

Gambardella C, Morgana S, Bramini M, Rotini A, Manfra L, Migliore L, Piazza V, Garaventa F, Faimali M. 2018. Ecotoxicological effects of polystyrene microbeads in a battery of marine organisms belonging to different trophic levels. Mar Environ Res. 141:313-321.

Garcia-Garin O, Vighi M, Sala B, Aguilar A, Tsangaris C, Digka N, Kaberi H, Eljarrat E, Borrell A. 2020. Assessment of organophosphate flame retardants in Mediterranean *Boops boops* and their relationship to anthropization levels and microplastic ingestion. Chemosphere. 252:126569.

García-Rivera S, Lizaso JLS, Millán JMB. 2018. Spatial and temporal trends of marine litter in the Spanish Mediterranean seafloor. Mar Pollut Bull. 2018. 137:252-261.

Garrido S, Linares M, Campillo JA, Albentosa M. 2019. Effect of microplastics on the toxicity of chlorpyrifos to the microalgae *Isochrysis galbana*, clone t-ISO. Ecotoxicol Environ Saf. 173:103-109.

Gasperi J, Dris R, Mirande C, Mandin C, Langlois V, Tassin B. 2015. First overview of microplastics in indoor and outdoor air. In: Paper Presented at 15th EuCheMS International Conference on Chemistry and the Environment (ICCE); 2015 Sep 20-24; Leipzig (DE): ICCE.

Gasperi J, Wright SL, Dris R, Collard F, Mandin C, Guerrouache M, Langlois V, Kelly FJ, Tassin B. 2018. Microplastics in air: Are we breathing it in? Curr Opin Environ Sci Health. 1:1-5.

[GCSC] Great Canadian Shoreline Cleanup. 2018a. Our History. Vancouver (BC): Ocean Wise. [accessed 2019 May 15].

[GCSC] Great Canadian Shoreline Cleanup. 2018b. 2018 Annual Report. Vancouver (BC): Ocean Wise. 15 p.

[GCSC] Great Canadian Shoreline Cleanup. [date unknown]. Site Coordinator Guide. Vancouver (BC): Ocean Wise. 17 p.

Gerdes Z, Ogonowski M, Nybom I, Ek C, Adolfsson-Erici M, Barth A, Gorokhova E. 2019. Microplastic-mediated transport of PCBs? A depuration study with *Daphnia magna*. PLoS One. 14(2):e0205378.

[GESAMP] Joint Group of Experts on the Scientific Aspects. 2016. Sources, fate and effects of microplastics in the marine environment: Part two of a global assessment. In: Kershaw PJ, Rochman CM, editors. IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP GESAMP, Reports and Studies No. 93. London, England: International Maritime Organization. 220 p.

Gewert B, Plassmann MM, MacLeod M. 2015. Pathways for degradation of plastic polymers floating in the marine environment. Environ Sci Process Impacts. 17(9):1513-1521.

Geyer R, Jambeck JR, Law KL. 2017. Production, use, and fate of all plastics ever made. Sci Adv. 3(7):e1700782.

Gies EA, LeNoble JL, Noël M, Etemadifar A, Bishay F, Hall ER, Ross PS. 2018. Retention of microplastics in a major secondary wastewater treatment plant in Vancouver, Canada. Mar Pollut Bull. 133:553-561.

Godoy V, Blázquez G, Calero M, Quesada L, Martín-Lara MA. 2019. The potential of microplastics as carriers of metals. Environ Pollut. 255(Pt 3):113363.

Gong M, Yang G, Zhuang L, Zeng EY. 2019. Microbial biofilm formation and community structure on low-density polyethylene microparticles in lake water microcosms. Environ Pollut. 252(Pt A):94-102.

Good TP, June JA, Etnier M, Broadhurst G. 2007. Quantifying the impact of derelict fishing gear on the marine fauna of Puget Sound and the Northwest Straits. In: Proceedings of the International Council for the Exploration of the Sea (ICES) Annual Science Conference; 2007 Sep 17-21; Helsinki (FI): ICES.

Goodman AJ, Walker TR, Brown CJ, Wilson BR, Gazzola V, Sameoto JA. 2020. Benthic marine debris in the Bay of Fundy, eastern Canada: Spatial distribution and categorization using seafloor video footage. Mar Pollut Bull. 150:110722.

Gouin T. 2020. Toward an improved understanding of the ingestion and trophic transfer of microplastic particles: Critical review and implications for future research. Environ Toxicol Chem. 39(6):1119-1137.

Gouin T, Becker RA, Collot AG, Davis JW, Howard B, Inawaka K, Lampi M, Ramon BS, Shi J, Hopp PW. 2019. Towards the development and application of an environmental risk assessment framework for microplastic. Environ Toxicol Chem. 38(10):2087-2100.

Grbić J, Helm P, Athey S, Rochman CM. 2020. Microplastics entering northwestern Lake Ontario are diverse and linked to urban sources. Water Res. 174:115623.

Grigorakis S, Drouillard KG. 2018. Effect of microplastic amendment to food on diet assimilation efficiencies of PCBs by fish. Environ Sci Technol. 52(18):10796-10802.

Grigorakis S, Mason SA, Drouillard KG. 2017. Determination of the gut retention of plastic microbeads and microfibers in goldfish (*Carassius auratus*). Chemosphere. 169:233-238.

Gündoğdu S. 2018. Contamination of table salts from Turkey with microplastics. Food Addit Contam Part A Chem Anal Control Expo Risk Assess. 35(5):1006-1014.

Guo X, Wang J. 2019a. The chemical behaviours of microplastics in marine environment: A review. Mar Pollut Bull. 142:1-14.

Guo X, Wang J. 2019b. Sorption of antibiotics onto aged microplastics in freshwater and seawater. Mar Pollut Bull. 149:110511.

Hahladakis JN, Velis CA, Weber R, Iacovidou E, Purnell P. 2018. An overview of chemical additives present in plastics: Migration, release, fate and environmental impact during their use, disposal and recycling. J Hazard Mater. 344:179-199. [cited in WHO 2019].

Hall-Stoodley L, Costerton JW, Stoodley P. 2004. Bacterial biofilms: From the natural environment to infectious diseases. Nat Rev Microbiol. 2(2):95-108.

Hallanger IG, Gabrielsen GW. 2018. Plastic in the European Arctic. Brief Report. Tromsø (NO): Norwegian Polar Institute. 23 p.

Hämer J, Gutow L, Köhler A, Saborowski R. 2014. Fate of microplastics in the marine isopod *Idotea emarginata*. Environ Sci Technol. 48:13451-13458.

Hammer S, Nager RG, Johnson PCD, Furness RW, Provencher JF. 2016. Plastic debris in great skua (*Stercorarius skua*) pellets corresponds to seabird prey species. Mar Pollut Bull. 103(1-2):206-210.

Hanachi P, Karbalaei S, Walker TR, Cole M, Hosseini SV. 2019. Abundance and properties of microplastics found in commercial fish meal and cultured common carp (*Cyprinus carpio*). Environ Sci Pollut Res Int. 26(23):23777-23787.

Hankins C, Duffy A, Drisco K. 2018. Scleractinian coral microplastic ingestion: Potential calcification effects, size limits, and retention. Mar Pollut Bull. 135:587-593.

Hansen E, Nilsson NH, Lithner D, Lassen C. 2013. Hazardous substances in plastic materials. Vejle (DK): Norwegian Climate and Pollution Agency. 151 p. Report No. TA 3017. [cited in WHO 2019].

Hantoro I, Löhr AJ, Van Belleghem FGAJ, Widianarko B, Ragas AMJ. 2019. Microplastics in coastal areas and seafood: implications for food safety. Food Addit Contam Part A Chem Anal Control Expo Risk Assess. 36(5):674-711.

Hara J, Frias J, Nash R. 2020. Quantification of microplastic ingestion by the decapod crustacean *Nephrops norvegicus* from Irish waters. Mar Pollut Bull. 152:110905.

Harrison JP, Schratzberger M, Sapp M, Osborn AM. 2014. Rapid bacterial colonization of low-density polyethylene microplastics in coastal sediment microcosms. BMC Microbiol. 14:232.

Harrison JP, Hoellein TJ, Sapp M, Tagg AS, Ju-Nam Y, Ojeda JJ. 2018. Microplastic-associated biofilms: a comparison of freshwater and marine environments. Handb Environ Chem. 58:181-201.

Helm PA. 2017. Improving microplastics source apportionment: a role for microplastic morphology and taxonomy? Anal Methods. 9:1328-1331.

Helmroth E, Rijk R, Dekker M, Jongen W. 2002. Predictive modelling of migration from packaging materials into food products for regulatory purposes. Trends Food Sci Technol. 13(3):102-109. [cited in Galloway 2015].

Hendrickson E, Minor EC, Schreiner K. 2018. Microplastic abundance and composition in Western Lake Superior as determined via microscopy, Pyr-GC/MS, and FTIR. Environ Sci Technol. 52(4):1787-1796.

Henry B, Laitala K, Klepp IG. 2019. Microfibres from apparel and home textiles: Prospects for including microplastics in environmental sustainability assessment. Sci Total Environ. 652:483-494.

Hermabessiere L, Paul-Pont I, Cassone AL, Himber C, Receveur J, Jezequel R, Rakwe ME, Rinnert E, Rivière G, Lambert C, et al. 2019. Microplastic contamination and pollutant levels in mussels and cockles collected along the channel coasts. Environ Pollut. 250:807-819.

Hermsen E, Mintenig SM, Besseling E, Koelmans AA. 2018. Quality criteria for the analysis of microplastic in biota samples: A critical review. Environ Sci Technol. 52(18):10230-10240.

Hernandez LM, Xu EG, Larsson HCE, Tahara R, Maisuria VB, Tufenkji N. 2019. Plastic Teabags Release Billions of Microparticles and Nanoparticles into Tea. Environ Sci Technol. 53(21):12300-12310.

Hernandez-Milian G, Lusher A, MacGabban S, Rogan E. 2019. Microplastics in grey seal (*Halichoerus grypus*) intestines: Are they associated with parasite aggregations?. Mar Pollut Bull. 146:349-354.

Hesterberg TW, McConnell EE, Miller WC, Hamilton R, Bunn WB. 1992. Pulmonary toxicity of inhaled polypropylene fibers in rats. Fundam Appl Toxicol. 19(3):358-366.

Hidalgo-Ruz V, Gutow L, Thompson RC, Thiel M. 2012. Microplastics in the marine environment: a review of the methods used for identification and quantification. Environ Sci Tech. 46(6):3060-3075.

Hipfner JM, Galbraith M, Tucker S, Studholme KR, Domalik AD, Pearson SF, Good TP, Ross PS, Hodum P. 2018. Two forage fishes as potential conduits for the vertical transfer of microfibers in Northeastern Pacific Ocean food webs. Environ Pollut. 239:215-222.

Hoellein T, Rojas M, Pink A, Gasior J, Kelly J. 2014. Anthropogenic litter in urban freshwater ecosystems: Distribution and microbial interactions. PLoS One. 9(6):e98485.

Horn D, Miller M, Anderson S, Steele C. 2019. Microplastics are ubiquitous on California beaches and enter the coastal food web through consumption by Pacific mole crabs. Mar Pollut Bull. 139:231-237.

Horton AA, Dixon SJ. 2017. Microplastics: An introduction to environmental transport processes. WIREs Water. 5(2):e1268.

Horton AA, Newbold LK, Palacio-Cortés AM, Spurgeon DJ, Pereira MG, Carter H, Gweon HS, Vijver MG, van Bodegom PM, Navarro da Silva MA, et al. 2020. Accumulation of polybrominated ethers and microbiome response in the great pond snail *Lymnaea stagnalis* with exposure to nylon (polyamide) microplastics. Ecotoxicol Environ Saf. 188:109882.

Hossain MS, Rahman MS, Uddin MN, Sharifuzzaman SM, Chowdhury SR, Sarker S, Nawaz Chowdhury MS. 2020. Microplastic contamination in Penaeid shrimp from the Northern Bay of Bengal. Chemosphere. 238:124688.

Huerta Lwanga E, Gertsen H, Gooren H, Peters P, Salánki T, van der Ploeg M, Besseling E, Koelmans AA, Geissen V. 2016. Microplastics in the terrestrial ecosystem: Implications for *Lumbricus terrestris* (Oligochaeta, Lumbricidae). Environ Sci Technol. 50(5):2685-2691.

Huerta Lwanga E, Gertsen H, Gooren H, Peters P, Salánki T, van der Ploeg M, Besseling E, Koelmans AA, Geissen V. 2017. Incorporation of microplastics from litter into burrows of *Lumbricus terrestris*. Environ Pollut. 220(Pt A):523-531.

Huerta Lwanga E, Thapa B, Yang X, Gertsen H, Salánka T, Geissen V, Garbeva P. 2018. Decay of low-density polyethylene by bacteria extracted from earthworm's guts: A potential for soil restoration. Sci Total Environ. 624:753-757.

Hurley RR, Nizzetto L. 2018. Fate and occurrence of micro(nano)plastics in soils: Knowledge gaps and possible risks. Curr Opin Environ Sci Health. 1:6-11.

Hurley RR, Woodward J, Rothwell JJ. 2018. Microplastic contamination of river beds significantly reduced by catchment-wide flooding. Nat Geosci. 11:251-257.

Hussain N, Jaitley V, Florence AT. 2001. Recent advances in the understanding of uptake of microparticulates across the gastrointestinal lymphatics. Adv Drug Deliv Rev. 50(1-2):107-142.

Imran M, Das KR, Naik MM. 2019. Co-selection of multi-antibiotic resistance in bacterial pathogens in metal and microplastic contaminated environments: An emerging health threat. Chemosphere. 215:846-857.

Iñiguez ME, Conesa JA, Fullana A. 2017. Microplastics in Spanish Table Salt. Sci Rep. 7(1):8620.

Jabeen K, Li B, Chen Q, Su L, Wu C, Hollert H, Shi H. 2018. Effects of virgin microplastics on goldfish (*Carassius auratus*). Chemosphere. 213:323-332.

Jacob H, Gilson A, Lanctôt C, Besson M, Metian M, Lecchini D. 2019. No effect of polystyrene microplastics on foraging activity and survival in a post-larvae coral-reef fish, *Acanthurus triostegus*. Bull Environ Contam Toxicol. 102(4):457-461.

Jacobsen JK, Massey L, Gulland F. 2010. Fatal ingestion of floating net debris by two sperm whales (*Physeter macrocephalus*). Mar Pollut Bull. 60(5):765-767.

Jahnke A, Arp HPH, Escher BI, Gewert B, Gorokhova E, Kühnel D, Ogonowski M, Potthoff A, Rummel C, Schmitt-Jansen M, et al. 2017. Reducing uncertainty and confronting ignorance about the possible impacts of weathering plastic in the marine environment. Environ Sci Technol Lett. 4(3):85-90. [cited in WHO 2019].

Jaikumar G, Baas J, Brun NR, Vijver MG, Bosker T. 2018. Acute sensitivity of three Cladoceran species to different types of microplastics in combination with thermal stress. Environ Pollut. 239:733-740.

Jarvis P, Jefferson B, Gregory J, Parsons SA. 2005. A review of floc strength and breakage. Water Res. 39(14):3121-3137.

Jeong CB, Won EJ, Kang HM, Lee MC, Hwang DS, Hwang UK, Zhou B, Souissi S, Lee SJ, Lee JS. 2016. Microplastic size-dependent toxicity, oxidative stress induction, and p-JNK and p-p38 activation in the monogonont rotifer (*Brachionus koreanus*). Environ Sci Technol. 50(16):8849-8857.

Jeong CB, Kang HM, Lee MC, Kim DH, Han J, Hwang DS, Souissi S, Lee SJ, Shin KH, Park HG, et al. 2017. Adverse effects of microplastics and oxidative stress-induced MAPK/Nrf2 pathway-mediated defense mechanisms in the marine copepod *Paracyclopina nana*. Sci Rep. 7:41323.

Jepsen EM, de Bruyn PJN. 2019. Pinniped entanglement in oceanic plastic pollution: A global review. Mar Pollut Bull. 145:295-305.

Jin Y, Lu L, Tu W, Luo T, Fu Z. 2019. Impacts of polystyrene microplastic on the gut barrier, microbiota and metabolism of mice. Sci Total Environ. 649:308-317.

Johansen MP, Cresswell T, Davis J, Howard DL, Howell NR, Prentice E. 2019. Biofilm-enhanced adsorption of strong and weak cations onto different microplastic sample types: Use of spectroscopy, microscopy and radiotracer methods. Wat Res. 158:392-400.

Jones KL, Hartl MGJ, Bell MC, Capper A. 2020. Microplastic accumulation in a *Zostera marina* L. bed at Deerness Sound, Orkney, Scotland. Mar Pollut Bull. 152:110883.

Jovanović B. 2017. Ingestion of microplastics by fish and its potential consequences from a physical perspective. Integr Environ Assess Manag. 13(3):510-515.

Jovanović B, Gökdağ K, Güven O, Emre Y, Whitley EM, Kideys AE. 2018. Virgin microplastics are not causing imminent harm to fish after dietary exposure. Mar Pollut Bull. 130:123-131.

Ju H, Zhu D, Qiao M. 2019. Effects of polyethylene microplastics on the gut microbial community, reproduction and avoidance behaviors of the soil springtail, *Folsomia candida*. Environ Pollut. 247:890-897.

Julienne F, Delorme N, Lagarde F. 2019. From macroplastics to microplastics: role of water in the fragmentation of polyethylene. Chemosphere. 236:124409.

Kaiser D, Kowalski N, Waniek JJ. 2017. Effects of biofouling on the sinking behaviour of microplastics. Environ Res Lett. 12(12):124003.

Kanhai LDK, Gårdfeldtc K, Lyashevska O, Hassellöv M, Thompson RC, O'Connor I. 2018. Microplastics in sub-surface waters of the Arctic Central Basin. Mar Pollut Bull. 130:8-18.

Karami A, Golieskardi A, Ho YB, Larat V, Salamatinia B. 2017a. Microplastics in eviscerated flesh and excised organs of dried fish. Sci Rep. 7(1):5473.

Karami A, Golieskardi A, Keong Choo C, Larat V, Galloway TS, Salamatinia B. 2017b. The presence of microplastics in commercial salts from different countries. Sci Rep. 7:46173.

Karami A, Golieskardi A, Choo CK, Larat V, Karbalaei S, Salamatinia B. 2018. Microplastic and mesoplastic contamination in canned sardines and sprats. Sci Total Environ. 612:1380-1386.

Karbalaei S, Golieskardi A, Watt DU, Boiret M, Hanachi P, Walker TR, Karami A. 2020. Analysis and inorganic composition of microplastics in commercial Malaysian fish meals. Mar Pollut Bull. 150:110687.

Karlsson TM, Vethaak AD, Almroth BC, Ariese F, van Velzen M, Hassellöv M, Leslie HA. 2017. Screening for microplastics in sediment, water, marine invertebrates and fish: method development and microplastic accumulation. Mar Pollut Bull. 122(1-2):403-408.

Karthik R, Robin RS, Purvaja R, Ganguly D, Anandavelu I, Raghuraman R, Hariharan G, Ramakrishna A, Ramesh R. 2018. Microplastics along the beaches of southeast coast of India. Sci Total Environ. 645:1388-1399.

Katsanevakis S, Verriopoulos G, Nicolaidou A, Thessalou-Legaki M. 2007. Effect of marine litter on the benthic megafauna of coastal soft bottoms: A manipulative field experiment. Mar Pollut Bull. 54(6):771-778.

Kay P, Hiscoe R, Moberley I, Bajic L, McKenna N. 2018. Wastewater treatment plants as a source of microplastics in river catchments. Environ Sci Pollut Res Int. 25(20):20264-20267.

Kazmiruk TN, Kazmiruk VD, Bendell LI. 2018. Abundance and distribution of microplastics within surface sediments of a key shellfish growing region of Canada. PLoS One. 13(5):e0196005.

Kedzierski M, Lechat B, Sire O, Le Maguer G, Le Tilly V, Bruzaud S. 2020. Microplastic contamination of packaged meat: Occurrence and associated risks. Food Packag Shelf Life. 24:100489.

Keswani A, Oliver DM, Gutierrez T, Quilliam RS. 2016. Microbial hitchhikers on marine plastic debris: Human exposure risks at bathing waters and beach environments. Mar Environ Res. 118:10-19.

Kettner MT, Rojas-Jimenez K, Oberbeckmann S, Labrenz M, Grossart H. 2017. Microplastics alter composition of fungal communities in aquatic ecosystems. Environ Microbiol. 19(11):4447-4459.

Kettner MT. 2018. Microbial colonization of microplastic particles in aquatic systems [dissertation]. Potsdam (DE): University of Potsdam.

Kienitz AT. 2013. Marine Debris in the Coastal Environment of Iceland's Nature Reserve, Hornstrandir - Sources, Consequences and Prevention Measures [Master's Thesis]. Ísafjörður (IS): University of Akureyri. [cited in PAME 2019].

Kim JS, Lee HJ, Kim SK, Kim HJ. 2018. Global pattern of microplastics (MPs) in commercial food-grade salts: Sea salt as an indicator of seawater MP pollution. Environ Sci Technol. 52(21):12819-12828.

Kim SW, An YJ. 2019. Soil microplastics inhibit the movement of springtail species. Environ Int. 126:699-706.

Kirstein IV, Kirmizi S, Wichels A, Garin-Fernandez A, Erler R, Löder M, Gerdts G. 2016. Dangerous hitchhikers? Evidence for potentially pathogenic *Vibrio* spp. on microplastic particles. Mar Environ Res. 120:1-8.

Klein M, Fischer EK. 2019. Microplastic abundance in atmospheric deposition within the Metropolitan area of Hamburg, Germany. Sci Total Environ. 685:96-103.

Koelmans AA, Besseling E, Shim WJ. 2015. Nanoplastics in the aquatic environment: Critical review. In: Bergmann M, Gutow L, Klages M, editors. Marine Anthropogenic Litter. Cham (CH): Springer. p. 325-340.

Koelmans AA, Bakir A, Burton GA, Janssen CR. 2016. Microplastics as a vector for chemicals in the aquatic environment: Critical review and model-supported reinterpretation of empirical studies. Environ Sci Technol. 50:3315-3326.

Koelmans AA, Mohamed Nor NH, Hermsen E, Kooi M, Mintenig SM, De France J. 2019. Microplastics in freshwaters and drinking water: Critical review and assessment of data quality. Water Res. 155:410-422.

Kole PJ, Löhr AJ, Van Belleghem FGAJ, Ragas AMJ. 2017. Wear and tear of tyres: a stealthy source of microplastics in the environment. Int J Environ Res Public Health. 14(10):1265.

Kooi M, Koelmans AA. 2019. Simplifying microplastic via continuous probability distributions for size, shape, and density. Environ Sci Technol Lett. 6(9):551−557.

Korez S. Gutow L, Saborowski R. 2019. Feeding and digestion of the marine isopod *Idotea emarginata* challenged by poor food quality and microplastics. Comp Biochem Physiol C Toxicol Pharmacol. 226:108586.

Kosuth M, Mason SA, Wattenberg EV. 2018. Anthropogenic contamination of tap water, beer, and sea salt. PLoS One. 13(4):e0194970.

Kratina P, Watts TJ, Green DS, Kordas RL, O'Gorman EJ. 2019. Interactive effects of warming and microplastics on metabolism but not feeding rates of a key freshwater detritivore. Environ Pollut. 255(Pt 2):113259.

Kreider ML, Unice KM, Panko JM. 2019. Human health risk assessment of tire and road wear particles (TRWP) in air. Hum Ecol Risk Assess. In press.

Kühn S, Bravo Rebolledo EL, van Franeker JA. 2015. Deleterious effects of litter on marine life. In: Bergmann M, Gutow L, Klages M, editors. Marine Anthropogenic Litter. Cham (CH): Springer. Chapter 4. p. 75-116.

Kumar VE, Ravikumar G, Jeyasanta KI. 2018. Occurrence of microplastics in fishes from two landing sites in Tuticorin, south east coast of India. Mar Pollut Bull. 135:889-894.

Kutralam-Muniasamy G, Pérez-Guevara F, Elizalde-Martínez I, Shruti VC. 2020. Branded milks - Are they immune from microplastics contamination? Sci Total Environ. 714:136823.

Lachenmeier DW, Kocareva J, Noack D, Kuballa T. 2015. Microplastic identification in German beer - an artefact of laboratory contamination? Dtsch Lebensmitt Rundsch. 111(10):437-440.

Laganà P, Caruso G, Corsi I, Bergami E, Venuti V, Majolino D, La Ferla R, Azzaro M, Cappello S. 2019. Do plastics serve as a possible vector for the spread of antibiotic resistance? First insights from bacteria associated to a polystyrene piece from King George Island (Antarctica). Int J Hyg Environ Health. 222(1):89-100.

Lahens L, Strady E, Kieu-Le TC, Dris R, Boukerma K, Rinnert E, Gasperi J, Tassin B. 2018. Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity. Environ Pollut. 236:661-671.

Lahive E, Walton A, Horton AA, Spurgeon DJ, Svendsen C. 2019. Microplastic particles reduce reproduction in the terrestrial worm *Enchytraeus crypticus* in a soil exposure. Environ Pollut. 255(Pt 2):113174.

Lamb JB, Willis BL, Fiorenza EA, Couch CS, Howard R, Rader DN, True JD, Kelly LA, Ahmad A, Jompa J, et al. 2018. Plastic waste associated with disease on coral reefs. Science. 359(6374):460-462.

Lambert S, Wagner M. 2017. Environmental performance of bio-based biodegradable plastics: the road ahead. Chem Soc Rev. 46(22):6855-6871.

Lares M, Ncibi MC, Sillanpaa Ma, Sillanpaa Mi. 2018. Occurrence, identification and removal of microplastic particles and fibers in conventional activated sludge process and advanced MBR technology. Water Res. 133:236-246.

Laskin S, Drew RT, Cappiello VP, Kuschner M. 1972. Inhalation studies with freshly generated polyurethane foam dust. In: Mercer TT, Morrow PE, Stober W, editors. Assessment of Airborne Particles. Springfield (IL): Charles C. Thomas. p. 382-404.

Lavers JL, Bond AL. 2016. Ingested plastic as a route for trace metals in Laysan Albatross (*Phoebastria immutabilis*) and Bonin Petrel (*Pterodroma hypoleuca*) from Midway Atoll. Mar Pollut Bull. 110:493-500.

Lavers JL, Hutton I, Bond AL. 2019. Clinical pathology of plastic ingestion in marine birds and relationships with blood chemistry. Environ Sci Technol. 53(15):9224-9231.

Lebreton L, Slat B, Ferrari F, Sainte-Rose B, Aitken J, Marthouse R, Hajbane S, Cunsolo S, Schwarz A, Levivier A, et al. 2018. Evidence that the Great Pacific Garbage Patch is rapidly accumulating plastic. Sci Rep. 8(1):4666.

Lee H, Kunz A, Shim WJ, Walther BA. 2019. Microplastic contamination of table salts from Taiwan, including a global review. Sci Rep. 9(1):10145.

Lee KW, Shim WJ, Kwon OY, Kang JH. 2013. Size-dependent effects of micro polystyrene particles in the marine copepod *Tigriopus japonicus*. Environ Sci Technol. 47(19):11278-11283.

Lei L, Liu M, Song Y, Lu S, Hu J, Cao C, Xie B, Shi H, He D. 2018b. Polystyrene (nano)microplastics cause size-dependent neurotoxicity, oxidative damage and other adverse effects in *Caenorhabditis elegans*. Environ Sci Nano. 5:2009-2020.

Lei L, Wu S, Lu S, Liu M, Song Y, Fu Z, Shi H, Raley-Susman KM, He D. 2018a. Microplastic particles cause intestinal damage and other adverse effects in zebrafish *Danio rerio* and nematode *Caenorhabditis elegans*. Sci Total Environ. 619-620:1-8.

Leonas KK, Gorden RW. 1993. An accelerated laboratory study evaluating the disintegration rates of plastic films in simulated aquatic environments. J Environ Polym Degrad. 1:45-51.

Letcher RJ, Bustnes JO, Dietz R, Jenssen BM, Jørgensen EH, Sonne C, Verreault J, Vijayan MM, Gabrielsen GW. 2010. Exposure and effects assessment of persistent organohalogen contaminants in arctic wildlife and fish. Sci Total Environ. 408(15):2995-3043.

Leung J, Chan KYK. 2018. Microplastics reduced posterior segment regeneration rate of the polychaete *Perinereis aibuhitensis*. Mar Pollut Bull. 129(2):782-786.

Li B, Ding Y, Cheng X, Sheng D, Xu Z, Rong Q, Wu Y, Zhao H, Ji X, Zhang Y. 2020. Polyethylene microplastics affect the distribution of gut microbiota and inflammation development in mice. Chemosphere. 244:125492.

Li J, Yang D, Li L, Jabeen K, Shi H. 2015. Microplastics in commercial bivalves from China. Environ Pollut. 207:190-195.

Li J, Green C, Reynolds A, Shi H, Rotchell JM. 2018a. Microplastics in mussels sampled from coastal waters and supermarkets in the United Kingdom. Environ Pollut. 241:35-44.

Li J, Liu H, Chen JP. 2018b. Microplastics in freshwater systems: A review on occurrence, environmental effects, and methods for microplastics detection. Water Res. 137:362-374.

Li N, Georas S, Alexis N, Fritz P, Xia T, Williams MA, Horner E, Nel A. 2016. A work group report on ultrafine particles (American Academy of Allergy, Asthma & Immunology): why ambient ultrafine and engineered nanoparticles should receive special attention for possible health outcomes in human subjects. J Allergy Clin Immunol. 138(2):386-396.

Li Q, Feng Z, Zhang T, Ma C, Shi H. 2020. Microplastics in the commercial seaweed nori. J Hazard Mater. 388:122060.

Li WC, Tse HF, Fok L. 2016. Plastic waste in the marine environment: A review of sources, occurrence and effects. Sci Total Environ. 566-567:333-349.

Li Y, Li M, Li Z, Yang L, Liu X. 2019. Effects of particle size and solution chemistry on triclosan sorption on polystyrene microplastic. Chemosphere. 231:308-314.

Li Y, Shao L, Wang W, Zhang M, Feng X, Li W, Zhang D. 2020. Airborne fiber particles: Types, size and concentration observed in Beijing. Sci Total Environ. 705:135967.

Liao Y-L, Yang J-Y. 2020. Microplastic serves as a potential vector for Cr in an in-vitro human digestive model. Sci Total Environ. 703:134805.

Liboiron F, Ammendolia J, Saturno J, Melvin J, Zahara A, Richard N, Liboiron M. 2018. A zero percent plastic ingestion rate by silver hake (*Merluccius bilinearis*) from the south coast of Newfoundland, Canada. Mar Pollut Bull. 131:267-275.

Liboiron M, Melvin J, Richard N, Saturno J, Ammendolia J, Liboiron F, Charron L, Mather C. 2019. Low incidence of plastic ingestion among three fish species significant for human consumption on the island of Newfoundland, Canada. Mar Pollut Bull. 141:244-248.

Liebezeit G, Liebezeit E. 2013. Non-pollen particulates in honey and sugar. Food Addit Contam Part A Chem Anal Control Expo Risk Assess. 30(12):2136-2140.

Liebezeit G, Liebezeit E. 2014. Synthetic particles as contaminants in German beers. Food Addit Contam Part A Chem Anal Control Expo Risk Assess. 31(9):1574-1578.

Liebezeit G, Liebezeit E. 2015. Origin of Synthetic Particles in Honeys. Pol J Food Nutr Sci. 65(2):143-147.

Lithner D, Larsson A, Dave G. 2011. Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. Sci Total Environ. 409:3309-3324.

Liu C, Li J, Zhang Y, Wang L, Deng J, Gao Y, Yu L, Zhang J, Sun H. 2019. Widespread distribution of PET and PC microplastics in dust in urban China and their estimated human exposure. Environ Int. 128:116-124.

Liu K, Wang X, Fang T, Xu P, Zhu L, Li D. 2019a. Source and potential risk assessment of suspended atmospheric microplastics in Shanghai. Sci Total Environ. 675:462-471.

Liu K, Wang X, Wei N, Song Z, Li D. 2019b. Accurate quantification and transport estimation of suspended atmospheric microplastics in megacities: Implications for human health. Environ Int. 132:105127.

Liu K, Wu T, Wang X, Song Z, Zong C, Wei N, Li D. 2019c. Consistent transport of terrestrial microplastics to the ocean through atmosphere. Environ Sci Technol. 53(18):10612-10619

Liu M, Lu S, Song Y, Lei L, Hu J, Lv W, Zhou W, Cao C, Shi H, Yang X, He D. 2018. Microplastic and mesoplastics pollution in farmland soils in suburbs of Shanghai, China. Environ Pollut. 242:855-862.

Liu S, Gunawan C, Barraud N, Rice SA, Harry EJ, Amal R. 2016. Understanding, monitoring, and controlling biofilm growth in drinking water distribution systems. Environ Sci Technol. 50(17):8954-8976.

Liu X, Xu J, Zhao Y, Shi H, Huang C-H. 2019. Hydrophobic sorption behaviors of 17β-Estradiol on environmental microplastics. Chemosphere. 226:726-735.

Liu Z, Yu P, Cai M, Wu D, Zhang M, Chen M, Zhao Y. 2019. Effects of microplastics on the innate immunity and intestinal microflora of juvenile *Eriocheir sinensis*. Sci Total Environ. 685:836-846.

Lo HKA, Chan KYK. 2018. Negative effects of microplastic exposure on growth and development of *Crepidula onyx*. Environ Pollut. 233:588-595.

Long M, Moriceau B, Gallinari M, Lambert C, Huvet A, Raffray J, Soudant P. 2015. Interactions between microplastics and phytoplankton aggregates: Impact on their respective fates. Mar Chem. 175:39-46.

Lu L, Wan Z, Luo T, Fu Z, Jin Y. 2018. Polystyrene microplastics induce gut microbiota dysbiosis and hepatic lipid metabolism disorder in mice. Sci Total Environ. 631-632:449-458.

Lu Y, Zhang Y, Deng Y, Jiang W, Zhao Y, Geng J, Ding L, Ren H. 2016. Uptake and accumulation of polystyrene microplastics in zebrafish (*Danio rerio*) and toxic effects in liver. Environ Sci Technol. 50(7):4054-4060.

Luo T, Wang C, Pan Z, Jin C, Fu Z, Jin Y. 2019a. Maternal polystyrene microplastic exposure during gestation and lactation altered metabolic homeostasis in the dams and their F1 and F2 offspring. Environ Sci Technol. 53(18):10978-10992.

Luo T, Zhang Y, Wang C, Wang X, Zhou J, Shen M, Zhao Y, Fu Z, Jin Y. 2019b. Maternal exposure to different sizes of polystyrene microplastics during gestation causes metabolic disorders in their offspring. Environ Pollut. 255(Pt 1):113122.

Luo H, Zhao Y, Li Y, Xiang Y, He D, Pan X. 2020. Aging of microplastics affects their surface properties, thermal decomposition, additives leaching and interactions in simulated fluids. Sci Total Environ. 714:136862.

Lusher AL, McHugh M, Thompson RC. 2013. Occurrence of microplastics in the gastrointestinal tract of pelagic and demersal fish from the English Channel. Mar Pollut Bull. 67(1-2):94-99.

Lusher AL, Hernandez-Milian G, O'Brien J, Berrow S, O'Connor I, Officer R. 2015b. Microplastic and macroplastic ingestion by a deep diving, oceanic cetacean: The True's beaked whale *Mesoplodon mirus*. Environ Pollut. 199:185-191.

Lusher AL, Tirelli V, O'Connor I, Officer R. 2015a. Microplastics in Arctic polar waters: the first reported values of particles in surface and sub-surface samples. Sci Rep. 5:14947.

Lusher AL, Welden NA, Sobral P, Cole M. 2017. Sampling, isolating and identifying microplastics ingested by fish and invertebrates. Anal Methods. 9:1346-1360.

Ma B, Xue W, Ding Y, Hu C, Liu H, Qu J. 2018. Removal characteristics of microplastics by Fe-based coagulants during drinking water treatment. J Environ Sci. 78:267-275.

Ma J, Zhao J, Zhu Z, Li L, Yu F. 2019. Effect of microplastic size on the adsorption behavior and mechanism of triclosan on polyvinyl chloride. Environ Pollut. 254(Pt B):113104.

Ma-Hock L, Landsiedel R, Wiench K, Geiger D, Strauss V, Gröters S, van Ravenzwaay B, Gerst M, Wohlleben W, Scherer G. 2012. Short-term rat inhalation study with aerosols of acrylic ester-based polymer dispersions containing a fraction of nanoparticles. Int J Toxicol. 31(1):46-57.

Maaß S, Daphi D, Lehmann A, Rillig MC. 2017. Transport of microplastics by two collembolan species. Environ Pollut. 225:456-459.

Macali A, Semenov A, Venuti V, Crupi V, D'Amico F, Rossi B, Corsi I, Bergami E. 2018. Episodic records of jellyfish ingestion of plastic items reveal a novel pathway for trophic transference of marine litter. Sci Rep. 8(1):6105.

Maes T, Jessop R, Wellner N, Haupt K, Mayes AG. 2017. A rapid-screening approach to detect and quantify microplastics based on fluorescent tagging with Nile Red. Sci Rep. 7:44501.

Magara G, Elia AC, Syberg K, Khan FR. 2018. Single contaminant and combined exposures of polyethylene microplastics and fluoranthene: accumulation and oxidative stress response in the blue mussel, *Mytilus edulis*. J Toxicol Environ Health A. 81(16):761-773.

Magnusson K, Norén F. 2014. Screening of microplastic particles in and down-stream a wastewater treatment plant [PDF]. Stockholm (SE): IVL Swedish Environmental Research Institute. [accessed 2019 May 29].

Magnusson K, Eliasson K, Fråne A, Haikonen K, Hultén J, Olshammar M, Stadmark J, Voisin A. 2016. Swedish sources and pathways for microplastics to the marine environment. A review of existing data. IVL - Swedish Environmental Research Institute.

Mahler GJ, Esch MB, Tako E, Southard TL, Archer SD, Glahn RP, Shuler ML. 2012. Oral exposure to polystyrene nanoparticles affects iron absorption. Nat Nanotechnol. 7(4):264-271.

Mahon AM, O'Connell B, Healy MG, O'Connor I, Officer R, Nas R, Morrison L. 2017. Microplastics in Sewage Sludge: Effects of Treatment. Environ Sci Technol. 51(2):810-818.

Malafaia G, de Souza AM, Pereira AC, Gonçalves S, da Costa Araújo AP, Ribeiro RX, Rocha TL. 2020. Developmental toxicity in zebrafish exposed to polyethylene microplastics under static and semi-static aquatic systems. Sci Total Environ. 700:134867.

Mani T, Hauk A, Walter U, Burkhardt-Holm P. 2015. Microplastics profile along the Rhine River. Sci Rep. 5:17988).

Mao Y, Ai H, Chen Y, Zhang Z, Zeng P, Kang L, Li W, Gu W, He Q, Li H. 2018. Phytoplankton response to polystyrene microplastics: Perspective from an entire growth period. Chemosphere. 208:59-68.

Martins A, Guilhermino L. 2018. Transgenerational effects and recovery of microplastics exposure in model populations of the freshwater cladoceran *Daphnia magna* Straus. Sci Total Environ. 631-632:421-428.

Mason SA, Kammin L, Eriksen M, Aleid G, Wilson S, Box C, Williamson N, Riley A. 2016. Pelagic plastic pollution within surface waters of Lake Michigan, USA. J Great Lakes Res. 42:753-759.

Mason SA, Welch VG, Neratko J. 2018. Synthetic Polymer Contamination in Bottled Water. Front Chem. 6:407.

Mason SA, Daily J, Aleid G, Ricotta R, Smith M, Donnelly K, Knauff R, Edwards W, Hoffman MJ. 2020. High levels of pelagic plastic pollution within the surface waters of Lakes Erie and Ontario. J Great Lakes Res. 46(2):277-288.

Mateos-Cárdenas A, Scott DT, Seitmaganbetova G, van Pelt FNAM, O'Halloran J, Jansen MAK. 2019. Polyethylene microplastics adhere to *Lemna minor* (L.), yet have no effects on plant growth or feeding by *Gammarus duebeni* (Lillj.). Sci Total Environ. 689:413-421.

Mathalon A, Hill P. 2014. Microplastic fibers in the intertidal ecosystem surrounding Halifax Harbor, Nova Scotia. Mar Pollut Bull. 81(1):69-79.

Mato Y, Isobe T, Takada H, Kanehiro H, Ohtake C, Kaminuma T. 2001. Plastic resin pellets as a transport medium for toxic chemicals in the marine environment. Environ Sci Technol. 35:318-324.

Mazurais D, Ernande B, Quazuguel P, Severe A, Huelvan C, Madec L, Mouchel O, Soudant P, Robbens J, Huvet A, et al. 2015. Evaluation of the impact of polyethylene microbeads ingestion in European sea bass (*Dicentrarchus labrax*) larvae. Mar Environ Res. 112:78-85.

McCormick AR, Hoellein TJ, Mason SA, Schluep J, Kelly JJ. 2014. Microplastic is an abundant and distinct microbial habitat in an urban river. Environ Sci Technol. 48(20):11863-11871.

McCormick AR, Hoellein TJ, London MG, Hittie J, Scott JW, Kelly JJ. 2016. Microplastic in surface waters of urban rivers: concentration, sources, and associated bacterial assemblages. Ecosphere. 7(11):e01556.

McWilliams M, Liboiron M, Wiersma Y. 2018. Rocky shoreline protocols miss microplastics in marine debris surveys (Fogo Island, Newfoundland and Labrador). Mar Pollut Bull. 129:480-486.

Menezes R, da Cunha-Neto MA, de Mesquita GC, da Silva GB. 2019. Ingestion of macroplastic debris by the common dolphinfish (*Coryphaena hippurus*) in the Western Equatorial Atlantic. Mar Pollut Bull. 141:161-163.

Menicagli V, Balestri E, Lardicci C. 2019a. Exposure of coastal dune vegetation to plastic bag leachates: A neglected impact of plastic litter. Sci Total Environ. 683:737-748.

Menicagli V, Balestri E, Vallerini F, Castelli A, Lardicci C. 2019b. Adverse effects of non-biodegradable and compostable plastic bags on the establishment of coastal dune vegetation: First experimental evidences. Environ Pollut. 252(Pt 2):188-195.

Merski JA, Johnson WD, Muzzio M, Lyang NL, Gaworski CL. 2008. Oral toxicity and bacterial mutagenicity studies with a spunbond polyethylene and polyethylene terephthalate polymer fabric. Int J Toxicol. 27(5):387-395.

Mikoczy Z, Welinder H, Tinnerberg H, Hagmar L. 2004. Cancer incidence and mortality of isocyanate exposed workers from the Swedish polyurethane foam industry: updated findings 1959–98. Occup Environ Med. 61(5):432-437.

Mintenig SM, Loder MGJ, Primpke S, Gerdts G. 2019. Low numbers of microplastics detected in drinking water from ground water sources. Sci Total Environ. 648:631-635.

Mohamed Nor NH, Koelmans AA. 2019. Transfer of PCBs from microplastic under simulated gut fluid conditions is biphasic and reversible. Environ Sci Technol. 53(4):1874-1883.

Mohsen M, Zhang L, Sun L, Lin C, Wang Q, Yang H. 2020. Microplastic fibers transfer from the water to the internal fluid of the sea cucumber *Apostichopus japonicus*. Environ Pollut. 257:113606.

Moore RC, Loseto L, Noel M, Etemadifar A, Brewster JD, MacPhee S, Bendell L, Ross PS. 2020a. Microplastics in beluga whales (*Delphinapterus leucas*) from the Eastern Beaufort Sea. Mar Pollut Bull. 150:110723.

Moore RE, Millar BC, Moore JE. 2020b. Antimicrobial resistance (AMR) and marine plastics: Can food packaging litter act as a dispersal mechanism for AMR in oceanic environments? Mar Pollut Bull. 150:110702.

Mouchi V, Chapron L, Peru E, Pruski AM, Meistertzheim A-L, Vétion G, Galand PE, Lartaud F. 2019. Long-term aquaria study suggests species-specific responses of two cold-water corals to macro-and microplastics exposure. Environ Pollut. 253:322-329.

Mühlschlegel P, Hauk A, Walter U, Sieber R. 2017. Lack of evidence for microplastic contamination in honey. Food Addit Contam Part A Chem Anal Control Expo Risk Assess. 34(11):1982-1989.

Munier B, Bendell LI. 2018. Macro and micro plastics sorb and desorb metals and act as a point source of trace metals to coastal ecosystems. PLoS One. 13(2):e0191759.

Muncke J. 2011. Endocrine disrupting chemicals and other substances of concern in food contact materials: an updated review of exposure, effect and risk assessment. J Steroid Biochem Mol Biol. 127(1-2):118-127. [cited in Galloway 2015].

Murphy F, Ewins C, Carbonnier F, Quinn B. 2016. Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment. Environ Sci Technol. 50(11):5800-5808.

Murphy F, Russell M, Ewins C, Quinn B. 2017. The uptake of macroplastic & microplastic by demersal & pelagic fish in the Northeast Atlantic around Scotland. Mar Pollut Bull. 122(1-2):353-359.

Murray F, Cowie PR. 2011. Plastic contamination in the decapod crustacean *Nephrops norvegicus* (Linnaeus, 1758). Mar Pollut Bull. 62(6):1207-1217.

Naik RK, Naik MM, D'Costa PM, Shaikh F. 2019. Microplastics in ballast water as an emerging source and vector for harmful chemicals, antibiotics, metals, bacterial pathogens and HAB species: A potential risk to the marine environment and human health. Mar Pollut Bull. 149:110525.

Naji A, Nuri M, Vethaak AD. 2018. Microplastics contamination in molluscs from the northern part of the Persian Gulf. Environ Pollut. 235:113-120.

Napper IE, Bakir A, Rowland SJ, Thompson RC. 2015. Characterisation, quantity and sorptive properties of microplastics extracted from cosmetics. Mar Pollut Bull. 99:178-85.

Napper IE, Thompson RC. 2019. Environmental deterioration of biodegradable, oxo-biodegradable, compostable, and conventional plastic carrier bags in the sea, soil, and open-air over a 3-year period. Environ Sci Technol. 53(9):4775-4783.

Ng EL, Huerta Lwanga E, Eldridge SM, Johnston P, Hu HW, Geissen V, Chen D. 2018. An overview of microplastic and nanoplastic pollution in agroecosystems. Sci Total Environ. 627:1377-1388.

Nguyen B, Claveau-Mallet D, Hernandez LM, Xu EG, Farner JM, Tufenkji N. 2019. Separation and analysis of microplastics and nanoplastics in complex environmental samples. Acc Chem Res. 52:858-866.

Noble WC, Lidwell OM, Kingston D. 1963. The size distribution of airborne particles carrying micro-organisms. J Hyg. 61(4):385-391. [cited in Prata 2018].

Nizzetto L, Futter M, Langaas S. 2016. Are agricultural soils dumps for microplastics of urban origin? Environ Sci Technol. 50:10777-10779.

Novotna K, Cermakova L, Pivokonska L, Cajthaml T, Pivokonsky M. 2019. Microplastics in drinking water treatment- Current knowledge and research needs. Sci Total Environ. 667:730-740.

Obbard RW, Sadri S, Wong YQ, Khitun AA, Baker I, Thompson RC. 2014. Global warming releases microplastic legacy frozen in Arctic Sea ice. Earth's Future. 2(6):315-320. [cited in PAME 2019].

Oberbeckmann S, Osborn AM, Duhaime MB. 2016. Microbes on a bottle: Substrate, season and geography influence community composition of microbes colonizing marine plastic debris. PLoS One. 11(8):e0159289.

Oberbeckmann S, Kreikemeyer B, Labrenz M. 2018. Environmental factors support the formation of specific bacterial assemblages on microplastics. Front Microbiol. 8:2709.

Oberbeckmann S, Labrenz M. 2020. Marine microbial assemblages on microplastics: Diversity, adaptation, and role in degradation. Ann Rev Mar Sci. 12:209-232.

Oberdörster G, Sharp Z, Atudorei V, Gelein EA, Kreyling W, Cox C. 2004. Translocation of inhaled ultrafine particles to the brain. Inhal Toxicol. 16(6-7):437-445.

Oßmann BE, Sarau G, Holtmannspötter H, Pischetsrieder M, Christiansen SH, Dicke W. 2018. Small-sized microplastics and pigmented particles in bottled mineral water. Water Res. 141:307-316.

Ogonowski M, Motiei A, Ininbergs K, Hell E, Gerdes Z, Udekwu KI, Bacsik Z, Gorokhova E. 2018a. Evidence for selective bacterial community structuring on microplastics. Environ Microbiol. 20(8):2796-2808.

Ogonowski M, Gerdes Z, Gorokhova E. 2018b. What we know and what we think we know about microplastic effects – A critical perspective. Curr Opin Environ Sci Health. 1:41-46.

O'Hara PD, Avery-Gomm S, Wood J, Bowes V, Wilson L, Morgan KH, Boyd WS, Hipfner JM, Desforges JP, Bertram DF, et al. 2019. Seasonal variability in vulnerability for Cassin's auklets (*Ptychoramphus aleuticus*) exposed to microplastic pollution in the Canadian Pacific region. Sci Total Environ. 649:50-60.

Ohtake Y, Kobayashi T, Asabe H, Murakami N. 1995. Biodegradation of low-density polyethylene, polystyrene, polyvinyl chloride, and urea formaldehyde resin buried under soil for over 32 years. J Appl Polym Sci. 56:1789-1796.

Ohtake Y, Kobayashi T, Asabe H, Murakami N, Ono K. 1996. Oxidative degradation and molecular weight change of LDPE buried under bioactive soil for 32-37 years. J Appl Polym Sci. 70:1643-1659.

Özdilek HG, Yalçın-Özdilek Ş, Ozaner FS, Sönmez B. 2006. Impact of accumulated beach litter on *Chelonia mydas* L. 1758 (Green Turtle) hatchlings of the Samandağ coast, Hatay, Turkey. Fresenius Environ Bull. 15(2):95-103.

Pacheco A, Martins A, Guilhermino L. 2018. Toxicological interactions induced by chronic exposure to gold nanoparticles and microplastics mixtures in *Daphnia magna*. Sci Total Environ. 628-629:474-483.

Page B, McKenzie J, McIntosh R, Baylis A, Morrissey A, Calvert N, Haase T, Berris M, Dowie D, Shaughnessy PD, et al. 2004. Entanglement of Australian sea lions and New Zealand fur seals in lost fishing gear and other marine debris before and after Government and industry attempts to reduce the problem. Mar Pollut Bull. 49(1-2):33-42.

[PAME] Protection of the Arctic Marine Environment. 2019. Desktop Study on Marine Litter including Microplastics in the Arctic. Akureyri (IS): PAME. 114 p.

Pan Z, Guo H, Chen H, Wang S, Sun X, Zou Q, Zhang Y, Lin H, Cai S, Huang J. 2019. Microplastics in the Northwestern Pacific: abundance, distribution, and characteristics. Sci Total Environ. 650:1913-1922

Panko JM, Chu J, Kreider ML, Unice KM. 2013. Measurement of airborne concentrations of tire and road wear particles in urban and rural areas of France, Japan, and the United States. Atmos Environ. 72:192-199.

Panko JM, Hitchcock KM, Fuller GW, Green D. 2019. Evaluation of tire wear contribution to PM2.5 in urban environments. Atmosphere. 10(2):99.

Panno SV, Kelly WR, Scott J, Zheng W, McNeish RE, Holm N, Hoellein TJ, Baranski EL. 2019. Microplastic contamination in karst groundwater systems. Groundwater. 57:189-196.

Park E-J, Han J-S, Park E-J, Seong E, Lee G-H, Kim D-W, Son H-Y, Han H-Y, Lee B-S. 2020. Repeated-oral dose toxicity of polyethylene microplastics and the possible implications on reproduction and development of the next generation. Toxicol Lett. 324:75-85.

Patterson J, Jeyasanta KI, Sathish N, Booth Am, Edward JKP. 2019. Profiling microplastics in the Indian edible oyster, Magallana bilineata collected from the Tuticorin coast, Gulf of Mannar, Southeastern India. Sci Total Environ. 691:727-735

Pauly JL, Stegmeier SJ, Allaart HA, Cheney RT, Zhang PJ, Mayer AG, Streck RJ. 1998. Inhaled cellulosic and plastic fibers found in human lung tissue. Cancer Epidemiol Biomarkers Prev. 7(5):419-428.

Pedà C, Caccamo L, Fossi MC, Gai F, Andaloro F, Genovese L, Perdichizzi A, Romeo T, Maricchiolo G. 2016. Intestinal alterations in European sea bass *Dicentrarchus labrax* (Linnaeus, 1758) exposed to microplastics: Preliminary results. Environ Pollut. 212:251-256.

Peeken I, Primpke S, Beyer B, Gutermann J, Katlein C, Krumpen T, Bergmann M, Hehemann L, Gerdts G. 2018. Arctic sea ice is an important temporal sink and means of transport for microplastic. Nat Commun. 9(1):1505.

Peixoto D, Pinheiro C, Amorim J, Oliva-Teles L, Guilhermino L, Vieira MN. 2019. Microplastic pollution in commercial salt for human consumption: A review. Estuar Coast Shelf Sci. 219:161-168.

Pellet Watch. 2019. Global monitoring of POPs using beached plastic resin pellets. Tokyo: Laboratory of Organic Geochemistry, Tokyo Univeristy of Agriculture and Technology. [accessed 2019 June 26]. [cited in WHO 2019].

Peng G, Xu P, Zhu B, Bai M, Li D. 2018. Microplastics in freshwater river sediments in Shanghai, China: case study of risk assessment in mega-cities. Environ Pollut. 234:448-456.

Peng J, Wang J, Cai L. 2017. Current understanding of microplastics in the environment: Occurrence, fate, risks, and what we should do. Integr Environ Assess Manag. 13(3):476-482.

Piarulli S, Scapinello S, Comandini P, Magnusson K, Granberg M, Wong JXW, Sciutto G, Prati S, Mazzeo R, Booth AM, et al. 2019. Microplastic in wild populations of the omnivorous crab *Carcinus aestuarii*: A review and a regional-scale test of extraction methods, including microfibres. Environ Pollut. 251:117-127.

Piehl S, Leibner A, Löder MGJ, Dris R, Bogner C, Laforsch C. 2018. Identification and quantification of macro- and microplastics on an agricultural farmland. Sci Rep. 8(1):17950.

Pieper C, Ventura MA, Martins A, Cunha RT. 2015. Beach debris in the Azores (NE Atlantic): Faial Island as a first case study. Mar Pollut Bull. 101:575-582.

Pierce KE, Harris RJ, Larned LS, Pokras MA. 2004. Obstruction and starvation associated with plastic ingestion in a Northern Gannet *Morus bassanus* and a Greater Shearwater *Puffinus gravis*. Mar Ornithol. 32(2):187-189.

Pigott GH, Ishmael J. 1979. A comparison between *in vitro* toxicity of PVC powders and their tissue reaction *in vivo*. Ann Occup Hyg. 22(2):111-126.

Pikuda O, Genbo Xu, E, Berk D, Tufenkji N. 2019. Toxicity assessments of micro- and nanoplastics can be confounded by preservatives in commercial formulations. Environ Sci Technol Lett. 6(1):21-25.

Pimentel JC, Avila R, Lourenço AG. 1975. Respiratory disease caused by synthetic fibres: a new occupational disease. Thorax. 30(2):204-219.

Pinkerton LE, Yiin JH, Daniels RD, Fent KW. 2016. Mortality among workers exposed to toluene diisocyanate in the US polyurethane foam industry: update and exposure-response analyses. Am J Ind Med. 59(8):630-643.

Pivokonsky M, Cermakova L, Novotna K, Peer P, Cajthaml T, Janda V. 2018. Occurrence of microplastics in raw and treated drinking water. Sci Total Environ. 643:1644-1651.

Plot V, Georges JY. 2010. Plastic debris in a nesting leatherback turtle in French Guiana. Chelonian Conserv Bi. 9(2):267-270.

Poon FE, Provencher JF, Mallory ML, Braune BM, Smith PA. 2017. Levels of ingested debris vary across species in Canadian Arctic seabirds. Mar Pollut Bull. 116(1-2):517-520.

Porter DW, Castranova V, Robinson VA, Hubbs AF, Mercer RR, Scabilloni J, Goldsmith T, Schwegler-Berry D, Battelli L, Washko R, et al. 1999. Acute inflammatory reaction in rats after intratracheal instillation of material collected from a nylon flocking plant. J Toxicol Environ Health Part A. 57(1):24-45.

Poulain M, Mercier MJ, Brach L, Martignac M, Routaboul C, Perez E, Desjean MC, Halle A. 2019. Small microplastics as a main contributor to plastic mass balance in the North Atlantic Subtropical Gyre. Environ Sci Technol. 53(3):1157-1164.

Prata JC. 2018. Airborne microplastics: Consequences to human health? Environ Pollut. 234:115-126.

Prata JC, Reis V, Matos JTV, da Costa JP, Duarte AC, Rocha-Santos T. 2019. A new approach for routine quantification of microplastics using Nile Red and automated software (MP-VAT). Sci Total Environ. 690:1277-1283.

Prest EI, Hammes F, van Loosdrecht MCM, Vrouwenvelder JS. 2016. Biological stability of drinking water: Controlling factors, methods, and challenges. Front Microbiol. 7:45.

Provencher JF, Bond AL, Avery-Gomm S, Borrelle SB, Bravo Rebolledo EL, Hammer S, Kühn S, Lavers JL, Mallory ML, Trevail A, et al. 2017. Quantifying ingested debris in marine megafauna: a review and recommendations for standardization. Anal Methods. 9(9):1454-1469.

Provencher JF, Ammendolia J, Rochman CM, Mallory ML. 2018b. Assessing plastic debris in aquatic food webs: what we know and don't know about uptake and trophic transfer. Environ Rev. p. 1-14.

Provencher JF, Avery-Gomm S, Liboiron M, Braune BM, Macauley JB, Mallory ML, Letcher RJ. 2018c. Are ingested plastics a vector of PCB contamination in northern fulmars from coastal Newfoundland and Labrador? Environ Res. 167:184-190.

Provencher JF, Vermaire JC, Avery-Gomm S, Braune BM, Mallory ML. 2018a. Garbage in guano? Microplastic debris found in faecal precursors of seabirds known to ingest plastics. Sci Total Environ. 644:1477-1484.

Provencher JP, Borrelle SB, Bond AL, Lavers JL, van Franeker JA, Kühn S, Hammer S, Avery-Gomm S, Mallory ML. 2019. Recommended best practices for plastic and litter ingestion studies in marine birds: collection, processing, and reporting. Facets (Ottawa). 4:111-130.

Qi Y, Yang X, Pelaez AM, Huerta Lwanga E, Beriot N, Gertsen H, Garbeva P, Geissen V. 2018. Macro- and micro- plastics in soil-plant system: Effects of plastic mulch film residues on wheat (*Triticum aestivum*) growth. Sci Total Environ. 645:1048-1056.

Qiang L, Cheng J. 2019. Exposure to microplastics decreases swimming competence in larval zebrafish (*Danio rerio*). Ecotoxicol Environ Saf. 176:226-233.

Qiao R, Sheng C, Lu Y, Zhang Y, Ren H, Lemos B. 2019a. Microplastics induce intestinal inflammation, oxidative stress, and disorders of metabolome and microbiome in zebrafish. Sci Total Environ. 662:246-253.

Qiao R, Deng Y, Zhang S, Wolosker MB, Zhu Q, Ren H, Zhang Y. 2019b. Accumulation of different shapes of microplastics initiates intestinal injury and gut microbiota dysbiosis in the gut of zebrafish. Chemosphere. 236:124334.

Rafiee M, Dargahi L, Eslami A, Beirami E, Jahangiri-Rad M, Sabour S, Amereh F. 2018. Neurobehavioral assessment of rats exposed to pristine polystyrene nanoplastics upon oral exposure. Chemosphere. 193:745-753.

Raju S, Carbery M, Kuttykattil A, Senathirajah K, Subashchandrabose SR, Evans G, Thavamani P. 2018. Transport and fate of microplastics in wastewater treatment plants: implications to environmental health. Rev Environ Sci Biotechnol. 17(4):637-653.

Rauert C, Harrad S, Suzuki G, Takigami H, Uchida N, Takata K. 2014. Test chamber and forensic microscopy investigation of the transfer of brominated flame retardants into indoor dust via abrasion of source materials. Sci Total Environ. 493:639-648.

Re V. 2019. Shedding light on the invisible: addressing the potential for groundwater contamination by plastic microfibers. Hydrogeol J. 1-9.

Redondo-Hasselerharm PE, Falahudin D, Peeters ETHM, Koelmans AA. 2018. Microplastic effect thresholds for freshwater benthic macroinvertebrates. Environ Sci Technol. 52(4):2278-2286.

Rehse S, Kloas W, Zarfl C. 2016. Short-term exposure with high concentrations of pristine microplastic particles leads to immobilisation of *Daphnia magna*. Chemosphere. 153:91-99.

Ren X, Tang J, Liu X, Liu Q. 2020. Effects of microplastics on greenhouse gas emissions and the microbial community in fertilized soil. Environ Pollut. 256:113347.

Renzi M, Blašković A. 2018. Litter & microplastics features in table salts from marine origin: Italian versus Croatian brands. Mar Pollut Bull. 135:62-68.

Renzi M, Guerranti C, Blašković A. 2018. Microplastic contents from maricultured and natural mussels. Mar Pollut Bull Part A. 131:248-251.

Renzi M, Čižmek H, Blašković A. 2019a. Marine litter in sediments related to ecological features in impacted sites and marine protected areas (Croatia). Mar Pollut Bull. 138:25-29.

Renzi M, Grazioli E, Blašković A. 2019b. Effects of different microplastic types and surfactant-microplastic mixtures under fasting and feeding conditions: A case study on *Daphnia magna*. Bull Environ Contam Toxicol. 103(3):367-373.

Renzi M, Blašković A, Broccoli A, Bernardi G, Grazioli E, Russo G. 2020. Chemical composition of microplastic in sediments and protected detritivores from different marine habitats (Salina Island). Mar Pollut Bull. 152:110918.

Rezania S, Park J, Md Din MF, Mat Taib S, Talaiekhozani A, Kumar Yadav K, Kamyab H. 2018. Microplastic pollution in different aquatic environments and biota: A review of recent studies. Mar Pollut Bull. 133:191-208.

Richard H, Carpenter EJ, Komada T, Palmer PT, Rochman CM. 2019. Biofilm facilitates metal accumulation onto microplastics in estuarine waters. Sci Total Environ. 683:600-608.

Rillig MC. 2012. Microplastic in terrestrial ecosystems and the soil? Environ Sci Technol. 46(12):6453-6454.

Rillig MC, Ziersch L, Hempel S. 2017. Microplastic transport in soil by earthworms. Sci Rep. 7:1362.

Rist S, Hartmann NB. 2018. Aquatic ecotoxicity of microplastics and nanoplastics: Lessons learned from engineered nanomaterials. In: Wagner M, Lambert S. editors. Freshwater microplastics: emerging environmental contaminants? Cham (CH): Springer. p. 25-49.

Rist S, Baun A, Almeda R, Hartmann NB. 2019. Ingestion and effects of micro- and nanoplastics in blue mussel (*Mytilus edulis*) larvae. Mar Pollut Bull. 140:423-430.

Rochman CM. 2015. The complex mixture, fate and toxicity of chemicals associated with plastic debris in the marine environment. In: Bergmann M, Gutow L, Klages M, editors. Marine anthropogenic litter. Cham (CH): Springer. p. 117-140.

Rochman CM, Hoh E, Kurobe T, Teh SJ. 2013. Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. Sci Rep. 3:3263.

Rochman CM, Kurobe T, Flores I, Teh SJ. 2014. Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment. Sci Total Environ. 493:656-661.

Rochman CM, Tahir A, Williams SL, Baxa DV, Lam R, Miller JT, Teh FC, Werorilangi S, Teh SJ. 2015. Anthropogenic debris in seafood: Plastic debris and fibers from textiles in fish and bivalves sold for human consumption. Sci Rep. 5:14340.

Rochman CM, Browne MA, Underwood AJ, van Franeker JA, Thompson RC, Amaral-Zettler LA. 2016. The ecological impacts of marine debris: unraveling the demonstrated evidence from what is perceived. Ecology. 97(2):302-312.

Rochman CM, Brookson C, Bikker J, Djuric N, Earn A, Bucci K, Athey S, Huntington A, McIlwraith H, Munno K, et al. 2019. Rethinking microplastics as a diverse contaminant suite. Environ Toxicol Chem. 38(4):703-711.

Rosato A, Barone M, Negroni A, Brigidi P, Fava F, Xu P, Candela M, Zanaroli G. 2020. Microbial colonization of different microplastic types and biotransformation of sorbed PCBs by a marine anaerobic bacterial community. Sci Total Environ. 705:135790.

Royer SJ, Ferrón S, Wilson ST, Karl DM. 2018. Production of methane and ethylene from plastic in the environment. PLoS One. 13(8):e0200574.

Rummel CD, Löder MG, Fricke NF, Lang T, Griebeler EM, Janke M, Gerdts G. 2016. Plastic ingestion by pelagic and demersal fish from the North Sea and Baltic Sea. Mar Pollut Bull. 102(1):134-41.

Rummel CD, Jahnke A, Gorokhova E, Kühnel D, Schmitt-Jansen M. 2017. Impacts of biofilm formation on the fate and potential effects of microplastic in the aquatic environment. Environ Sci Technol Lett. 4:258-267.

Ryan PG. 2018. Entanglement of birds in plastics and other synthetic materials. Mar Pollut Bull. 135:159-164.

Ryan PG, Perold V, Osborne A, Moloney CL. 2018. Consistent patterns of debris on South African beaches indicate that industrial pellets and other mesoplastic items mostly derive from local sources. Environ Pollut. 238:1008-1016.

Sadler DE, Brunner FS, Plaistow SJ. 2019. Temperature and clone-dependent effects of microplastics on immunity and life history in *Daphnia magna*. Environ Pollut. 255:113178.

[SAPEA] Science Advice for Policy by European Academies. 2019. A scientific perspective on microplastics in nature and society. Evidence Review Report No. 4. Berlin (DE): SAPEA. 173 p.

Scherer C, Brennholt N, Reifferscheid G, Wagner M. 2017. Feeding type and development drive the ingestion of microplastics by freshwater invertebrates. Sci Rep. 7:17006.

Scherer C, Weber A, Lambert S, Wagner M. 2018. Interactions of microplastics with freshwater biota. In: Wagner M, Lambert S, editors. 2018. Freshwater microplastics: Emerging environmental contaminants? The Handbook of Environmental Chemistry vol 58. Cham (CH): Springer. p. 153-180.

Schettini E, Stefani L, Vox G. 2014. Interaction between agrochemical contaminants and UV stabilizers for greenhouse EVA plastic films. Appl Eng Agric. 30(2):229-239.

Scheurer M, Bigalke M. 2018. Microplastics in Swiss floodplain soils. Environ Sci Technol. 52(6):3591-3598.

Schneider T, Burdett G, Martinon L, Brochard P, Guillemin M, Teichert U, Draeger U. 1996. Ubiquitous fiber exposure in selected sampling sites in Europe. Scand J Work Environ Health. 22(4):274-284.

Schrank I, Trotter B, Dummert J, Schölz-Bottcher BM, Löder MJG, Laforsch C. 2019. Effects of microplastic particles and leaching additive on the life history and morphology of *Daphnia magna*. Environ Pollut. 255:113233.

Schür C, Rist S, Baun A, Mayer P, Hartmann NB, Wagner M. 2019. When fluorescence is not a particle: The tissue translocation of microplastics in *Daphnia magna* seems an artifact. Environ Toxicol Chem. 38(7):1495-1503.

Schuyler Q, Hardesty BD, Wilcox C, Townsend K. 2014. Global analysis of anthropogenic debris ingestion by sea turtles. Conserv Biol. 28(1):129-139.

Schwabl P, Koppel S, Konigshofer P, Bucsics T, Trauner M, Reiberger T, Liebmann B. 2019. Detection of various microplastics in human stool: A prospective case series. Ann Intern Med. 171(7):453-457.

Schymanski D, Goldbeck C, Humpf HU, Fürst P. 2018. Analysis of microplastics in water by micro-Raman spectroscopy: release of plastic particles from different packaging into mineral water. Water Res. 129:154-162.

Seiler C, Berendonk TU. 2012. Heavy metal driven co-selection of antibiotic resistance in soil and water bodies impacted by agriculture and aquaculture. Front Microbiol. 3:399.

Selke S, Auras R, Nguyen TA, Castro Aguirre E, Cheruvathur R, Liu Y. 2015. Evaluation of biodegradation-promoting additives for plastics. Environ Sci Technol. 49(6):3769-3777.

Selonen S, Dolar A, Kokalj AJ, Skalar T, Dolcet LP, Hurley R, van Gestel CAM. 2020. Exploring the impacts of plastics in soil - The effects of polyester textile fibers on soil invertebrates. Sci Total Environ. 700:134451.

Seth CK, Shriwastav A. 2018. Contamination of Indian sea salts with microplastics and a potential prevention strategy. Environ Sci Pollut Res Int. 25(30):30122-30131.

Sieber R, Kawecki D, Nowack B. 2020. Dynamic probabilistic material flow analysis of rubber release from tires into the environment. Environ Pollut. 258:113573.

Silva CJM, Silva ALP, Gravato C, Pestana JLT. 2019. Ingestion of small-sized and irregularly shaped polyethylene microplastics affect *Chironomus riparius* life-history traits. Sci Total Environ. 672:862-868.

Sjollema SB, Redondo-Hasselerharm P, Leslie HA, Kraak MHS, Vethaak AD. 2016. Do plastic particles affect microalgal photosynthesis and growth? Aquat Toxicol. 170:259-261.

Slootmaekers B, Catarci Carteny C, Belpaire C, Saverwyns S, Fremout W, Blust R, Bervoets L. 2019. Microplastic contamination in gudgeons (*Gobio gobio*) from Flemish rivers (BE). Environ Pollut. 244:675-684.

Sobhani Z, Lei Y, Tang Y, Wu L, Zhang X, Naidu R, Megharaj M, Fang C. 2020. Microplastics generated when opening plastic packaging. Sci Rep. 10(1):4841.

Song Y, Cao C, Qiu R, Hu J, Liu M, Lu S, Shi H, Raley-Susman KM, He D. 2019. Uptake and adverse effects of polyethylene terephthalate microplastics fibers on terrestrial snails (*Achatina fulica*) after soil exposure. Environ Pollut. 250:447-455.

Sorahan T, Pope D. 1993. Mortality and cancer morbidity of production workers in the United Kingdom flexible polyurethane foam industry. Br J Ind Med. 50(6):528-536.

Sørensen L, Rogers E, Altin D, Salaberria I, Booth AM. 2020. Sorption of PAHs to microplastic and their bioavailability and toxicity to marine copepods under co-exposure conditions. Environ Pollut. 258:113844.

Sperling LH. 2006. Introduction to physical polymer science. 4th ed. Hoboken (NJ): John Wiley & Sons, Inc. 845 p.

Stalder T, Top E. 2016. Plasmid transfer in biofilms: A perspective on limitations and opportunities. NPJ Biofilms Microbiomes. 2:16022.

Stamper MA, Spicer CW, Neiffer DL, Mathews KS, Fleming GJ. 2009. Morbidity in a juvenile green sea turtle (*Chelonia mydas*) due to ocean-borne plastic. J Zoo Wildl Med. 40(1):196-198.

Stemmer KL, Bingham E, Barkley W. 1975. Pulmonary response to polyurethane dust. Environ Health Perspect. 11:109-113.

Stock V, Böhmert L, Lisicki E, Block R, Cara-Carmona J, Pack LK, Selb R, Lichtenstein D, Voss L, Henderson CJ, et al. 2019. Uptake and effects of orally ingested polystyrene microplastic particles in vitro and in vivo. Arch Toxicol. 93(7):1817-1833.

Stock V, Fahrenson C, Thuenemann A, Dönmez MH, Voss L, Böhmert L, Braeuning A, Lampen A, Sieg H. 2020. Impact of artificial digestion on the sizes and shapes of microplastic particles. Food Chem Toxicol. 135:111010.

Strand J, Feld L, Murphy F, Mackevica A, Hartmann NB. 2018. Analysis of Microplastic Particles in Danish Drinking Water. Aarhus (DK): Aarhus University, Danish Centre for Environment and Energy (DCE). 34 p. Report No. 291.

Su L, Cai H, Kolandhasamy P, Wu C, Rochman CM, Shi H. 2018. Using the Asian clam as an indicator of microplastic pollution in freshwater ecosystems. Environ Pollut. 234:347-355.

Sun J, Dai X, Wang Q, van Loosdrecht MCM, Ni BJ. 2019. Microplastics in wastewater treatment plants: Detection, occurrence and removal. Water Res. 152:21-37.

Sundet JH, Herzke D, Jenssen M. 2016. Forekomst og kilder i mikroplastikk i sediment, og konsekvenser for bunnlevende fisk og evertebrater på Svalbard. In: Miljøvernfond S, editor. Sluttrapport. Sysselmannen, Norway, p. 1-13. [cited in Hallanger and Gabrielsen 2018].

Sundt P, Shulze PE, Syversen F. 2014. Sources of microplastic pollution to the marine environment [PDF]. Trondheim (NO): Norwegian Environment Agency. [accessed 2019 July 5]. Report No. M-321|2015. 86 p.

Suhrhoff TJ, Scholz-Böttcher BM. 2016. Qualitative impact of salinity, UV radiation and turbulence on leaching of organic plastic additives from four common plastics—a lab experiment. Mar Pollut Bull. 102:84-94. [cited in WHO 2019].

Sussarellu R, Suquet M, Thomas Y, Lambert C, Fabioux C, Pernet MEJ, Le Goïc N, Quillien V, Mingant C, Epelboin Y, et al. 2016. Oyster reproduction is affected by exposure to polystyrene microplastics. Proc Natl Acad Sci U S A. 113(9):2430-2435.

Talvitie J, Mikola A, Koistinen A, Setälä O. 2017. Solutions to microplastic pollution – Removal of microplastics from wastewater effluent with advanced wastewater treatment technologies. Water Res. 123:401-407.

Tanaka K, Takada H, Yamashita R, Mizukawa K, Fukuwaka M, Watanuki Y. 2013. Accumulation of plastic-derived chemicals in tissues of seabirds ingesting marine plastics. Mar Pollut Bull. 69:219-222.

Tang BL. 2017. Commentary: Tissue accumulation of microplastics in mice and biomarker responses suggest widespread health risks of exposure. Front Environ Sci. 5:63.

Tang J, Wang X, Yin J, Han Y, Yang J, Lu X, Xie T, Akbar S, Lyu K, Yang Z. 2019. Molecular characterization of thioredoxin reductase in waterflea *Daphnia magna* and its expression regulation by polystyrene microplastics. Aquat Toxicol. 208:90-97.

Thyssen J, Kimmerle G, Dickhaus S, Emminger E, Mohr U. 1978. Inhalation studies with polyurethane foam dust in relation to respiratory tract carcinogenesis. J Environ Pathol Toxicol. 1(4):501-508.

Toussaint B, Raffael B, Angers-Loustau A, Gilliland D, Kestens V, Petrillo M, Rio-Echevarria IM, Van den Eede G. 2019. Review of micro- and nanoplastic contamination in the food chain. Food Addit Contam Part A Chem Anal Control Expo Risk Assess. 36(5):639-673.

Trestrail C, Walpitagama M, Hedges C, Truskewycz A, Miranda A, Wlodkowic D, Shimeta J, Nugegoda D. 2020. Foaming at the mouth: Ingestion of floral foam microplastics by aquatic animals. Sci Total Environ. 705:135826.

Tunahan Kaya A, Yurtsever M, Çiftçi Bayraktar S. 2018. Ubiquitous exposure to microfiber pollution in the air. 2018. Eur Phys J Plus. 133:488.

Uhl W, Eftekhardadkhah M, Svendsen C. 2018. Mapping microplastic in Norwegian drinking water. Hamar (NO): Norwegian Water. p. 43. Report No. 241/2018.

Unice KM, Weeber MP, Abramson MM, Reid RCD, van Gils JAG, Markus AA, Vethaak AD, Panko JM. 2019a. Characterizing export of land-based microplastics to the estuary — Part I: Application of integrated geospatial microplastic transport models to assess tire and road wear particles in the Seine watershed. Sci Total Environ. 646:1639-1649.

Unice KM, Weeber MP, Abramson MM, Reid RCD, van Gils JAG, Markus AA, Vethaak AD, Panko JM. 2019b. Characterizing export of land-based microplastics to the estuary – Part II: Sensitivity analysis of an integrated geospatial microplastic transport modeling assessment of tire and road wear particles. Sci Total Environ. 646:1650-1659.

[UNEP] United Nations Environment Programme. 2015. Biodegradable plastics and marine litter: Misconceptions, concerns, and impacts on marine environments. Nairobi (KY): United Nations. 38 p. [accessed 2019 July 5].

[UNEP] United Nations Environment Programme. 2016. Marine plastic debris and microplastics: Global lessons and research to inspire action and guide policy change. Nairobi (KY): United Nations. 252 p.

[US EPA] US Environmental Protection Agency. 2002. Health risks from microbial growth and biofilms in drinking water distribution systems [PDF]. Washington (DC): US EPA. [accessed 2019 July].

[US EPA] US Environmental Protection Agency. 2009. Integrated science assessment (ISA) for particulate matter (Final Report, Dec 2009). Washington (DC): US EPA. Report No. EPA/600/R-08/139F.

Van Cauwenberghe L, Janssen CR. 2014. Microplastics in bivalves cultured for human consumption. Environ Pollut. 193:65-70.

Van Cauwenberghe L, Claessens M, Vandegehuchte MB, Janssen CR. 2015. Microplastics are taken up by mussels (*Mytilus edulis*) and lugworms (*Arenicola marina*) living in natural habitats. Environ Pollut. 199:10-17.

Vassilenko K, Watkins M, Chastain S, Posacka A, Ross PS. 2019. Me, my clothes and the ocean: The role of textiles in microfiber pollution. Science Feature. Vancouver (BC): Ocean Wise. 16 p.

Verschoor AJ. 2015. Towards a definition of microplastics: Considerations for the specification of physico-chemical properties. RIVM Letter report. Bilthoven (NL): National Institute for Public Health and the Environment. 38 p.

Vert M, Doi Y, Hellwich KH, Hess M, Hodge P, Kubisa P, Rinaudo M, Schué F. 2012. Terminology for biorelated polymers and applications (IUPAC recommendations 2012). Pure Appl Chem. 84(2):377-410.

Vidyasakar A, Neelavannan K, Krishnakumar S, Prabaharan G, Sathiyabama Alias Priyanka T, Magesh NS, Godson PS, Srinivasalu S. 2018. Macrodebris and microplastic distribution in the beaches of Rameswaram Coral Island, Gulf of Mannar, Southeast coast of India: A first report. Mar Pollut Bull. 137:610-616.

Vianello A, Jensen RL, Liu L, Vollertsen J. 2019. Simulating human exposure to indoor airborne microplastics using a breathing thermal manikin. Sci Rep. 9:8670.

Votier SC, Archibald K, Morgan G, Morgan L. 2011. The use of plastic debris as nesting material by a colonial seabird and associated entanglement mortality. Mar Pollut Bull. 62(1):168-172.

Volkheimer G. 1975. Hematogenous dissemination of ingested polyvinyl chloride particles. Ann N Y Acad Sci. 246:164-171.

Von Wintersdorff CJH, Penders J, van Niekerk JM, Mills ND, Majumder S, van Alphen LB, Savelkoul PHM, Wolffs PFG. 2016. Dissemination of antimicrobial resistance in microbial ecosystems through horizontal gene transfer. Front Microbiol. 7:173.

Walkinshaw C, Lindeque PK, Thompson R, Tolhurst T, Cole M. 2020. Microplastics and seafood: lower trophic organisms at highest risk of contamination. Ecotoxicol Environ Saf. 190:110066.

Wang W, Ndungu AW, Li Z, Wang J. 2017. Microplastics pollution in inland freshwaters of China: A case study in urban surface waters of Wuhan, China. Sci Total Environ. 575:1369-1374. [cited in WHO 2019].

Wang W, Yuan W, Chen Y, Wang J. 2018. Microplastics in surface waters of Dongting Lake and Hong Lake, China. Sci Total Environ. 633:539-545.

Wang X, Li C, Liu K, Zhu L, Song Z, Li D. 2020. Atmospheric microplastic over the South China Sea and East Indian Ocean: abundance, distribution and source. J Hazard Mater. 389:121846.

Wang Z, Lin T, Chen W. 2020. Occurrence and removal of microplastics in an advanced drinking water treatment plant (ADWTP). Sci Total Environ. 700:134520.

Warheit DB, Webb TR, Reed KL, Hansen JF, Kennedy GL. 2003. Four-week inhalation toxicity study in rats with nylon respirable fibers: rapid lung clearance. Toxicology. 192(2-3):189-210.

Watts AJR, Urbina MA, Corr S, Lewis C, Galloway TS. 2015. Ingestion of plastic microfibers by the crab *Carcinus maenas* and its effect on food consumption and energy balance. Environ Sci Technol. 49(24):14597-14604.

Weber A, Scherer C, Brennholt N, Reifferscheid G, Wagner M. 2018. PET microplastics do not negatively affect the survival, development, metabolism and feeding activity of the freshwater invertebrate *Gammarus pulex*. Environ Pollut. 234:181-189.

Welden NA, Abylkhani B, Howarth LM. 2018. The effects of trophic transfer and environmental factors on microplastic uptake by plaice, *Pleuronectes plastessa*, and spider crab, *Maja squinado.* Environ Pollut. 239:351-358.

Welden NAC, Cowie PR. 2016. Long-term microplastic retention causes reduced body condition in the langoustine, *Nephrops norvegicus*. Environ Pollut. 218:895-900.

Weinstein JE, Crocker BK, Gray AD. 2016. From macroplastic to microplastic: Degradation of high-density polyethylene, polypropylene, and polystyrene in a salt marsh habitat. Environ Toxicol Chem. 35(7):1632-1640.

Werner S, Budziak A, van Franeker J, Galgani F, Hanke G, Maes T, Matiddi M, Nilsson P, Ooesterbaan L, Priestland E, et al. 2016. Harm caused by marine litter. Joint Research Centre [JRC] Technical Report. Luxembourg: European Union. 92 p.

Wesch C, Barthel AK, Braun U, Klein R, Paulus M. 2016. No microplastics in benthic eelpout (*Zoarces viviparus*): An urgent need for spectroscopic analyses in microplastic detection. Environ Res. 148:36-38.

[WHO] World Health Organization. 2019. Microplastics in drinking-water. Geneva (CH): WHO. 101 p.

Wick P, Malek A, Manser P, Meili D, Maeder-Althaus X, Diener L, Diener PA, Zisch A, Krug HF, von Mandach U. 2010. Barrier capacity of human placenta for nanosized materials. Environ Health Perspect. 118(3):432-436.

Wieczorek AM, Croot PL, Lombard F, Sheahan JN, Doyle TK. 2019. Microplastic ingestion by gelatinous zooplankton may lower efficiency of the biological pump. Environ Sci Technol. 53:5387-5395.

Wiesheu AC, Anger PM, Baumann T, Niessner R, Ivleva NP. 2016. Raman microspectroscopic analysis of fibers in beverages. Anal Methods. 8:5722-5725.

Windsor FM, Durance I, Horton AA, Thompson RC, Tyler CR, Ormerod SJ. 2019. A catchment-scale perspective of plastic pollution. Glob Chang Biol. 25(4):1207-1221.

Winkler A, Santo N, Ortenzi MA, Bolzoni E, Bacchetta R, Tremolada P. 2019. Does mechanical stress cause microplastic release from plastic water bottles? Water Res. 166:115082.

Wójcik-Fudalewska D, Normant-Saremba M, Anastácio P. 2016. Occurrence of plastic debris in the stomach of the invasive crab Eriocheir sinensis. Mar Pollut Bull. 113(1-2):306-311.

Woodall LC, Sanchez-Vidal A, Canals M, Paterson GLJ, Coppock R, Sleight V, Calafat A, Rogers AD, Narayanaswamy BE, Thompson RC. 2014. The deep sea is a major sink for microplastic debris. R Soc Open Sci. 1(4):140317.

Wright MS, Loeffler-Peltier GL, Stepanauskas R, McArthur JV. 2006. Bacterial tolerances to metals and antibiotics in metal-contaminated and reference streams. FEMS Microbiol Ecol. 58(2):293-302.

Wright SL, Rowe D, Thompson RC, Galloway TS. 2013. Microplastic ingestion decreases energy reserves in marine worms. Curr Biol. 23(23):R1031-R1033.

Wright SL, Kelly FJ. 2017. Plastic and human health: A micro issue? Environ Sci Technol. 51(12):6634-6647.

[WRF] Water Research Foundation. 2017. Distribution system management: Understanding and controlling biofilms [PDF]. Denver (CO): WRF. [updated 2017 July].

Wu F, Wang Y, Leung JYS, Huang W, Zeng J, Tang Y, Chen J, Shi A, Yu X, Xu X, et al. 2020. Accumulation of microplastics in typical commercial aquatic species: A case study at a productive aquaculture site in China. Sci Total Environ. 708:135432.

Wu N, Zhang Y, Zhao Z, He J, Li W, Li J, Xu W, Ma Y, Niu Z. 2020. Colonization characteristics of bacterial communities on microplastics compared with ambient environments (water and sediment) in Haihe Estuary. Sci Total Environ. 708:134876.

Wu X, Pan J, Li M, Li Y, Bartlam M, Wang Y. 2019. Selective enrichment of bacterial pathogens by microplastic biofilm. Wat Res. 165:114979.

Xiong X, Tu Y, Chen X, Jiang X, Shi H, Wu C, Elser JJ. 2019. Ingestion and egestion of polyethylene microplastics by goldfish (*Carassius auratus*): Influence of color and morphological features. Heliyon. 5(12):e03063.

Xu H, Verbeken E, Vanhooren HM, Nemery B, Hoet PHM. 2004. Pulmonary toxicity of polyvinyl chloride particles after a single intratracheal instillation in rats. Time course and comparison with silica. Toxicol Appl Pharmacol. 194(2):111-121.

Yadav MK. 2017. Role of biofilms in environment pollution and control. In: Patra J, Vishnuprasad C, Das G, editors. Microbial biotechnology. Singapore (SG): Springer. p. 377-398.

Yang D, Shi H, Li L, Li J, Jabeen K, Kolandhasamy P. 2015. Microplastic Pollution in Table Salts from China. Environ Sci Technol. 49(22):13622-13627.

Yoo JW, Doshi N, Mitragotri S. 2011. Adaptive micro and nanoparticles: temporal control over carrier properties to facilitate drug delivery. Adv Drug Deliv Rev. 63(14-15):1247-1256.

Yin L, Chen B, Xia B, Shi X, Qu K. 2018. Polystyrene microplastics alter the behavior, energy reserve and nutritional composition of marine jacopever (*Sebastes schlegelii*). J Hazard Mater. 360:97-105.

Yu P, Liu Z, Wu D, Chen M, Lv W, Zhao Y. 2018. Accumulation of polystyrene microplastics in juvenile *Eriocheir sinensis* and oxidative stress effects in the liver. Aquat Toxicol. 200:28-36.

Yu F, Yang C, Zhu Z, Bai X, Ma J. 2019. Adsorption behavior of organic pollutants and metals on micro/nanoplastics in the aquatic environment. Sci Total Environ. 694:133643.

Zalasiewicz J, Waters CN, Ivar do Sul JA, Corcoran PL, Barnosky AD, Cearreta A, Edgeworth M, Galuszka A, Jeandel C, Leinfelder R, et al. 2016. The geological cycle of plastics and their use as a stratigraphic indicator of the Anthropocene. Anthropocene. 13:4-17.

Zbyszewski M, Corcoran PL. 2011. Distribution and degradation of fresh water plastic particles along the beaches of Lake Huron, Canada. Water Air Soil Pollut. 220(1-4):365-372.

Zbyszewski M, Corcoran PL, Hockin A. 2014. Comparison of the distribution and degradation of plastic debris along shorelines of the Great Lakes, North America. J Great Lakes Res. 40:288-299.

Zeri C, Adamopoulou A, Bojanić Varezić D, Fortibuoni T, Kovač Viršek M, Kržan A, Mandic M, Mazziotti C, Palatinus A, Peterlin M, et al. 2018. Floating plastics in Adriatic waters (Mediterranean Sea): From the macro- to the micro-scale. Mar Pollut Bull. 136:341-350.

Zettler ER, Mincer TJ, Amaral-Zettler LA. 2013. Life in the "plastisphere": Microbial communities on plastic marine debris. Environ Sci Technol. 47(13):7137-7146.

Ziajahromi S, Kumar A, Neale PA, Leusch FDL. 2018. Environmentally relevant concentrations of polyethylene microplastics negatively impact the survival, growth and emergence of sediment-dwelling invertebrates. Environ Pollut. 236:425-431.

Zhang C, Jeong C-B, Lee J-S, Wang D, Wang M. 2019. Transgeneration proteome plasticity in resilience of a marine copepod in response to environmentally relevant concentrations of microplastics. Environ Sci Technol. 53(14):8426-8436.

Zhang GS, Liu YF. 2018. The distribution of microplastics in soil aggregate fractions in southwestern China. Sci Total Environ. 642:12-20.

Zhang J, Gao D, Li Q, Zhao Y, Li L, Lin H, Bi Q, Zhao Y. 2020a. Biodegradation of polyethylene microplastic particles by the fungus *Aspergillus flavus* from the guts of wax moth *Galleria mellonella*. Sci Total Environ. 704:135931.

Zhang J, Wanga L, Kannan K. 2020b. Microplastics in house dust from 12 countries and associated human exposure. Environ Int. 134:105314.

Zhang Y, Gu AZ, Cen T, Li X, He M, Li D, Chen J. 2018. Sub-inhibitory concentrations of heavy metals facilitate the horizontal transfer of plasmid-mediated antibiotic resistance genes in water environment. Environ Pollut. 237:74-82.

Zhang Y, Lu J, Wu J, Wang J, Luo Y. 2020. Potential risks of microplastics combined with superbugs: enrichment of antibiotic resistant bacteria on the surface of microplastics in mariculture system. Ecotoxicol Environ Saf. 187:109852.

Zhang Z, Chen Y. 2020. Effects of microplastics on wastewater and sewage sludge treatment and their removal: A review. Chem Eng J. 382:122955.

Zhou Q, Zhang H, Fu C, Zhou Y, Dai Z, Li Y, Tu C, Luo Y. 2018. The distribution and morphology of microplastics in coastal soils adjacent to the Bohai Sea and Yellow Sea. Geoderma. 322:201-208.

Zubris KAV, Richards BK. 2005. Synthetic fibres as an indicator of land application of sludge. Environ Pollut. 138:201-211.

Zuccarello P, Ferrante M, Cristaldi A, Copat C, Grasso A. Sangregorio D, Fiore M, Oliveri Conti G. 2019. Exposure to microplastics (<10 μm) associated to plastic bottles mineral water consumption: The first quantitative study. Water Res. 157:365-371.

Zumstein MT, Narayan R, Kohler HE, McNeil K, Sander M. 2019. Dos and do nots when assessing the biodegradation of plastics. Environ Sci Technol. 53(17):9967-9969.

Appendix A: New information published or received through public consultation

The following is a summary of relevant new information published since the literature review cut-off date for the Draft Science Assessment of Plastic Pollution (i.e., from June 2019 to March 2020), as well as relevant new information received through public consultation.

Table A-1: Sources of plastic waste and pollution

Relevant Section	Summary of New Information	Source
3.1 Sources to water	Laundering is a source of microplastics to the environment. Unlike studies by Carney Almroth et al. (2018) and De Falco et al. (2018), Cesa et al. (2020) found that the use of detergent significantly reduced the release of fibres from synthetic garments, but not from cotton. They also found that cotton released the highest amount of fibres compared to acrylic, PE and PA. The reason for this discrepancy is not clear; however, it likely arises from the difference in methods used in each study. Examples of these differences include differences in temperature, concentration of detergents, and filtration size.	Cesa et al. 2020
3.1 Sources to water	The results of these studies confirmed the occurrence of microplastics in the air above aquatic environments. The modelled results found that atmospheric microplastics could be contributing to marine pollution.	Liu K et al. 2019b; Wang X et al. 2020
3.1.1 Wastewater treatment	This review found that microplastics affect the performance of wastewater treatment. It found that microplastics could cause blockages, alter microbial-mediated processes, and wear down equipment thus affecting performance.	Zhang and Chen 2020
3.3 Sources to air	The study authors determined the number of fibres released to air from volunteers carrying out everyday activities while wearing different types of textiles. They concluded that the release of microfibres to air from the wearing of PE clothing is of the same order of magnitude as that of microfibres released to wastewater by laundering.	De Falco et al. 2020

Table A-2: Environmental fate

Relevant Section	Summary of New Information	Source
4.1 Degradation	This study looked at the ability of potential plastic degrading bacteria to form biofilms and confirmed that bacterial community composition is dependent on plastic type. The study also found that the bacteria *Alcanivorax borkumensis* can form thick biofilms on LDPE. The authors noted that further research into degradation mechanisms is required.	Delacuvellerie et al. 2019
4.1 Degradation	This paper studied the weathering of LDPE and PP films in water in order to identify their fragmentation mechanism and understand the pathways leading from macroplastics to microplastics. They found that crack initiation and propagation depended on the crystalline morphology of the polymer.	Julienne et al. 2019
4.1 Degradation	This study looked at the ability of a fungus to degrade PE microplastic and found that HDPE was degraded into microplastic particles with lower molecular weights after 28 days of incubation.	Zhang J et al. 2020a

Table A-3: Occurrence

Relevant Section	Summary of New Information	Source
5.1.1 Occurrence in the aquatic environment Surface water	This study found plastics in all surface water samples collected from Lake Erie and Lake Ontario. Plastic particles were separated into three size classifications: 0.355 to 0.999 mm, 1.00 to 4.749 mm, and ≥4.75 mm. The majority of the particles in both lakes were in the small size classification. In Lake Erie, pellets and fibres made up the majority of the samples, and in Lake Ontario, fragments dominated. In the largest size classification, 46% of the polymers were identified as PE, and 43% were PP. It should be noted that the smallest size classification was analyzed by SEM/EDS and the large classification was analyzed by FTIR, but the middle size classification was not characterized.	Mason et al. 2020

Relevant Section	Summary of New Information	Source
5.1.3 Occurrence in air Indoor air	The study authors quantitatively determined PET- and PC-based microplastics (size <150 µm) in 286 indoor dust samples collected from 12 countries in North and South America, Europe, and Asia. PET-based microplastics were detected in all dust samples at concentrations of 0.03 to 110 µg/mg, whereas PC-based microplastics were detected in 99% of samples at <0.0001 to 1.7 µg/mg. These concentrations are similar to those reported in China by Liu C et al. (2019).	Zhang J et al. 2020b
5.1.3 Occurrence in air Outdoor air	Various types of fibres were measured in outdoor air in Beijing, China. Microplastic fibres were measured at mean concentrations of 5 600 to 5 700 fibres/m^3 at two sampling site heights (1.5 m and 18 m above ground respectively). Microplastic fibres represented 35% and 41% of total fibres at the two sample heights. Other types of fibres measured included natural organic fibres, man-made mineral fibres, asbestos, calcium sulfate fibres and metal fibres. The microplastic concentrations reported in this study represent higher levels than reported by Dris et al. (2017) and Liu K et al. (2019a). Differences may be attributable to differences in sampling heights, proximity to city centre, etc.	Li Y et al. 2020
5.1.3 Occurrence in air Outdoor air	Microplastics were measured on a roof in Shanghai (38 m above ground) at a mean concentration of 0.06 microplastics/m^3 (range: 0.05 to 0.07 microplastics/m^3). Fibres and fragments were the predominant shapes, representing 43% and 48% of sampled microplastics, respectively. Sizes varied between 12 and 2 191 µm, with an average of 247 µm, with higher concentrations of microplastics generally observed for smaller size fractions. This study reports lower concentrations than were reported by Dris et al. (2017) and Liu K et al. (2019a). Differences may be attributable to differences in sampling heights, proximity to city centre, etc.	Liu K et al. 2019b
5.1.3 Occurrence in air Outdoor air	These studies measured airborne microplastics over the western Pacific Ocean.	Liu K et al. 2019c; Wang X et al. 2020

Relevant Section	Summary of New Information	Source
	Liu K et al. (2019c) reported a median concentration of 0.01 microplastics/m^3 (range: 0 to 1.37 microplastics/m^3). Higher concentrations of microplastics were observed in coastal areas versus pelagic areas, and shape composition was also less diverse in pelagic regions. Wang X et al. (2020) measured the abundance of microplastics in the ambient air over the ocean at three sites: Pearl River Estuary (PRE), South China Sea and East Indian Ocean (EIO). Concentrations ranged from 0.0042 microplastics/m^3 in PRE to 0.0004 microplastics/m^3 in EIO (average: 0.001 microplastics/m^3).	
5.2.1 Occurrence in food	The mean microplastic concentrations in the GI tract of shrimp collected from the Mediterranean Sea (*Aristeus antennatus*) and the Northern Bay of Bengal, Bangladesh (*Metapenaeus monocerous; Penaeus monodon*) ranged from 1.66 ± 0.11 to 3.87 ± 1.05 microplastics per shrimp and consisted of fibres, filaments and fragments. The mean microplastic concentration in the flesh of shrimp collected from an aquaculture site at Xiangshan Bay (*Parapenaeopsis harwickii*) was 0.95 microplastics per shrimp and consisted of fibres. A previous study reported microplastics in the digestive tract, head, and gills of shrimp (*Crangon crangon*) collected from the Clyde Sea (mean: 1.23 microplastics per shrimp), but not in the abdominal muscle tissue (Devriese et al. 2015). The mean microplastic concentrations in prawn muscle (*Penaeus semisulcatus*) collected from the northeast of the Persian Gulf was 0.360 items/g of muscle and consisted of mostly fibres and fragments. A previous study (Abbasi et al. 2018) reported microplastic concentrations in prawns as items per individuals and	Akhbarizadeh et al. 2019; Cau et al. 2019; Hossain et al. 2020; Wu F et al. 2020

Relevant Section	Summary of New Information	Source
	therefore, the previous results could not be compared to these results.	
5.2.1 Occurrence in food	These are the first studies to report microplastics in the GI tract of crabs (*Carcinus aestuarii*) sampled from the northern Adriatic coast of Italy and crab muscle (*Portunus armatus*) sampled from the northeast of the Persian Gulf. The mean microplastic concentration in the GI tract was 1.1 ± 0.7 microplastics per crab and consisted of mostly fibres (100 to 5 000 µm), whereas the mean microplastic concentration in crab muscle was 0.256 items/g of muscle and consisted of mostly fibres (100 to >5 000 µm) and fragments (<50 to 500 µm).	Akhbarizadeh et al. 2019; Piarulli et al. 2019
5.2.1 Occurrence in food	The mean microplastic concentration on fish skin was 6.40 ± 0.65 items/individual, ranging from 4.23 to 9.30 items/individual and consisted of mostly fibres <1 000 µm. Microplastic concentrations on fish skin were generally higher in species of scaleless fish with mucus compared to scaly fish.	Feng et al. 2019
5.2.1 Occurrence in food	This is the first study to report microplastics in fleur de sel (a type of unprocessed, flaky, moist salt harvested from the sea surface). Higher concentrations of microplastics (<1000 µm; shape not reported) were reported in fleur de sel (520 µg/kg salt) compared to sea salt (37 µg/kg salt).	Fischer et al. 2019
5.2.1 Occurrence in food	Microplastics were detected in commercial brands of fish meal sourced from Malaysia and Southern Iran. Based on a single laboratory experiment, a positive relationship was observed between microplastic levels in Iranian fish meal and the gills and GI tract of cultured common carp (*Cyprinus carpio*), suggesting that fish meal may be a potential source of microplastics in farmed aquatic species.	Hanachi et al. 2019; Karbalaei et al. 2020
5.2.1 Occurrence in food	This was the first study to investigate whether plastic teabags released particles	Hernandez et al. 2019

Relevant Section	Summary of New Information	Source
	under conditions that mimicked the tea steeping process. While the plastic source was known in this experimental study, the novel analytical method used for identifying and quantifying particles in the tea did not individually confirm the particles to be plastic. The plastic teabags were also cut open and the tea leaves were removed prior to analysis and therefore, it cannot be ruled out that the cutting of the plastic teabags led to the formation of particles.	
5.2.1 Occurrence in food	This is the first study to investigate the presence of microplastics in packaged poultry products. The concentration of microplastics per kg of meat reported in this study represented a combined estimate of microplastics on the surface of the meat and inside of the packaging after the meat was removed, with microplastics suspended in the air of the food production facilities identified as the possible contamination source of the meat/packaging. Procedural blanks were not completed in this study and therefore, background contamination of the samples in the laboratory cannot be excluded.	Kedzierski et al. 2020
5.2.1 Occurrence in food	This is the first study to report microplastics in milk (purchased from stores in Mexico). Microplastic concentrations ranged from 3 to 11 microplastics per litre of milk (varying across brand and type of milk), with mainly fibres detected. Approximately 40% of microplastics were <500 µm, 28% were between 500 and 100 µm, and 25% were between 1 000 and 2 000 µm.	Kutralam-Muniasamy et al. 2020
5.2.1 Occurrence in food	This is the first study to report microplastics in noris, which were collected from local markets, factories, and farms in China. Microplastic concentrations ranged from 0.9 to 3.0 microplastics per gram of nori (dw),	Li Q et al. 2020

Relevant Section	Summary of New Information	Source
	with mainly fibres detected (range: 70 to 47 40 µm; mean: 850 µm).	
5.2.1 Occurrence in food	The authors report that tearing open plastic packaging and opening plastic bottle caps can generate approximately 10 to 30 ng of microplastics per 300 cm of plastic packaging.	Sobhani et al. 2020
5.2.1 Occurrence in food	A review of 32 studies of commercially important aquatic species suggested that microplastics do not biomagnify in the aquatic food chain. Higher concentrations of microplastics were reported in organisms at lower trophic levels, such as shellfish, compared to apex predators, such as predatory fish.	Walkinshaw et al. 2020
5.2.1 Occurrence in food Bottled water	This study investigated the effects of mechanical stress on the generation of microplastics (≥3 µm) in single-use PET plastic water bottles. Opening the bottles once resulted in no obvious deformities, abrasions, or particle release on the caps or bottlenecks, whereas opening/closing the bottles 10 times resulted in minor abrasions and deformities on caps (not bottlenecks) and a few loose particles on caps and outside of bottlenecks and opening/closing the bottles 100 times (outside the normal conditions of use) resulted in signs of mechanical damage and detached particles on caps and outside of bottlenecks. The particles most likely originated from the bottle caps. Squeezing/crushing the bottles under a weight (5 kg for 10 minutes) had no effect on microplastic concentrations in bottled water and there was no evidence of breaks or abrasions on the surface of the bottles' inner wall.	Winkler et al. 2019
5.2.2 Occurrence in drinking water	This study measured microplastics above 25 µm in water treated by eight drinking water treatment plants (DWTP) across the United Kingdom. In tap water, the average microplastic concentration was 0.00011 particles/L; however, microplastics were	Ball et al. 2019

Relevant Section	Summary of New Information	Source
	often not quantifiable with only two sites having detections above the limit of quantification. Acrylonitrile butadiene styrene (ABS) and PS were measured at concentrations of 0.0020 and 0.0008 particles/L, respectively, in a treated water sample derived from a groundwater site, whereas PS was measured at a concentration of 0.0016 particles/L in a treated water sample derived from a pumped storage site receiving water from a major river. These low concentrations are similar to data reported by Mintenig et al. (2019), who assessed microplastics as low as 20 µm in treated groundwater, but much lower than those reported by Pivokonsky et al. (2018), who measured microplastics as small as 1 µm in tap water sourced from surface water. Ball et al. (2019) also reported that PS and ABS were the most abundant polymers in drinking water samples, whereas PET and PP were the most predominant polymer types identified in drinking water by the WHO (2019).	
5.2.2 Occurrence in drinking water	In drinking water sourced from the Yangtze River in China, the average microplastic concentration was determined to be 930 ± 71 particles/L. These levels are higher than those from a previous study in treated surface water (Pivokonsky et al. 2018) and several orders of magnitude higher than studies in tap water from groundwater sources (Strand et al. 2018; Mintenig et al. 2019). Nearly all microplastics were identified as being <10 µm in size, with most between 1 to 5 µm. PET was identified as the most abundant polymer type, followed by PE and PP, then PA. Fibres were the most prevalent shape, followed by fragments then spheres.	Wang Z et al. 2020
5.2.3 Drinking water treatment	This study reported a microplastic removal rate greater than 99.99% for particles >25 µm by eight DWTPs in the United Kingdom using conventional treatment processes. The polymers identified in the treated water (ABS and PS) differed from the ones identified in the raw water source (PE, PET, and PP),	Ball et al. 2019

Relevant Section	Summary of New Information	Source
	suggesting they may have been generated during the treatment; however, the authors recognize that limitations in microplastic detection and quantification preclude any definite conclusions.	
5.2.3 Drinking water treatment	Microplastic removal efficiency was evaluated at each treatment step of a DWTP in the Yangtze River Delta in China using conventional treatment followed by ozonation combined with granular activated carbon (GAC) filtration. The overall microplastic removal efficiency for the DWTP was 82.1% to 88.6%, which is slightly higher than the data presented by Pivokonsky et al. (2018), who reported removal rates of 70% to 82%. Between 58.9% and 70.5% of the microplastic removal was attributable to conventional treatment (coagulation/sedimentation/sand filtration). Coagulation combined with sedimentation provided the highest removal efficiency, with 40.5% to 54.5% removal (with a preference for microplastics >10 µm and fibres), followed by sand filtration with 29.0 to 44.4% removal. The advanced processes of ozonation combined with GAC filtration increased the removal by 17.2% to 22.2%.	Wang Z et al. 2020

Table A-4: Impacts on environmental health

Relevant Section	Summary of New Information	Source
6.1 Macroplastic	Balestri et al. (2019) investigated the effects of leachates from HDPE bags and compostable Mater-bi (MB) bags on seed germination. The study exposed *Lepidum sativum* L. to leachates from both virgin and weathered bags in the natural environment. Both bags were determined to alter water quality (i.e., pH, salinity, total dissolved solids, and phytotoxic substances). Seed germination of *L. sativum* was not affected; however, 2% to 40% of the seeds exposed to plastic leachates had developmental abnormalities. Additionally, reduced growth	Balestri et al. 2019

Relevant Section	Summary of New Information	Source
	was seen in seeds exposed to leachates of both HPDE and MB bags, though leachates of virgin bags impacted a greater number of seedlings.	
6.1 Macroplastic	A laboratory study was conducted on coastal dune vegetation using HDPE bags and Mater-bi (MB) compostable bags. The study authors found that leachates affected germination and seedling growth. When exposed to low concentrations of HDPE and MB bag leachates, *Thinopyrum junceum* and *Glaucium flavum* both had increased seed germinability. At high concentrations of MB bag leachate, *G. flavum* germinated earlier, while *T. junceum* germinated later, compared to controls. A second study determined that macroplastic fragments of HDPE and MB bags incorporated into beach sand can affect the sexual recruitment and growth of *T. junceum, Ammophila grenaria,* and *G. flavum*. When exposed to HDPE bags, *T. junceum* had a reduced number of roots, reduced root length, reduced seedling height and reduced above ground biomass. *A. grenaria* seedlings exposed to MB bags had reduced seedling height, a reduced number of roots and reduced root length. Very few emerged *A. grenaria* seedlings survived after HDPE exposure. All *G. flavum* seedlings died, regardless of the treatment applied.	Menicagli et al. 2019a; 2019b
6.1.1 Entanglement	This review compiled reported pinniped entanglements from 1980 to 2018. A total of 69 articles contained information related to pinniped entanglement, and most of the articles were from North America and Oceania with a focus on populous species. All articles claimed abandoned, lost or discarded fishing gear and packaging strapping as responsible for the majority of entanglements.	Jepsen and de Bruyn 2019
6.1.1 Entanglement	This study investigated the effect of plastic debris on the susceptibility of reef-building corals to disease. When corals were not in contact with plastic debris, the likelihood of disease was 4.4%. This likelihood increased to	Lamb et al. 2018

Relevant Section	Summary of New Information	Source
	89.1% in the presence of plastic debris. Plastic debris can cause physical injury and abrasion to corals, facilitating invasion by pathogens, or exhausting resources for immune system function. Additionally, plastic debris can directly introduce resident and foreign pathogens, such as cross-ocean bacterial colonization of PVC dominated by *Rhodobacterales*, a group of pathogens associated with outbreaks of several coral diseases.	
6.1.2 Ingestion	This study observed the diet-related selectivity of macroplastics in green turtles (*Chelonia mydas*) in the eastern Mediterranean. The turtles favoured sheet and threadlike forms, and black, clear and green colours. There was a negative correlation between turtle size and body burden of plastic, which suggests naivety in young turtles. All 19 green turtles contained plastics in their GI tracts with an ingested average of 61.8 items per turtle.	Duncan et al. 2019
6.1.2 Ingestion	This study determined the effects of micro- and macroplastics on two cold-water coral species*, Lophelia pertusa* and *Madrepora oculate*, over an exposure period of five months. LDPE microplastics were found to impair prey capture and growth rates of *L. pertusa*. Due to avoidance behaviour, macroplastic films had little effect on the growth of *L. pertusa*. However, *M. oculate* were not affected by macro- or microplastic exposure. This provides evidence of a species-specific response.	Mouchi et al. 2019
6.2.1 Uptake, ingestion and egestion	This study examined the presence of plastics in seal stomachs from the eastern Canadian Arctic. Stomachs were collected from 135 ringed seals (*Phoca hispida*), 6 bearded seals (*Erignathus barbatus*), and 1 harbour seal (*Phoca vitualina*) from Nunavut and examined for plastics >425 µm. No evidence of plastic accumulation was observed in the stomachs. It should be noted that plastic identification was not confirmed using an analytical method, and that microfibres were not quantified.	Bourdages et al. 2020

Relevant Section	Summary of New Information	Source
6.2.1 Uptake, ingestion and egestion	Earthworm ingestion of microplastics was studied by Chen Y et al. (2020). The study exposed *Eisenia fetida* to microplastics sized 100 to 200 µm and found that microplastics were ingested in a dose-dependent manner. More egested microplastics were recorded on day 28 compared with day 14, suggesting that microplastics may be accumulated in the bodies of the earthworms. Additionally, smaller sizes of LDPE were egested by the earthworms. Similarly, Lahive et al. (2019) also found that the number of PA microplastics ingested was greater in earthworm *Enchytraeus crypticus* exposed to the smaller sized particles (13 to 18 µm) compared to the larger size fractions (63 to 90 µm and 90 to 150 µm).	Chen Y et al. 2020; Lahive et al. 2019
6.2.1 Uptake, ingestion and egestion	This study determined the presence of microplastics and its frequency of ingestion in 39 cod (*Gadus morhua*) and 46 saithe (*Pollachius virens*) caught along the west coast of Iceland. Microplastics were found in 20.5% of the cod with an average of 0.23 microplastics/individual and in 17.4% of the saithe with an average of 0.28 microplastics/individual. It was also determined that microplastic ingestion did not relate to body size, gut fullness, or the general health of the fish. In large individuals, it was found that microplastics were not retained to a large extent and the health of the fish was likely not affected.	de Vries et al. 2020
6.2.1 Uptake, ingestion and egestion	The following additional species have been found to ingest microplastics: • Sea cucumbers (*Holothuria tubulosa*) • *Alepisaurus ferox Lowe* • *Nephrops norvegicus* • *Zostera marina L.* bed • Common cockle (*Cerastoderma edule*) • Grey seals (*Halichoerus gyrpus*)	Gago et al. 2020; Hara et al. 2020; Hermabessiere et al. 2019; Hernandez-Milian et al. 2019; Jones et al. 2020; Renzi et al. 2020
6.2.1 Uptake, ingestion and egestion	This study examined the presence of microplastics in the GI tracts of seven beluga whales (*Delphinapterus leucas*) from the Eastern Beaufort Sea. Microplastics were	Moore et al. 2020a

Relevant Section	Summary of New Information	Source
	found in the GI tract of every whale studied, with each whale containing an estimated 18 to 147 microplastics. Microplastics consisted of both fragments (51%) and fibres (49%). Polyester made up 44% of the polymers identified.	
6.2.1 Uptake, ingestion and egestion	This study investigated the influence of the colour, size, and shape of microplastics on the ingestion and egestion of PE microplastics in goldfish (*Carassius auratus*). Goldfish were exposed to white microplastic fragments of different size categories: 0.5 to 2 mm, 2 to 3 mm, and >3 mm, to investigate the effect of size on ingestion and egestion. Goldfish were also exposed to white, black, blue, red, and green fragments, transparent films, and cyan filaments, all sized 0.5 to 2 mm. In the size experiment, ingestion only occurred in the 0.5 to 2 mm group in the presence of food. Microplastics larger than 2 mm were ingested but were not retained. In the colour experiment, green and black microplastics were ingested at a significantly higher rate than other colours. All shapes ingested by the goldfish could be egested. Fragments were thoroughly egested from all fish after 72 hours, while filament and film particles were not. Greater food availability reduced the quantity of film and filament within the fish after 72-hour exposure.	Xiong et al. 2019
6.2.1 Uptake, ingestion, and egestion 6.2.2. Ecotoxicological effects Sediment	This study investigates the uptake and effects of 1 to 5 mm PS fibres on the sea cucumber *Apostichopus japonicus*. For uptake experiments from sediment, *A. japonicus* were fed 40% algae powder and 60% sea mud mixed with microfibres. The results indicate that microfibres are not transferred to the coelomic fluid upon ingestion of contaminated sediment. For uptake experiments from water, *A. japonicus* were exposed to microplastics in	Mohsen et al. 2020

Relevant Section	Summary of New Information	Source
	different concentrations. The results indicate that microfibres are ingested along with water during respiration, and become stuck in branches of the respiratory tree when expulsion is attempted. The numbers of fibres transferred increased over time. Additionally, fibres were present in the coelomic fluid 72 hours post-exposure. There were no significant effects on the velocity or distance travelled by treated *A. japonicus*. In order to determine the effect on the immune system, the activity of several enzymes were assessed at 24, 48 and 72 hours post-exposure. The activity of lysozyme in treated *A. japonicus* was higher than in controls, and significantly increased levels were observed at 48 hours post-exposure. Myeloperoxidase, acid phosphatase and alkaline phosphatase levels were not affected.	
6.2.2 Ecotoxicological effects Water – Vertebrates	This study investigated the effects of PE microplastics on the livers of *Physalamus cuvieri* tadpoles. Tadpoles were exposed to 35.6 µm particles at a concentration of 60 mg/L for seven days. Microplastic concentration in the liver after the experimental period was 0.215 mg/g of liver tissue. Tadpoles exposed to microplastics presented larger areas with blood vessel dilation, infiltration, congestion, hydropic degeneration, hypertrophy, and hyperplasia. The study notes that effects demonstrated cannot be extrapolated to adult individuals, and that further evaluation of liver function through analysis of liver enzymes and ultrastructural changes in hepatocytes is required.	da Costa Araújo et al. 2020
6.2.2 Ecotoxicological effects Water – Vertebrates	This study investigated the effects of 10 µm PS microplastics on marine medaka (*Oryzias melastigma*). The measured concentration of the suspension was 0.758×10^5 particles/L. Microplastics were ingested by larvae and adults throughout the 48-hour exposure, and	Cong et al. 2019

Relevant Section	Summary of New Information	Source
	adults that were starved beforehand consumed a significantly greater quantity of microplastics. In the post-exposure recovery stage, egestion was rapid within the first day; however, more than 20% of particles were still retained in the non-feeding larvae, and 11.4% within the feeding larvae, after seven days of recovery. There was no significant difference in retention rate. Mortality was significantly higher in fish exposed to PS at the end of the 120-day exposure following exposure to microplastics at early life stage (larvae). Growth and egg production were both significantly reduced.	
6.2.2. Ecotoxicological effects Water – Vertebrates	This study investigated the effects of PVC and PE microplastics on the sea bass *Dicentrarchus labrax* L. The fish were fed diets containing 100 or 500 mg/kg of PE or PVC for three weeks. Microplastics ranged in size from 40 to 150 µm. PE decreased the activity of antioxidant enzymes at both concentrations and increased skin mucus immunoglobulin M levels at 500 mg/kg. PVC caused an increase in the phagocytic burst activities of head kidney leucocytes at both concentrations. Treatments of 100 mg/kg PVC and 500 mg/kg PE increased the respiratory burst of head-kidney leucocytes. In all cases, increased concentration of microplastics magnified the histopathological alterations in sea bass.	Espinosa et al. 2019
6.2.2 Ecotoxicological effects Water – Vertebrates	This study investigated the effects of microplastics on zebrafish (*Danio rerio*) in static and semi-static conditions. *D. rerio* were exposed to PE microplastics (38.26 mm ± 15.64 µm) at concentrations of 6.2, 12.5, 25, 50, and 100 mg/L for 144 hours. Early hatching was observed for embryos under static conditions and survival rate was lower compared to controls for all concentrations.	Malafaia et al. 2020

Relevant Section	Summary of New Information	Source
	Morphological features were affected in *D. rerio* exposed to PE under semi-static conditions. Fish exposed to 12.5 to 100 mg/L had higher head height and larger optic vesicle area than controls. The impacts on fish exposed to 50 and 100 mg/L were longer interocular distance and a wider angle between myosepts.	
6.2.2 Ecotoxicological effects Water – Vertebrates	This study investigated the accumulation and effects of different shape of microplastics (PP fibres, PS fragments and PS beads) on zebrafish (*D. rerio*). Microplastic accumulation was highest for fibres (8.0 µg/mg), followed by fragments (1.7 µg/mg) and microbeads (0.5 µg/mg). Fibres caused increased cavitation within the gut compared to fragments and beads, and led to more serious intestinal epithelial cell necrosis and inflammation. All three forms resulted in the downregulation of differentially expressed genes related to lipid metabolism, hormone metabolism, and protein secretion. Fibres and fragments caused the most severe effects in gut microbiota.	Qiao et al. 2019b
6.2.2. Ecotoxicological effects Water – Invertebrates	This study looked at the effects of PS microplastics on *D. magna* that were exposed to 0, 0.125, 1.25 and 12.5 µg/mL for 21 days. Ingested microplastics remained in the digestive tract after a 96-hour egestion test in clean medium. Exposure at the highest concentration resulted in significant increase in the mean number of offspring.	De Felice et al. 2019
6.2.2. Ecotoxicological effects Water – Invertebrates	This study investigated the effects of PMMA microplastics on ingestion rates and gastrointestinal enzyme activities of marine isopods (*Idotea emarginata*), which were also exposed to natural food. The organisms were not affected by microplastics when receiving sufficient natural food; microplastics in low nutrient food caused alteration of food uptake and digestive enzyme activities.	Korez et al. 2019
6.2.2. Ecotoxicological effects	This study investigated the independent and combined impacts of PMMA microplastics on	Kratina et al. 2019

Relevant Section	Summary of New Information	Source
Water – Invertebrates	metabolic and feeding rates of *Gammarus pulex*. *G. pulex* were exposed to PMMA (40.2 µm) for 24 hours at different concentrations and temperatures (9°C, 15°C and 19°C). Exposure to microplastics altered metabolic rates but not feeding rates, with greater inhibition of metabolic rates at higher temperatures (15°C and 19°C).	
6.2.2. Ecotoxicological effects Water – Invertebrates	This study investigated the effects of PS microplastics on juvenile Chinese mitten crabs (*Eriocheir sinensis*), which were exposed to PS at concentrations of 0, 0.04, 0.4, 4, and 40 mg/L for 7, 14, and 21 days. Low concentrations (0.04 and 0.4 mg/L) of microplastics or short-term (7- or 14-day) exposure promoted immune enzyme activity and immune system gene expression. High concentrations (4 and 40 mg/L) or long-term (21 days) exposure negatively affected the innate immunity of *E. sinensis*.	Liu Z et al. 2019
6.2.2. Ecotoxicological effects Water – Invertebrates	This experiment studied the effects of different microplastics and their mixture with surfactants on the mortality and immobilization of *D. magna*. *D. magna* were exposed to PE, PVC, PP, a mixture of PVC and PE, surfactant alone, and to mixtures of microplastics and surfactants, under fasting and feeding conditions. After a 96-hour exposure, mortality was greater than 30% for all tests (PE, PVC, PP, PVC/PE, surfactants, as well as microplastic and surfactant mixtures). Surfactant was found to increase the mortality and immobilization in all treatments. Exposures to mixtures of PVC and surfactant under feeding conditions caused the highest rate of mortality and immobilization, followed by PE or PP mixed with surfactant, surfactant alone, and PE alone.	Renzi et al. 2019b
6.2.2. Ecotoxicological effects Water – Invertebrates	This study investigated the effects of 500 nm PS spheres in *D. magna* at two temperatures (18°C and 24°C). Groups of *D. magna* were exposed to PS beads at a concentration of 1 mg/L. Microplastic exposure was shown to interfere with *D. magna* immunity, and consequentially their ability to respond to	Sadler et al. 2019

Relevant Section	Summary of New Information	Source
	parasites. Reproductive traits were also affected by microplastics, with smaller offspring, and significantly greater clutch size at 24°C, but not at 18°C.	
6.2.2. Ecotoxicological effects Water – Invertebrates	This study investigated the effects of PS microplastics on the marine copepod *Tigriopus japonicus* under two-generation exposure followed by a generation recovery. Copepods were exposed to PS beads at concentrations of 0.023 and 0.23 mg/L in seawater. Ingestion of PS in F0 and F1 were observed under both exposure concentrations. Copepods exposed to 0.23 mg/L had reduced survival rate, number of nauplii/clutch and fecundity. Traits were restored in F2. In addition, microplastics exposure at the recovery stage (F2) resulted in an increase in cellular biosynthesis processes and reduced energy storage.	Zhang et al. 2019
6.2.2 Ecotoxicological effects Soil	The study exposed the earthworm *Aporrectodea rosea* to HDPE microplastics in soil. *A. rosea* exposed to soil with HDPE lost significantly more biomass than controls.	Boots et al. 2019
6.2.2 Ecotoxicological effects Soil	Several studies investigated the effects of microplastics on terrestrial plants. De Souza Machado et al. (2019) found that microplastics altered biomass and root traits in *Allium fistulosum*. Polyester, PS, HDPE, PET and PE increased root biomass, whereas PA decreased dry biomass. Contrary to De Souza Machado et al. (2019) results for HDPE, Boots et al. (2019) found that HDPE microplastics decreased root biomass in *Lolium perenne*, as well as decreased germination and shoot length.	Boots et al. 2019; De Souza Machado et al. 2019
6.2.2 Ecotoxicological effects Soil	Studies were conducted on soil microbiota. Ren et al. (2020) found that larger microplastics (<150 µm) lowered microbial and fungal community richness and diversity, whereas treatments with smaller microplastic particles (<13 µm) resulted in an increase in these effects. Chen H et al. (2020) studied the effect of polylactic acid microplastics on soil microbiota and found no significant effect on the diversity or composition of bacterial	Chen H et al. 2020, Ren et al. 2020

Relevant Section	Summary of New Information	Source
	communities or related ecosystem functions and processes.	
6.2.2 Ecotoxicological effects Soil	The study investigated defence responses in earthworms exposed to microplastics. *E. fetida* were exposed to LDPE microplastics (100 to 200 µm) in soil at concentrations of 0.1, 0.25, 0.5, 1.0 and 1.5 g/kg dw. Catalase activity and malondialdehyde content significantly increased in the 1.0 g/kg treatment after 28-day exposure. Significantly higher activity of acetylcholine esterase was observed at 1.0 g/kg on day 28, and at 1.5 g/kg on day 21. Additionally, surface damage was observed on earthworms exposed to the highest concentration of LDPE after 28-day exposure.	Chen Y et al. 2020
6.2.2 Ecotoxicological effects Soil	The earthworm *E. crypticus* was exposed to PA microplastics (13 to 18 µm and 90 to 150 µm) for 21 days. Reproduction was significantly reduced for earthworms exposed to 13 to 18 µm and 90 to 150 µm microplastics. The 21-day EC_{50} for microplastics 13 to 18 µm was 108 g/kg. Juvenile production at the highest exposure concentration for 90–150 µm was <50% so an EC_{50} could not be reliably estimated.	Lahive et al. 2019
6.2.2 Ecotoxicological effects Soil	This study exposed soil invertebrates to PE microfibres. Enchytraeids (*E. crypticus*), springtails (*Folsomia candida*), isopods (*Porcellio scaber*) and oribatid mites (*Oppia nitens*) were exposed to 200, 600, 1 700, 5 000, and 15 000 mg fibres/kg dry soil. *E. crypticus* and *F. candida* were also exposed to PE microfibres in spiked food. *E. crypticus* reproduction was slightly reduced at all concentrations in soil, except 600 mg/kg dw. No significant effects were observed in the other invertebrates.	Selonen et al. 2020
6.2.2 Ecotoxicological effects Soil	This study investigated the effects of microfibres on snails, *Achatina fulica*. Soil was treated with PET microfibres at concentrations of 0.014, 0.14 and 0.71 g/kg dw. The average excretion rates were decreased by 29.3%, 46.6% and 69.7%, respectively. A significant decrease in food intake was observed at 0.14 and 0.71 g/kg	Song et al. 2019

Relevant Section	Summary of New Information	Source
	dw. There was also noticeable tissue damage in the stomachs and intestines of snails exposed to microfibres.	
6.2.2. Ecotoxicological effects Sediment	This study investigated the predator-avoidance emergence response of *Littorina littorea* with microplastic ingestion in the field. The study authors recovered 118 microplastics (98% fibres and 2% fragments) in organisms collected from the field, ranging from 0 to 6 microplastics/individual. They found that microplastic exposure did not affect emergence likelihood or emergence latency.	Doyle et al. 2020
6.2.2 Ecotoxicological effects Sediment	This study investigated the effects of microplastics on *Chironomus riparius*. The size ranges of PE particles used were 32 to 63 µm, 63 to 250 µm, and 125 to 500 µm, and *C. riparius* were exposed to concentrations ranging from 1.25 to 20 g/kg of sediment. Larvae preferentially consumed smaller microplastics and consumed more at higher concentrations. Microplastics of all sizes significantly reduced growth after 10-day exposure and a significant delay in the time to emergence was observed, with the greatest effects seen with microplastics 32 to 63 µm. Mortality was not observed.	Silva et al. 2019
6.2.3 Trophic transfer	This report reviewed 421 studies from 1929 to 2019 on the ingestion, bioaccumulation and trophic transfer of microplastics. Though the review supported trophic transfer of microplastics, organisms at higher trophic levels were able to egest the microplastics, which resulted in a lower potential for magnification through the food web. Gouin (2020) concluded that the weight-of-evidence does not support bioaccumulation of microplastics.	Gouin 2020
6.2.3 Trophic transfer	This study investigated the effects of PE microplastics (10 to 45 µm) on *Lemna minor* and the trophic transfer of microplastics from *L. minor* to *Gammarus duebeni*. *L. minor* were exposed to an estimated concentration of 50 000 microplastics/mL and colonies were fed to *G. duebeni* adults.	Mateos-Cárdenas et al. 2019

Relevant Section	Summary of New Information	Source
	In the *L. minor* bioassays, microplastics were found to adhere to exposed *L. minor* colonies, with increasing concentrations over time. No impact was found on growth rate, chlorophyll fluorescence and root length. In the trophic transfer experiment, microplastics were found to readily transfer through the food chain from *L. minor* to *G. duebeni*. However, following 24-hour depuration, only 28.6% of the *G. duebeni* contained 1 to 2 microplastics in their gut. *G. duebeni* mortality was not affected.	

Table A-5: Impacts on human health

Relevant Section	Summary of New Information	Source
7.2.1 Effects from oral exposure Studies in experimental animals	C57BL/6 male mice were continuously exposed to PE microplastics (10 to 150 µm) for five consecutive weeks in feed (0, 2, 20, 200 µg/g feed; equivalent to 0, 6, 60, or 600 µg/mouse/day). The highest dose of microplastics led to an increase in the number of gut microbial species, bacterial abundance, and flora diversity indicative of intestinal dysbacteriosis. Serum levels of the pro-inflammatory cytokine interleukin-1α were also increased in microplastic-treated mice compared to control.	Li B et al. 2020
7.2.1 Effects from oral exposure Studies in experimental animals	This study evaluated the effects of maternal microplastic exposure on dams and offspring (F1 and F2 generation). Pregnant ICR mice were exposed to pristine 5 µm PS microplastics administered through drinking water (0, 100 or 1 000 µg/L) during gestation and lactation (approx. 42 days); however, actual exposure concentrations are unknown, making it difficult to evaluate the significance of the findings.	Luo et al. 2019a

Relevant Section	Summary of New Information	Source
	Maternal exposure to microplastics was associated with impaired gut barrier function and gut microbiota dysbiosis in dams, an increase in relative liver weight in F1 offspring, altered serum and hepatic lipid profiles in F1 offspring (at PND 42 and PND 280) and F2 offspring (at PND 42), and changes in genes related to hepatic energy metabolism in all three generations. The study also reported transcriptomic and metabolomic effects; however, these findings are unclear and require further research.	
7.2.1 Effects from oral exposure Studies in experimental animals	Similar to the experimental exposure conditions described in Luo et al. (2019a), this study evaluated the effects of maternal microplastic exposure (0.5 or 50 µm) on dams and specifically the F1 offspring. Maternal exposure to microplastics had no effect on the sex ratio, survival, body weight, or relative liver weight, but did result in altered amino acid, carnitine, and fatty acid metabolism in the FI offspring. The study also reported transcriptomic and metabolomic effects; however, these findings are unclear and require further research.	Luo et al. 2019b
7.2.1 Effects from oral exposure Studies in experimental animals	In a reproductive/developmental study, male and female ICR mice were continuously exposed to pristine 40 to 48 µm PE microplastics (0, 0.125, 0.5, 2 mg/day; equivalent to 0, 3.75 15, or 60 mg/kg bw) via gavage for 90 days (6 days/week). On days 80 to 89, the male and female mice were mated and euthanized on day 90, except for a subset of female mice that were continuously dosed with microplastics (as described above) during the gestational and lactation period and euthanized on PND 21 along with the F1 offspring. Microplastic exposure led to reduced body weight gain in male F0 mice, an increase in the relative proportion of neutrophils in the blood of male and female F0 mice, and the	Park et al. 2020

Relevant Section	Summary of New Information	Source
	migration of granules to the mast cell membrane, damage/degeneration of the mast cell membrane, the persistence of PE microplastic-like material in the stomach, the accumulation of damaged organelles in the spleen, and an increase in IgA in the blood of female F0 mice. The number of live births per dam and the body weight of pups six hours after birth were significantly reduced and the proportion of T cells in the blood of F1 male and female pups was altered following maternal exposure to the highest dose of microplastics compared to controls.	
7.2.1 Effects from oral exposure Toxicokinetics	Human stool samples from eight participants living in different countries worldwide (one stool sample per person per country) were examined for the presence of microplastics. All stool samples tested positive for microplastics, with PP and PET being the most commonly detected polymers. On average, 20 microplastics (50 to 500 µm) per 10 g of human stool were detected; however, the sample size in this study is too small and too diverse to draw any meaningful conclusions.	Schwabl et al. 2019
7.2.1 Effects from oral exposure Toxicokinetics	The impact of artificial in vitro digestion on the fate of PE, PP, PVC, PET, and PS microplastics (1 to 200 µm) was investigated by measuring the sizes and shapes of the particles before and after different digestion steps (i.e., simulated saliva, gastric, and intestinal fluid). All five polymer types were highly resistant to degradation from the artificial digestive fluids, suggesting that the human GI tract does not degrade microplastic particles.	Stock et al. 2020
7.2.3 Effects of biofilms	Some studies have identified potential human pathogens in microplastic-associated biofilms. For example, Gong et al. (2019) detected the nosocomial pathogen *Chryseobacterium* in LDPE in lake water microcosms. Moore et al. (2020b) isolated	Gong et al. 2019; Moore et al. 2020b; Wu et al. 2019; Curren and Leong 2019; Wu N et al. 2020.

Relevant Section	Summary of New Information	Source
	seven bacterial species capable of causing human infection from food-related marine macroplastic litter around the coast of Northern Ireland. Wu et al. (2019) detected *Pseudomonas monteilii* and *Pseudomonas mendocina* on microplastics isolated from river water, and Curren and Leong (2019) identified *Vibrio* and *Arcobacter* on microplastics from tropical coastal environments in Singapore. Lastly, Wu N et al. (2020) found the abundance of potentially pathogenic bacteria (e.g., *Bacillus*) on microplastics was significantly higher than that in the ambient environment, for microplastics in an estuarine area of China.	
7.2.3 Effects of biofilms	This study showed measurable adsorption for Cs and Sr radiotracers in microplastics from freshwater, estuarine and marine conditions, suggesting that plastics may act as a sink for pervasive environmental radionuclides. However, in most cases, the adsorption rates of all types of plastic biofilm were much lower than those of reference sediments.	Johansen et al. 2019
7.2.3 Effects of biofilms	In bacteria isolated from food-related marine macroplastic litter, antibiotic resistance ranged from 16.1% to 98.1% and included resistance to several classes of critically important antibiotics (e.g., ampicillin, erythromycin).	Moore et al. 2020b
7.2.3 Effects of biofilms	This review suggests that microplastics in ballast waters serve as 'hotspots' for the development and spread of multiple drug-resistant human pathogens through co-selection mechanisms.	Naik et al. 2019
7.2.3 Effects of biofilms	Metal accumulation on microplastics submerged in natural estuarine waters was positively correlated with the amount of associated biofilm, suggesting that biofilm facilitates metal accumulation.	Richard et al. 2019
7.2.3 Effects of biofilms	This study examined colonization dynamics of pristine or PCB-contaminated microplastics, in anaerobic laboratory microcosms of a marine sediment. Microplastic-associated biofilms were able to convert PCBs, via reductive dechlorination, more rapidly than those associated with natural substrates,	Rosato et al. 2020

Relevant Section	Summary of New Information	Source
	making them less toxic but more bioavailable.	
7.2.3 Effects of biofilms	Antimicrobial resistant genes (ARG) were detected in biofilms on microplastics, rocks and leaves in a bioreactor using river water. The ARG subtype with the highest relative abundance in microplastics biofilm was multidrug resistance type, suggesting that microplastics may be selecting for multidrug resistance.	Wu et al. 2019
7.2.3 Effects of biofilms	Some studies have reported higher community diversity on microplastic biofilms compared to diversity on natural substrates (e.g., rocks and leaves) (Wu et al. 2019), and in water samples from a mariculture system (Zhang Y et al. 2020). In contrast, the meta-analysis of Oberbeckmann and Labrenz (2020) concluded that bacterial communities associated with plastics did not differ significantly from those on natural substrates, and moreover, that geographical region influenced bacterial communities more than the surface characteristics of the plastics.	Wu et al. 2019; Zhang Y et al. 2020; Oberbeckmann and Labrenz 2020
7.2.3 Effects of biofilms	This study investigated the enrichment of antibiotic resistance bacteria (ARB) on the surface of microplastics in a mariculture system. The percentage of ARB to total cultivable bacteria in microplastic samples was significantly higher than that in water samples.	Zhang Y et al. 2020

Table A-6: Transport of chemicals

Relevant Section	Summary of New Information	Source
Sorbed chemicals	Several studies found that microplastics are able to transport environmental pollutants, such as antibiotics and endocrine disrupting chemicals (EDCs). Furthermore, Chen Q et al. (2019) found that smaller microplastics sized 0.5 to 1.5 mm leached greater concentrations of EDCs than particles sized 1.5 to 5 mm and 5 to 15 mm. Additionally, the release of EDCs was affected by the environmental conditions. Solar irradiation	Chen Q et al. 2019; Guo and Wang 2019b; Liu X et al. 2019

Relevant Section	Summary of New Information	Source
	was found to increase EDC concentrations in leachates.	
Sorbed chemicals	Metal adsorption to microplastics was found to be enhanced in water with high chemical oxygen demand and biological oxygen demands, such as urban wastewater. Additionally, the accumulation of metals on plastic debris may be affected by biofilms on the plastics.	Godoy et al. 2019; Yu et al. 2019
Sorbed chemicals	This study investigated the effects of PA microplastics on PBDE accumulation in snails (*Lymnaea stagnalis*). Microplastics consisted of PA fragments with a mean size of 13 to 19 µm. No food was provided to the 47 tested snails over the 96-hour experiment. No mortality was reported when exposed to microplastics; however, snails not exposed to microplastics lost significantly more weight than those that were. Increased PBDE concentration in sediment resulted in an increase in PBDE concentration in snails, but the presence of microplastics had no effect on PBDE uptake.	Horton et al. 2020
Sorbed chemicals	Similar to results by Chen Q et al. (2019), studies found that smaller microplastic particles have stronger adsorption capacities than larger particles.	Li et al. 2019; Ma et al. 2019
Sorbed chemicals	In laboratory studies, chromium in water adsorbed to microplastic (PE, PP, PVC, PS, and PLA) particles, with low desorption. PLA adsorbed the least amount of chromium, while PS adsorbed the highest amount of chromium, with all five polymer types reaching saturation within 48 hours. In an in vitro, whole human digestive system model, bioavailability of microplastic-bound hexavalent and trivalent chromium was observed, with increased desorption in synthetic digestive juices (particularly in gastric juices, but also intestinal juices).	Liao and Yang 2020
Sorbed chemicals	This study investigated the bioavailability of PAHs sorbed to microplastics to the marine copepod species *Acartia tonsa* and *Calanus finmarchicus*. In the acute toxicity and bioaccumulation studies that were conducted, it was found that microplastic-	Sørensen et al. 2020

Relevant Section	Summary of New Information	Source
	sorbed PAHs did not significantly accumulate or contribute to toxicity when the copepods were co-exposed with the same chemicals in dissolved phase.	
Sorbed chemicals/Additives	Barboza et al. (2020) reported a significant and positive correlation between the total concentration of bisphenols in the muscle and liver and the total number of microplastics in fish. In other studies (Garcia-Garin et al. 2020; Hermabessiere et al. 2019), no significant correlation was found between microplastic loads and concentrations of plastic additives or hydrophobic organic compounds in bivalves or between microplastic loads in the fish GI tract and organophosphate concentrations in fish muscle.	Barboza et al. 2020; Garcia-Garin et al. 2020; Hermabessiere et al. 2019
Additives	This study investigated the effects of leachate from expanded PS on four microalgal species: *Dunaliella salina, Scenedesmus rubescens, Chlorella saccharophila,* and *Stichococcus bacillaris.* The microalgae were exposed to leachate from three size ranges of expanded PS fragments and spheres for seven days, with one concentration of fragments, and two concentrations of spheres. Hexabromocyclododecanes (HBCD) concentration in the small fragment leachate was found to be significantly higher than the low concentration of large spheres; however, concentrations of BPA and dissolved organic carbon (DOC) were similar. All four species experienced varying degrees of increased photosynthetic activity and increased growth.	Chae et al. 2020
Additives	Leaching tests were conducted to determine the release of red pigment from powdered LDPE microplastics (<500 µm) in an in vitro simulated mammalian digestive system. Microplastic aging led to an increase in the release of pigment, with longer aging times resulting in more pigment release.	Luo et al. 2020

Relevant Section	Summary of New Information	Source
	The release rate of pigment from microplastics was slower in simulated intestinal fluid compared to simulated gastric fluid. Once released into the simulated digestive fluids, the pigments formed complexes with proteins.	
Additives	This study investigated the effects of PVC microplastics that contained the plasticizer diisononylphthalate (DiNP) on *Daphnia magna*. Groups of *D. magna* were exposed to rigid PVC (without DiNP) or flexible PVC (with added DiNP) for 31 days, at a concentration ratio of 1:10 for PVCs and algae. A concentration of 2.67 mg/L DiNP leached out from flexible PVC into the testing solution and led to an increased body length and a reduced number of offspring.	Schrank et al. 2019
Additives	This study investigated the effects of ingestion of foam microplastics on *Physa acuta*, *Bembicium nanum*, the marine bivalve *Mytilus galloprovincialis*, *D. magna*, *Allorchestes compressa*, and nauplii of the marine crustacean *Artemia* sp. Microplastics were generated from two types of foam: "regular foam" was generated from petroleum-based phenol-formaldehyde and "biofoam" was generated from plant-based phenol-formaldehyde. Both types of foams were ingested by all six species tested. While ingestion was similar for both foams, biofoam microplastics leached more than twice as many phenolic compounds than regular foam microplastics. The study also examined the toxicity of microplastic leachates by conducting toxicity tests on *Artemia* nauplii, *D. magna* and *D. rerio*. The leachates from regular foam and biofoam microplastics showed the same acute toxicity to *Artemia* nauplii and *D. magna*. However, biofoam microplastic leachate was twice as toxic to *D. rerio*	Trestrail et al. 2020

Relevant Section	Summary of New Information	Source
	embryos compared to leachate from regular foam microplastics. In order to differentiate between the effects of leachate and the physical effects of microplastics, *M. galloprovincialis* were exposed to four treatments: microplastics only (1 mg microplastic/mL); leachate only (at a concentration equivalent to leachate from 1 mg microplastics/mL); microplastics and leachate; and seawater control. The magnitude of change for effects on catalase activity, glutathione-s-transferase activity and lipid peroxidation were least severe for leachate only, followed by microplastics only, then microplastics and leachate. The authors hypothesize that microplastic leachate and physical presence of microplastics have separate and cumulative effects on organisms.	

Appendix B: Additional occurrence of plastics in the global environment

B-1. Shoreline

Plastic pollution has been detected on shorelines around the world. For example, one study found macroplastics on every beach surveyed on an island in French Polynesia, where plastics accounted for 20% to 100% of all litter items (Connors 2017).

Chen H et al. (2019) collected marine litter around a tourist city in East China and found that plastic pollutants made up the majority of all floating, benthic, and beached litter. Grocery bags were the most commonly found litter item in all three areas. On shorelines, foams from fishing and aquaculture were found at similar concentrations as grocery bags. The average density of floating microplastics was 36 456 items/km^2.

In a study by Horn et al. (2019), microplastics were found on all 51 Californian beaches sampled. The average microplastic count was 11.8 items per 100 mL of sediment. Fibres accounted for 95% of the microplastic items. The polymers identified were PP, isotactic PP, atactic PP, polyacrylate, PE, and polyester.

Ryan et al. (2018) collected litter items sized 2 to 25 mm on South African beaches and reported that plastics comprised 99% of all litter items by number, and that industrial pellets (which form the feedstock of the plastics industry) were the most abundant type of plastic. Typically, pellets enter the environment via accidental spills on land or at sea. Corcoran et al. (2015) found that weather conditions are a factor in industrial pellet accumulation, as is the presence of beached organic material in which they may become entrapped.

Zhou et al. (2018) studied the occurrence of microplastics on beaches adjacent to China's Bohai Sea and Yellow Sea. Microplastics were both visually identified and analyzed using FTIR, which determined that PE and PP made up the majority of samples. Flakes were the most abundant microplastic, followed by foams, fragments, fibres, pellets, films, and sponges. The abundance of microplastics between sampling sites varied significantly, ranging from 1.3 to 14 712.5 particles/kg dry weight (dw), with an overall average of 740 particles/kg. Similarly, Karthik et al. (2018) studied the occurrence of microplastics on beaches along the southeast coast of India. Microplastic particle concentration along the coast ranged from 2 to 178 particles/m^2, with a mean of 46.6 particles/m^2. FTIR analysis identified PE, PP, and PS as the main components of identified plastics.

Plastics, both macro and micro, are widely found in the Arctic, despite its distance from industrialized and highly populated areas. Plastics have been found in all abiotic environments of the European Arctic, and monitoring of beach litter in the Atlantic Arctic in 2017 revealed that the amount of beach litter varied from a mean of 1 475 items per 100 m in the spring to 195 items per 100 m in the summer months (PAME 2019).

B-2. Surface water

Plastic pollution is found in fresh and marine surface waters worldwide, and extensive research has been done on the occurrence of microplastics in marine surface waters. A brief summary of selected papers is presented below.

In the United States, Mason et al. (2016) collected surface water samples from Lake Michigan and found an abundance of microplastic particles ranging from about 1 400 to 100 000 particles/km^2 (mean of 17 267 particles/km^2), with 59% of the particles in the size range of 0.355 to 0.999 mm. Microplastic abundance was fairly evenly distributed across the lake surface, despite a seasonal gyre that developed in the southern end of the lake. Fragments dominated, followed by fibres and line, and the most common type of microplastics was PE, followed by PP. A study by Wang et al. (2018) investigated microplastics in freshwater in China. Concentrations in Dongting Lake and Hong Lake ranged from 900 to 4 659 particles/m^3, and the concentrations were much higher in the outlet channel between Dongting Lake and the Yangtze River, an area with heavy shipping traffic. The microplastics were mainly PE and PP, and the majority were fibres. Additionally, more than 65% of all microplastics were smaller than 2 mm (Wang et al. 2018).

Surface water samples were collected along the Rhine River in Europe, and microplastics were found in all samples with an average concentration of 892 777 particles/km^2. A peak concentration of 3.9 million particles/km^2 was measured in a single sample collected at Rees, in Germany, supporting the finding that higher microplastic concentrations are found near densely populated areas. Most of the microplastics recovered were spheres, followed by fragments (Mani et al. 2015).

Macroplastics have been observed floating on the Arctic Ocean surface, and microplastics have been found in Arctic Ocean surface waters and in the water column. Of the microplastics observed in surface and subsurface waters (to a depth of 6 m), 95% were fibres (Hallanger and Gabrielsen 2018). Plastics may also become entrapped in sea ice, and microplastics levels ranging from 38 to 234 particles/m^3 of ice have been measured (Obbard et al. 2014). More recently, Peeken et al. (2018) measured microplastic abundances in Arctic sea ice ranging from 1.1×10^6 particles/m^3 to 1.2×10^7 particles/m^3, with highly variable concentrations. Most of these microplastics were smaller than 50 µm in size. Of the 17 polymers identified, PE was the most common, with a mean of 48%.

In the Adriatic Sea, Zeri et al. (2018) found significantly higher macroplastic abundance in offshore waters (>4 km) than in inshore waters, but higher abundance of microplastics in nearshore waters (≤4 km) than in offshore waters. The authors collected 22 245 particles of floating microplastics from surface waters, and visually identified 658 floating macroplastics, which accounted for 91.4% of litter items recorded. They found that 29% of the macroplastics was plastic bags, 22% was plastic pieces, 15% was sheets, 13% was fish boxes of expanded PS, 8.8% was cover/packaging, 4.3% was PS pieces, and 1.4% was plastic bottles.

Floating litter collected in Vietnam consisted of a mean of 26% plastics by weight. Of the total plastic mass, 37% was plastic bags, 14% plastic packaging (single-use food containers), and 48% was other plastics, such as plastic bottles, food wrappers, cups, and cutlery (Lahens et al. 2018).

The occurrence and aggregation potential of microplastics in the Mediterranean Sea has been reported by several researchers. For example, de Haan et al. (2019) collected surface water samples using 335 µm mesh nets, which yielded 2 489 plastic particles. Microplastics made up 94.6% of plastic abundance and 55% of plastics by weight, averaging 0.10 items/m^2. The three most abundant polymers were LDPE and HDPE (54.5%), PP (16.5%) and PS (9.7%) (de Haan et al. 2019).

Bordós et al. (2019) examined microplastic occurrence in Hungary. Given the use of a 2 mm pre-filter during sampling, microplastics between 2 mm and 5 mm were not sampled. Suspected plastic particles were visually identified and analyzed under a FTIR microscope and six polymer types were identified: PE, PP, PS, PTFE, polyacrylate, and polyester. Of the 13 water samples taken, 12 contained microplastics ranging from 3.52 to 32.1 particles/m^3 with an average of 13.8 particles/m^3. All water entering sampling locations (i.e., influents) had higher microplastic concentrations than the water leaving that sampling location (i.e., effluents).

Pan et al. (2019) reported microplastics in surface waters across the northwestern Pacific Ocean. The concentration of particles collected from 18 stations varied significantly, ranging from 6.4 x 10^2 to 4.2 x 10^4 items/km^2. Microplastics were analyzed by Micro-Raman spectroscopy, yielding a distribution of 57.8% PE, 36.0% PP, and 3.4% nylon.

Poulain et al. (2019) investigated the concentration of microplastics in the North Atlantic Subtropical Gyre. Microplastics were categorized as small microplastics (SMPs, 0.025 to 1 mm) and large microplastics (LPMs, 1 to 5 mm). SMPs were collected by a 25 µm mesh net, and LMPs by a 300 µm mesh net. The authors accounted for the decreased buoyancy of SMPs compared to LMPs and applied a correction factor for the increased susceptibility of microplastics to wind-driven vertical transport. The concentrations of LMPs and SMPs corrected for vertical transport are 50 to 1000 g/km^2 and 5 to 14 000 g/km^2, respectively.

Eriksen et al. (2014) conducted 680 net tows of global surface water and found plastics in 92.3% of the tows. Visual surveys in the South Pacific, North Pacific, South Atlantic, Indian Ocean, and water around Australia also indicated that foamed PS items were the most frequently observed macroplastics. The authors estimated that there are 5.25 trillion particles of plastic floating at sea, totalling 268 940 tonnes. Their results indicate that plastic pollution has spread throughout the world's oceans and that plastics accumulate in subtropical gyres. There is an area with accumulation of buoyant plastics in the North Pacific Subtropical Gyre that is commonly referred to as the Great Pacific Garbage Patch (Eriksen et al. 2014). Lebreton et al. (2018) predicted that a 1.6 million km^2 zone of the Great Pacific Garbage Patch contains 1.8 trillion pieces of plastics and weighs 79 000 tonnes. The average plastic mass concentration measured inside the Great Pacific Garbage Patch has shown exponential increase over the last decades, from 0.4 kg/km^2 (n = 20) in the 1970s to 1.23 kg/km^2 (n = 288) in 2015 (Lebreton et al. 2018).

B-3. Benthic zone

Plastic pollution has also been detected in marine sediments around the world and is typically dominated by microplastics. Dai et al. (2018) reported the occurrence of microplastics in the surface water, water column and sediment of the Bohai Sea of the Pacific Ocean. Microplastics were detected in all 20 surface water samples, ranging from 0.4 to 5.2 particles/L, with an average of 2.2 particles/L. The average concentration of microplastics in the water column ranged from 1.6 to 6.9 particles/L. There was no clear trend in microplastic accumulation at any specific depth along the water column, and the abundance in sediments was inconsistent with the water column. The surface sediment concentration ranged from 31.1 to 256.3 particles/kg. Fibres dominated the type of microplastics found in both water and sediment, followed by fragments. μ-FTIR analysis identified that the polymer with the highest density in surface waters was PS, whereas PET and PVC were found at highest densities in deeper water.

In Argentina, an average of 25 macroplastic items/m^2 and 704 microplastic fragments/m^2 were collected from sediment. The macroplastic pollutants were categorized into 24 types, and the most dominant types were food wrappers (PP and PS), bags (HDPE and LDPE), bottles (PET), and disposable Styrofoam food containers (PS) (Blettler et al. 2017). In a study in the United Kingdom, the main types of macroplastic pollution found in sediment were packaging, fishing and shipping waste (Browne et al. 2010). Macroplastics and microplastics were found in sediments from a marine protected area in Italy, ranging from a mean of 11.9 to 46.4 pieces and 151.0 to 678.7 pieces per kg dw of sediment, respectively. Greater than 85% of the microplastics were fibres (Fastelli et al. 2016). Bordós et al. (2019) sampled sediment in Hungary and found that 9 of the 12 sediment samples contained microplastics ranging from 0.46 to 1.62 particles/kg, with an average of 0.81 particles/kg. The most dominant polymer was PP. Marine litter in Croatia ranged from 3.4 items/kg dw to 528 items/kg dw, with macroplastics making up 1.3% to 11.3% of samples. Like in Italy, fibres were the most abundant type of microplastic found in Croatia, ranging from 39.9% to 90.1% of the total number of plastic items (Renzi et al. 2019a; Blašković et al. 2017). Blašković et al. (2017) found no correlation between the extent and pattern of plastic contamination and sediment grain size or sampling depth. In Svalbard in the Arctic, fibres were once again found to be the dominant microplastic in sediment at depths of 40 to 79 m, where they were sampled at a density of 9.2 fibres/kg (Sundet et al. 2016).

Vidyasakar et al. (2018) conducted the first study on the distribution and characteristics of plastic pollutants in marine sediment on Rameswaram Island, along the southeast coast of India. PP was the most abundant polymer type, followed by PE, PS, nylon, and PVC. Irregularly shaped plastics were most plentiful at 69.2%, followed by fibres at 17.9% and pellet-shaped plastics at 12.9%.

Microplastics have been found in large quantities in river sediment in Shanghai (Peng et al. 2018), at concentrations ranging from 5.3 particles/100 g dw to 160 particles/100 g dw. The average concentration across all sites was 80.2 particles/100 g dw. Residential areas showed the highest level of microplastic concentration, followed by parks, rural areas, and tourist areas. Spheres constituted the majority of microplastics at 88.98%, followed by fibres (7.55%) and fragments (3.47%). The two most dominant polymer types identified by μ-FTIR were PP and polyesters (Peng et al. 2018).

García-Rivera et al. (2018) derived data from the MEDITS (International Bottom Trawl Survey in the Mediterranean) program surveys and found that, over 11 years, 2197.8 kg of marine litter was collected from the Spanish Mediterranean seafloor (collected five stratum levels at depths from 0 to 800 m) and was comprised of 29.3% plastics by weight. They reported that the amount of marine litter generally remained stable over the survey period. Deep sea litter in the Arctic reportedly increased from 346 items/km^2 in 2004 to 8082 items/km^2 in 2014, with plastics accounting for 47% of litter (PAME 2019). A plastic bag was found in the Mariana Trench at a depth of 10 898 m (Chiba et al. 2018).

Appendix C: Additional information on occurrence of microplastics in food

Table C-1: Summary of the occurrence data for microplastics in food

Food Item	Concentration	Size (µm)	Shape	Source
Fish	0 to 20 microplastic particles/fish (gastrointestinal tract)	130 to 5 000	Predominantly fragments and fibres	Lusher et al. 2013; EFSA 2016; Campbell et al. 2017; FAO 2017; Barboza et al. 2018; Liboiron et al. 2018, 2019; Slootmaekers et al. 2019; Hantoro et al. 2019; Toussaint et al. 2019
Fish	0 to 4.6 microplastic particles/fish (muscle); 0.57 to 1.85 microplastic particles/g fish (muscle)	100 to 5 000 (fibres) 100 to 500 (fragments)	Predominantly fragments and fibres	Karami et al. 2017a; Abbasi et al. 2018; Akhbarizadeh et al. 2018
Molluscs	0 to 10 microplastic particles/individual mussel;[a] 0.2 to 2.9 microplastic particles/g mussel	5 to 4 700	Predominantly fragments and fibres	De Witte et al. 2014; Van Cauwenberghe and Janssen 2014; Li et al. 2015, 2018a; Van Cauwenberghe et al. 2015; Catarino et al. 2018; Naji et al. 2018; Su et al. 2018; Patterson et al. 2019; Toussaint et al. 2019
Crustaceans	1.23 microplastic particles/individual whole shrimp;[b] 0.68 microplastic particles/g whole shrimp wet weight	200 to 1 000	Predominantly fibres	Devriese et al. 2015
Crustaceans	7.8 microplastic particles/individual prawn (muscle tissue and exoskeleton)	100 to 250	Predominantly filamentous fragments	Abbasi et al. 2018
Crustaceans	0.80 mg of microplastic particles/individual lobster (gastrointestinal tract)	Not reported	Predominantly fibres	Murray and Cowie 2011; Welden and Cowie 2016
Salt[c]	0 to 19 800 microplastic particles/kg sea salt	4 to 5 000	Fragments and fibres were most abundant shape for all salt types	Yang et al. 2015; Iñiguez et al. 2017; Karami et al. 2017b; Gündoğdu 2018; Kim et al. 2018; Renzi and Blašković 2018; Seth and Shriwastav 2018; Lee et al. 2019; Peixoto et al. 2019

[a] Mussels are the most frequently investigated species of mollusc. Similar concentrations of microplastics have been reported in clams, oysters, scallops, and snails.

[b] Microplastics were only observed in the digestive tract, head, and gills of the whole shrimp and not in the abdominal muscle tissue of peeled shrimp.

[c] Microplastic concentrations in salt varied considerably depending on the origin and type of salt.

Table C-2: Summary of the occurrence data for microplastics in bottled water

Type of Bottle	Concentration (microplastics/L)	Size (µm)	Shape	Location	Source
Plastic (not specified if single or multi-use)	10.4	>100	Predominantly fragments	Multiple locations worldwide (Brazil, China, France, Germany, India, Indonesia, Italy, Lebanon, Mexico, United Kingdom, United States of America)	Mason et al. 2018
Single-use PET plastic	2 649 ± 2 857	≥1	Not reported	Germany	Oßmann et al. 2018
Multi-use PET plastic (newer bottles)	2 689 ± 4 371				
Multi-use PET plastic (older bottles)	8 339 ± 7 043				
Glass	6 292 ± 10 521				
Single-use PET plastic	14 ± 14	≥5	Not reported	Germany	Schymanski et al. 2018
Multi-use PET plastic	118 ± 88				
Beverage Cartons	11 ± 8				
Glass	50 ± 52				

Appendix D: Additional information on ecotoxicological studies

Table D-1: Aquatic: freshwater

Organism and Exposure Duration	Microplastic Type and Concentration	Summary of Effects	Source
Daphnia magna (Water flea) For uptake experiments, exposure was 15, 30, 60, 120 and 240 minutes For depuration experiments, exposure was 1 hour For differential food regime experiments, exposure was 1 hour For chronic toxicity tests, exposure was 21 days	Yellow-green fluorescent, carboxylate-modified PS (2 µm) were used for uptake and depuration experiments Non-fluorescent PS microplastics (2 µm) were used for toxicity tests For uptake and depuration experiments, microplastic concentration was 1.46×10^{2} mg/L and algae concentration was 1.00×10^{-1} mg/L For differential food regime experiments, microplastic concentrations were 6.93×10^{-4}, 1.39×10^{-3}, 2.77×10^{-3}, 5.54×10^{-3}, 8.31×10^{-3}, and 1.11×10^{-2} mg/L; algae concentrations were 5.00×10^{-2}, 1.00×10^{-1}, 2.00×10^{-1}, 4.00×10^{-1}, 6.00×10^{-1}, 8.00×10^{-1} mg/L Control groups for uptake, depuration, and differential food regime experiments were not exposed to algae For chronic toxicity tests, microplastic concentrations were 1.39	Uptake and depuration tests of microplastics indicate that *D. magna* fed both microplastics and algae consumed a significantly lower amount of microplastics than *D. magna* that only ate microplastics. Using differential food regime experiments, it was also found that this effect could also be seen when using low concentrations of algae and that increasing algal concentrations led to decreasing microplastic uptakes. In adult *D. magna*, mortality was seen in all treatment groups compared to the control following seven days of exposure. When using a low algal concentration (1.00×10^{-1} mg/L) with a relatively higher microplastic concentration (1.11×10^{-2} mg/L), the LT_{50} was 10.09 ± 0.70%, which is slightly lower than the control at the same algal concentration. No impact on reproduction was seen. In neonate *D. magna*, mortality in those fed a low algal concentration (1.00×10^{1} mg/L) and microplastics was significantly higher than neonates fed only algae. No effect on mortality was found for a high algal concentration (8.00×10^{-1} mg/L) and microplastic uptake. There were no significant differences in reproduction between identical food regimes with and without microplastics.	Aljaibachi and Callaghan 2018

	× 10^{-3} mg/L (low) and 1.11 × 10^{-2} mg/L (high); algae concentrations were 1.00 × 10^{-1} mg/L (low) and 8.00 × 10^{-1} mg/L Control group for chronic toxicity tests was not exposed to microplastic		
Xenopus laevis (African clawed frog) tadpoles Developmental stages 36 to 46	Blue PS microplastics (2.75 ± 0.09 µm) at 0.125 µg/mL, 1.25 µg/mL, and 12.5 µg/mL (nominal) Control group was not exposed to microplastic	Microplastics were found in the tadpoles' digestive tract from each tested concentration; however, SEM analyses suggest no mechanical damage in the epithelium walls as a result. Microplastics were not found in the gills. No significant effects on mortality, body growth, or swimming activity (swimming speed or distance moved) during early life stages were seen.	De Felice et al. 2018
Carassius auratus (Goldfish) 6 weeks	Ethylene vinyl acetate fibres (0.7–5.0 mm), PS fragments (2.5–3.0 mm), and polyethylene acrylate pellets (4.9–5.0 mm) Fish were fed concentrations of 1.36%, 1.94%, and 3.81% (g (food+microplastics)/g ww[b] fish) for the fibres, fragments, and pellets, respectively Control group was given food pellets that contained no microplastic	Various sublethal effects, but no mortality, were observed. Fish exposed to plastic fibres, fragments, and pellets showed significant weight loss compared to the control group. Fragments and pellets were chewed and expelled by fish. The highest occurrence of changes in the upper (27.0%) and lower (30.4%) jaws were seen in the fragment group, followed by the fish exposed to pellets. Damage to the buccal cavity was seen in 80.0% of fish that chewed plastic fragments. This damage ranged from slight exfoliation to deep incisions. In addition, 13.1% of fish exposed to fragments showed sinusoid dilation in their livers.	Jabeen et al. 2018

		Fibres were found in the gills, gastrointestinal (GI) tract, and feces, but were not likely to accumulate in the GI tract. The frequency of pronounced changes in the upper jaw was the highest in fish exposed to fibres. Additionally, this group showed pronounced and severe damage in their livers. The highest organ index values for the upper jaw, liver, and intestines of fish were also seen in those exposed to fibres. The distal intestine displayed more pronounced and severe changes in comparison to the proximal intestine, which could also be a result of fibre ingestion.	
Daphnia magna *Daphnia pulex* *Ceriodaphnia dubia* (Water fleas) 96 hours	Green fluorescent plastic microspheres (1–5 µm) were used as primary microplastic models Irregularly-shaped PE microplastics (approx. 1–10 µm) were used as secondary microplastic models Concentrations were 10^3, 10^4, 10^5, 10^6, 10^7 particles/mL Control group was not exposed to microplastic	Using no-effect concentration estimates and three different temperatures (18°C, 22°C, 26°C), the sensitivity of *D. magna* and *D. pulex* to primary and secondary microplastics was found to drastically increase with temperature. This effect was not seen in *C. dubia*. At the lowest tested temperature (18°C), *C. dubia* was the most sensitive species. At the highest temperature (26°C), *D. magna* and *D. pulex* were more sensitive. Primary microplastics were found to be more toxic than secondary microplastics in *C. dubia*. For all species, survival was time-dependent as seen in LC_{50} estimates compared at 48 hours and 96 hours. In *D. magna*, for example, the 48-hour LC_{50} was 32.0 particles/mL, whereas the 96-hour LC_{50} was 18.0 particles/mL at 18°C.	Jaikumar et al. 2018
Danio rerio (Zebrafish)	Virgin PA, PE, PP, and PVC particles (mean diameter of about 70 µm)	In *D. rerio*, there were no significant differences in lethality following 0.001–10.0 mg/L microplastic	Lei et al. 2018a

10 days	Two kinds of PS (nominal sizes of 1.0 µm and 5.0 µm) particles were used: virgin PS for the toxicity test and red-fluorescently-labelled PS to examine microplastic distribution in *C. elegans* For *D. rerio*, concentrations of 0.001 mg/L, 0.01 mg/L, 0.1 mg/L, 1.0 mg/L and 10.0 mg/L were used For *D. rerio*, dechlorinated tap water was used for the control group	exposure. In the surviving fish, PA, PE, PP, and PVC particles caused intestinal damage (including cracking of villi and splitting of enterocytes) in 73.3% to 86.7% of individuals.	
Chlorella pyrenoidosa (Green algae) 30 days (comprised of three growth periods: lag phase, logarithmic phase, and stationary phase)	PS microbeads (1.0 µm) at 10 mg/L, 50 mg/L, and 100 mg/L in algal cultures Control group was pre-cultured *C. pyrenoidosa* in the logarithmic growth phase added into BG-11 medium without microplastic	1.0 µm PS caused a dose-dependent decrease in *C. pyrenoidosa* growth from the lag to early logarithmic phases (day 0 to 22). At 10, 50 and 100 mg/L PS, there was a growth inhibition ratio of 20.9%, 28.4% and 38.1%, respectively. From the lag to early logarithmic phases, microplastics (100 mg/L) had a negative effect on photosynthesis. However, the end of the stationary phase onwards showed a stimulation of photosynthesis that was also dose-dependent. In the presence of microplastics, distorted thylakoids and cell wall thickening were also observed. Following 25 days of exposure, cell morphology mostly recovered.	Mao et al. 2018
Daphnia magna (Water flea) 21 day exposure	Red fluorescent microspheres (1–5 µm) at 0.1 mg/L	Chronic exposure of *D. magna* to microplastics caused parental mortality (10% to 100%) and a significant decrease in growth, reproduction (total offspring and	Martins and Guilhermino 2018

Four sequential generations	Control group was exposed to a clean test medium	mobile juveniles), and population growth rate. In two treatment groups, microplastic-exposed populations were extinct in the F_1 (2^{nd}) generation. Juveniles produced by microplastic-exposed females were immobile. Some recovery was visible in the F_1 population, such as an increase in production of mobile juveniles and earlier first brood release. However, females descending from the exposed population in F_0 (called the recovery model population) still experienced a significant reduction in growth, reproduction, and population growth rate up to the F_3 generation, in comparison to controls. These findings demonstrate that full recovery from developmental and reproductive effects may take several generations.	
Daphnia magna (Water flea) 14 and 21 days	Fluorescent red microspheres (1–5 µm) at 0.02 mg/L and 0.2 mg/L (nominal) Control group was exposed to hard water without microplastic	When exposing *D. magna* for 14 days to microplastics, there was a significant reduction in the number of total offspring and a higher frequency of immobile juveniles. No effects on parental female mortality were seen. When exposing *D. magna* for 21 days to microplastics, there was a dose-dependent effect on mortality. At 0.02 mg/L, microplastics induced 10% of mortality. However, at 0.2 mg/L, microplastics induced 30% of mortality. There were no significant effects on growth. In the 21-day exposure treatment, microplastics also reduced the reproductive fitness of *D. magna*. Exposure increased the time of first	Pacheco et al. 2018

		brood release (49%) and reduced the total number of broods released by 71%. Similar to the 14-day treatment, there was also a decrease in the number of offspring and induction of immobile juveniles. There was no effect on the number of aborted eggs in both exposure regimes.	
Danio rerio (Zebrafish) For distribution experiments, exposure times were 20 hours (4–24 hpf[c]) and 92 hours (4–96 hpf) For uptake and qPCR[d] experiments, exposure was 92 hours (4–96 hpf) For developmental effects experiments, exposure was 68 hours (4–72 hpf) For free swimming and light-to-dark experiments, exposure was 116 hours (4–120 hpf)	Green fluorescent PS microplastics (1 µm) For distribution, developmental effects, free swimming, light-to-dark, and qPCR analysis, concentrations used were 100 µg/L and 1 000 µg/L For uptake experiments, concentrations used were 10, 100 and 1 000 µg/L Control group was exposed to embryo medium without microplastic	Microplastics were found to adhere to the embryo chorion and its distribution increased with increasing PS concentration. Microplastic uptake also increased with increasing exposure concentrations. Hatching rate was slightly reduced with exposure; however, this result was not significant. Development speed (in terms of body length and yolk sac area) of larvae was also not impacted significantly from 4–72 hpf. Larvae did not display any obvious malformations. In the free swimming test, exposure to 1 000 µg/L microplastics led to a significant decrease in both swimming distance and larvae speed in dark conditions by 3.2% and 3.5%, respectively. Using an alternating light-to-dark photoperiod stimulation, a significant reduction in swimming competence was also seen in dark conditions. At 1 000 µg/L exposure, total swimming distance was reduced by 2.6% and swimming speed was 2.8% lower in comparison to the control. No significant differences were found when exposed to light conditions. In the 1 000 µg/L exposure group, *il1b* and *cat* expression were upregulated to 165% and 121%,	Qiang and Cheng 2019

		respectively. No significant changes were seen in *sod* expression.	
Danio rerio (Zebrafish) 21 days	PS microbeads (5 µm) at 50 µg/L and 500 µg/L Control group was exposed to culture water without microplastic	Significant intestinal damage was observed in 78% and 86% of the histological sections sampled for the 50 µg/L and 500 µg/L treatment groups, respectively. Microplastic exposure was found to induce intestinal oxidative stress and increased permeability. In addition, there were significant alterations in the intestinal metabolic profiles and gut microbiome.	Qiao et al. 2019a
Daphnia magna (Water flea) 10 days	Uncoated PS particles (1.25 µm) at 2 mg/L, 4 mg/L, and 8 mg/L Control group was not exposed to microplastic	No mortality occurred in all treatments. Reduction in body growth rate, an indicator of population fitness, was also seen with microplastic exposure. Following PS exposure, transcript level of TRxR in *D. magna* (vital in mediating oxidative defence) significantly increased (2.5–5-fold) with PS concentrations of 2 and 4 mg/L. Transcript level declined at 8 mg/L, but was still significantly higher in comparison to the control group. Arginine kinase (vital in cellular energy production and ATP buffering) transcript level was significantly elevated in the presence of PS (approx. 5-fold at 2 mg/L). Transcript level of permease (facilitates removal of cytotoxic compounds from cells) increased 1.4–1.8 fold when exposed to 2 and 4 mg/L PS. Exposure to 8 mg/L lowered transcription compared to the control.	Tang et al. 2019
Gammarus pulex (Amphipod)	Irregular particles (10–150 µm) were prepared from green fluorescent	In the uptake experiment, no mortality was found. In addition, body burden was found to be dependent on dose and age. Body	Weber et al. 2018

In the uptake study, exposure was 24 hours In the chronic exposure study, exposure was 48 days	soft drink bottles made from PET In the uptake study, concentrations used were 0.8 particles/mL, 40 particles/mL, and 4000 plastics/mL In the chronic exposure study, concentrations used were 0.8 particles/mL, 7 particles/mL, 40 particles/mL, 400 particles/mL and 4000 particles/mL Negative control group was exposed only to ISO medium; solvent control group was exposed to ISO medium with 10% cetyl alcohol	burden was significantly higher in juveniles in comparison to adults for the 0.8 particles/mL and 4000 particles/mL treatments. No significant difference was seen at 40 particles/mL. Furthermore, a higher dosage of microplastics was associated with a significantly higher body burden in both juveniles and adults. In the chronic exposure study, no significant effects were seen on feeding activity, energy reserves and molt periods. Mortality rates also did not vary in juveniles; however, mortality was significantly increased in adults for the 7 particles/mL and 400 particles/mL treatments compared to the control.	
Eriocheir sinensis (Chinese mitten crab) For uptake experiments, exposure was seven days For toxicity tests, exposure was 21 days	Two kinds of PS microspheres (5 µm) were used: fluorescent microspheres for uptake and accumulation experiments, and virgin microspheres for toxicity tests For uptake experiments, a concentration of 40 000 µg/L was used For toxicity tests, nominal concentrations were 40 µg/L (5.4×10^2 particles/mL), 400 µg/L (5.4×10^3 particles/mL), 4 000 µg/L (5.4×10^4 particles/mL) and 40 000 µg/L (5.4×10^5 particles/mL)	No significant differences in survival were seen with microplastic exposure. Weight gain, specific growth rate, and hepatosomatic index generally decreased with increasing microplastic concentration, with the exception of specific growth rate in the 40 µg/L group. In the uptake experiments, microplastics (40 000 µg/L) accumulated in the gills, liver and guts of *E. sinensis*. Acetylcholinesterase, alanine aminotransferase, and catalase activities in all treatment groups were significantly lower than seen in the control. The activities of superoxide dismutase, aspartate transaminase, GSH[e], and GPx[f] increased in crabs	Yu et al. 2018

	Control group was not exposed to microplastic	exposed to 40 and/or 400 µg/L microplastics. However, there was a general decrease in activity with high exposure (4 000 and 40 000 µg/L). Genes encoding the antioxidants SOD[g], catalase, GPx, and GST[h] in the liver initially increased and then decreased in expression following exposure. Further, there was an increased expression of the gene encoding p38 in the MAPK[i] signaling pathway with treatment of 4 000 µg/L and 40 000 µg/L microplastics, but significant reductions in the expression of ERK[j], AKT[k], and MEK[l]. No significant differences in transcription were found with the gene encoding c-Jun N-terminal kinase. These results show that microplastic exposure can induce oxidative stress in the liver of *E. sinensis*.	

[a] Median lethal time
[b] Wet weight
[c] Hours post fertilization
[d] Quantitative polymerase chain reaction
[e] Glutathione
[f] Glutathione peroxidase
[g] Superoxide dismutase
[h] Glutathione-S-transferase
[i] Mitogen-activated protein kinase
[j] Extracellular signal-regulated kinase
[k] Protein kinase B
[l] Mitogen-activated protein kinase (MAPK) kinase

Table D-2: Aquatic: marine

Organism and Exposure Duration	Microplastic Type and Concentration	Summary of Effects	Source
Oncorhynchus mykiss (Rainbow trout) Four weeks	Colourless PS particles (100–400 µm) at approx. 500– 700 particles/day/fish Control group was not exposed to microplastic	Using histological analysis, no significant effects were seen on the abundance of mucus-secreting goblet cells in the proximal and distal segments of the trout intestine. In addition, there were no adverse changes in tissue morphology, paracellular permeability, and intestinal transporting functions (^{3}H-lysine transport, ion transport capacity, and net ion flow) in the intestines following exposure. PS microplastics did not induce pro-inflammatory or anti-inflammatory responses in the distal and proximal segments of the intestines.	Ašmonaitė et al. 2018
Brachionus plicatilis (Rotifer) 48 hours *Tigriopus fulvus* (Crustacean) 48 hours *Acartia clausi* (Marine copepod) 48 hours *Mytilus galloprovincialis* (Mussel) 48 hours	Non fluorescent LDPE microplastics (1–500 µm) Fluorescent green and red PE microplastics were used to examine particle ingestion in rotifers, copepod, and mussel larvae (nominal size of 1–5 µm) Virgin microplastic loads tested varied with each organism and consisted of 0, 0.01, 0.1, 1, 3, 10, 20, 30, 50, 100 mg/L Control group was exposed to 0.22 µm-filtered seawater without microplastic	Virgin microplastics had no significant effect on mussel embryonic development at any concentration under static conditions or in a rotary wheel. However, orbital shaking at 200 rpm significantly reduced the percentage of D-veliger larvae following exposure. Virgin microplastics did not cause any significant effect at any concentrations below 30 mg/L in any of the species tested. Exceptions to this were for the 1–4 µm particles, which produced a LOEC of 0.01 mg/L for *B. plicatilis* immobility, LOEC of 1 mg/L for *B. plicatilis* mortality (LC_{50} >10 mg/L), and a LOEC of 1 mg/L for *T. fulvus* mortality (LC_{50} = about 1.82 mg/L).	Beiras et al. 2018

Paracentrotus lividus (Sea urchin) 48 hours *Oryzias melastigma* (Fish) 1–13 days post fertilization			
Lophelia pertusa (Cold-water coral) For capture rate and polyp activity experiments, exposure was 7, 20, or 47 days For coral growth rate experiments, exposure was 69 days	LDPE microbeads (500 µm) at 350 beads/L Control group was not exposed to microplastic; control measurements were done in flumes containing no corals to quantify zooplankton sedimentation for the prey capture rate experiment	The capture rates of corals were significantly lower than in the controls at 7 and 20 days after microplastic exposure. After 47 days, however, they were not significantly different from the controls, indicating a possible behavioural compensatory response over time. Although microplastics did not impact polyp behaviour, coral exposed to microplastics also had a significantly lower skeletal growth rate in comparison to the control and *in situ* experimental conditions. Calcification was also reduced.	Chapron et al. 2018
Isochrysis galbana, clone T-ISO (Microalgae) 72 hours	PE micronized powder (1.4–42 µm; average particle size of 3.29 µm) at 0.5 mg/L, 1 mg/L, 10 mg/L and 25 mg/L Control group was microalgae with surfactant at its highest concentration	Daily growth rate was not affected by microplastic exposure for all test concentrations. A lower percentage of cellular inhibition was seen when chlorpyrifos were sorbed to microplastics, indicating that it could modulate its toxicity in *I. galbana*.	Garrido et al. 2019
Montastraea cavernosa (Large polyp coral) *Orbicella faveolata* (Small polyp coral)	Experiment 1 (Effects of microbeads on calcification): Fluorescent, PE microbeads (size ranges of 90–106 µm, 425–	In experiment 1, no significant differences in calcification were seen between the control and the exposed group treated with microplastics.	Hankins et al. 2018

2 days	500 µm, and 850–1 000 µm). Experiment 2 (Determination of ingestion size ranges and retention): Uncured, PE microbeads (size ranges of 212–250 µm, 425–500 µm, 850–1000 µm, 1.7–2.0 mm, and 2.4–2.8 mm). Polyps were fed three microbeads from each size class. Experiment 3 (Comparing microbeads and microfibres): Uncured, fluorescent, PE microbeads (425–500 µm) and uncured, fluorescent polyester microfibres (3–5 mm long). Polyps were fed three plastics of each type. Control groups were not exposed to microbeads; for experiment 2, control group was given food that contained no microbeads	In experiment 2, it was determined that *M. cavernosa* and *O. faveolata* ingested 425–500 µm, 850–1 000 µm, 1.7–2.0 mm, and 2.4–2.8 mm microbeads offered. However, a 212–250 µm size class did not elicit a feeding response in either species. No significant differences in egestion were evident in any size classes. In experiment 3, *M. cavernosa* egested 100% of the microbeads and microfibres. *O. faveolata* egested means of 80.0% ± 23.3 and 76.7% ± 35.3 for microbeads and microfibres, respectively. There was no significant difference in ingestion between microbeads and microfibres.	
Acanthurus triostegus (Convict surgeonfish) 3, 5 and 8 days	PS microbeads (90 µm) at 5 particles/mL (nominal) Control group was exposed to seawater without microplastic	Exposure to microbeads for 3, 5 and 8 days did not alter the foraging activity (measured as number of bites) in *A. triostegus*. The survival of post-larvae to predation was also not significantly affected, compared to the control.	Jacob et al. 2019
Brachionus koreanus (Monogonont rotifer)	Non-functionalized PS microbeads (0.5 µm and 6 µm) For toxicity tests, concentrations used	Toxicity of beads was size- and concentration-dependent. In the 6 µm treatment group, *B. koreanus* had slightly irregular growth, and no significant	Jeong et al. 2016

For toxicity tests, exposure was 12 days For ingestion, egestion, ROS[a] levels, MAPK activation, and antioxidant enzyme experiments, exposure was 24 hours	were 0.1 µg/mL, 1 µg/mL, 10 µg/mL, and 20 µg/mL For ingestion, egestion, ROS levels, MAPK activation, and antioxidant enzyme experiments, the concentration used was 10 µg/mL Control group was not exposed to microplastic	changes in fecundity and life span. Microbeads were ingested by the rotifers at both sizes. The authors hypothesize that 0.5 µm microplastics have longer retention times that correlate to more negative effects. Increased enzymatic activities of GPx, GR[b], GST, and SOD were seen for the 0.5 µm beads. Exposure to 6 µm microplastics had levels similar to that of control conditions. The level of total GSH content was not significantly different for any exposure concentration.	
Paracyclopina nana (Marine copepod) 24 hours	Non-functionalized PS microbeads (0.5 µm and 6 µm) For toxicity tests and ROS levels experiments, concentrations used were 0.1 µg/mL, 1 µg/mL, 10 µg/mL, and 20 µg/mL For ingestion, egestion, western blot, and antioxidant enzyme experiments, the concentration used was 10 µg/mL Control group was not exposed to microplastic	Microbeads of both sizes were ingested but egestion was size-dependent; fluorescence was present for the 0.5 µm microbeads 24 hours after exposure, but not in the 6 µm group. *P. nana* exposed to 0.5 µm microbeads showed delayed molting. No observable effects were seen with 6 µm microbeads. ROS levels were increased in the 0.5 µm group compared to the control, however not significantly. In addition, the antioxidant enzymes GPx, GR, GST, and SOD had higher activity in the 0.5 µm group.	Jeong et al. 2017
Sparus aurata (Gilt-head seabream) 45 days	6 microplastics were used: PVC (high molecular weight; 75.6 ± 15.3 µm), PA (111.7 ± 32.2 µm), PE (ultra-high molecular weight; 23.4 ± 7.6 µm), PS (51.0 ±	Total biomass of the fish per tank was not affected by microplastic exposure. Levels of glucose, aspartate transaminase, alanine transaminase, lactate	Jovanović et al. 2018

	36.3 µm), PE (average molecular weight medium density; 54.5 ± 21.3 µm), PVC (low molecular weight; 87.6 ± 16.8 µm) Concentration used was 0.1 g/kg body weight/day Control group was given food that contained no microplastic	dehydrogenase, and gamma-glutamyl transferase did not differ significantly from control conditions following exposure, indicating a lack of stress. The retention of virgin microplastics in the *S. aurata* GI tract was low. However, 5.3% of all livers examined contained at least one plastic particle following 24 hours. In addition, there was no significant difference in overall histopathology between the different treatment groups.	
Crepidula onyx (Slipper limpet) 95 days post hatching	PS microplastics (2–5 µm) In the first trial, microplastic concentrations were 30% (low plastic ratio) and 70% (high plastic ratio) of algal concentration used. Final microplastic concentrations were 6 x 10^4 particles/mL and 1.4 x 10^5 particles/mL for the low and high plastic ratio treatments, respectively. In the second trial, an additional treatment of 10 particles/mL was added Control group was fed algae	Exposure to 10 particles/L microplastics had no significant effect on growth rate and settling rate in larval *C. onyx*. No significant difference was seen for juveniles at this concentration. Larval survival was not affected by microplastic addition at a high plastic ratio. In trial 1, adding microplastics appeared to negatively affect growth rates in larvae. Growth rate was reduced when using low plastic ratio and high plastic ratio treatments in comparison to the control. However, settling rate increased in larvae exposed to microplastic. Settling occurred earlier at a smaller size in this group as a result of their reduced growth rate. Microplastic exposure did not have an effect on survival rates or penis development in juveniles; however, there was a negative effect on growth rate. The microplastic group had a 25% slower growth rate	Lo and Chan 2018

		in comparison to the control group. *C. onyx* exposed to only microplastics during their larval stage continued to display slower growth rates than the control, even if microplastics were not present in their environment for 65 days. This finding indicates a legacy effect of microplastic exposure.	
Mytilus edulis (Blue mussel) For ingestion and egestion tests, exposure time was Four hours For the larval growth tests, exposure time was 15 days	Fluorescent PS beads (2 µm) For ingestion and egestion tests, PS concentrations used were 0.70 mg/L, 1.05 mg/L, and 1.40 mg/L (based on a plastic-to-algae ratio) For the larval growth tests, concentrations used were 0.42 µg/L, 28.2 µg/L and 282 µg/L Control group was not exposed to microplastic; for ingestion and egestion tests, control group was only exposed to algae	The body burden (mass of microplastics per individual) was found to be 4.9 ng/larvae, 3.4 ng/larvae, and 3.1 ng/larvae for the 2 µm beads for bead concentrations of 1.40 mg/L, 1.05 mg/L, and 0.70 mg/L, respectively. No significant effect on larval growth rate was seen, but exposure to beads led to an increase in abnormally developed larva. Malformations were more frequent with increasing concentrations and exposure times. From day 11 on, 40% to 60% of all larvae showed signs of abnormal development.	Rist et al. 2019
Thalassiorira pseudonana (Marine diatom algae) *Dunaliella tertiolecta* (Marine flagellate algae) *Chorella vulgaris* (Green microalgae) 72 hours	Uncharged PS microbeads (0.5 µm and 6.0 µm) were used on *D. tertiolecta* Negatively charged carboxylated PS microbeads (0.5 µm) were used on all three test species Concentrations used were 25 mg/L and 250 mg/L (nominal).	Using pulse amplitude modulation fluorometry, uncharged and negatively charged beads displayed no significant effect on photosynthetic efficiency in all three test species. A small decrease (11%) in *D. tertiolecta* growth was observed with exposure to uncharged 0.5 µm beads along with a 13% inhibition of growth	Sjollema et al. 2016

	Authors noted that the average measured concentration was up to 9x lower than the nominal concentration in the 6.0 µm treatment group. Control group was not exposed to microplastic	rate. Effects were less than 10% for the 6 µm beads.	
Sebastes schlegelii (Jacopever) 14 days	Green fluorescent PS microbeads (15 µm) at 1×10^6 microplastics/L Control group was not exposed to microplastic	Microplastics were found in the gills and intestines following 14-day exposure and 7-day depuration. No translocation to the liver was seen, however. 14-day exposure to microplastics caused feeding time to significantly increase (by approximately two-fold). Foraging time was rapidly reduced and shoaling behaviour (staying in close proximity to one another) was shown through a reduction in mean distance between fish. In addition, mean swimming speed was reduced and fish used a significantly smaller volume of their tank when foraging in comparison to control fish. Histopathological changes in the liver (hyperaemia), gallbladder (bile turned black in colour), and intestines (altered morphology) of fish were seen following 14 day exposure to microplastics. After 14-day exposure and 7-day depuration, no mortalities were observed; however, there was a significant reduction in growth and energy reserves. Weight gain rate decreased	Yin et al. 2018

		from 8.92 ± 0.98% in controls to 3.09 ± 0.32% in the microplastic-exposed group.	

[a] Reactive oxygen species
[b] Glutathione reductase

Table D-3: Soil

Organism and Exposure Duration	Microplastic Type and Concentration	Summary of Effects	Source
Folsomia Candida (Soil springtail) 28 days	PE beads (<500 µm; size distribution of 32% with <50 µm, 25% between 50 and 200 µm, and 43% between 200 and 500 µm) Concentrations used were 0.005%, 0.02%, 0.1%, 0.5%, and 1% microplastics w/w in dry soil Control group was exposed to soil without microplastic	Average survival rates were higher than 80% in all three conditions. Springtails displayed significant avoidance behaviours at 0.5% and 1% (microplastics w/w in dry soil) that appeared to be concentration-dependent. The avoidance rates were 59% and 69%, respectively. Reproduction rate decreased with increasing microplastic concentrations. At the highest tested concentration of 1% microplastics, the reproduction rate was reduced by 70.2%. The EC_{50} was 0.29% microplastics w/w in dry soil. At concentrations of 0.5% dw soil, microplastics significantly altered the microbial community (and decreased bacterial diversity in the springtail gut). Alphaproteobacteria and *Wolbachia* were significantly less prevalent when exposed to microplastics. However, Bradyrhizobiaceae and *Ensifer* were significantly increased in the exposed group.	Ju et al. 2019
Lobella sokamensis (Soil springtail) Three minutes	Plastic microbeads (average diameters of 0.50 ± 0.01 µm,	The influx of microplastic particles in soil disrupted the movement of *L. sokamensis*. The springtails moved to avoid	Kim and An 2019

	29 ± 4 µm, and 248 ± 14 µm) Plastic fragments (average diameters of 44 ± 39 µm, 282 ± 131 µm, and 676 ± 479 µm) Concentrations in soil were 4 and 8 mg/kg for the roughly 0.50 µm microbeads and for the remaining microplastic sizes, the concentration used was 1 000 mg/kg Concentrations in solution were 10 mg/L and 20 mg/L for the roughly 0.50 µm microbeads Control group was exposed to soil without microplastic and deionized water	becoming trapped, and this behaviour created bio-pores in the soil system. The influx of plastic particles into these cavities subsequently immobilized the springtails within. Using a movement index to quantify springtail behaviour, it was found that movement was significantly different in all size groups in comparison to the control. Specifically in the roughly 0.50 µm microbead solution at 8 mg/kg, movement decreased significantly compared to the other treatment groups.	
Caenorhabditis elegans (Nematodes) Three days	PS microplastics (0.5 µm, 1.0 µm, 2.0 µm, and 5.0 µm) at 1.0 mg/L Control group was exposed to suspension solution without microplastic	PS microplastics displayed size-dependent effects on lethality. Survival rates were reduced in all treatment groups. The 1.0 µm group had the lowest mean reduction in survival of 32.27%. In addition, the 1.0 µm group also had significant decreases in body length and average lifespan. Microplastic exposure resulted in an increase in the number of head thrashes and body bends in the 0.5 µm group but decreases in locomotion for the other treatment groups. However, exposure to 2.0 µm PS led to significant increases in mean crawling speed.	Lei et al. 2018b

		Exposure to microplastics led to damage in cholinergic neurons (i.e., broken ciliated dendrites) in all treatment groups, indicating a downregulation of *unc-17* (encodes acetylcholine in cholinergic neurons). Damage to GABAergic neurons was also seen in the 1.0 µm group. PS microplastics upregulated the expression of *gst-4* (encodes glutathione S-transferase-4, a key enzyme involved in oxidative stress).	

Table D-4: Sediment

Organism and Exposure Duration	**Microplastic Type and Concentration**	**Summary of Effects**	**Source**
Ennucula tenuis (Bivalve) *Abra nitida* (Saltwater clam) Four weeks	PE fragments (size ranges of 4–6 µm, 20–25 µm, and 125–500 µm) at 1 mg/kg, 10 mg/kg, and 25 mg/kg dry sediment A low background contamination with perfluorooctane sulfonate was found in microplastics Control group was exposed to clean sediment	No significant changes in mortality, condition index, or burrowing behaviour were seen between treatments in both species. In *E. tenuis*, there were no significant changes in protein and carbohydrate content. However, there was a significant reduction in lipid content (64%) for individuals exposed to 20–25 µm at 10 mg/kg. In addition, a dose-dependent decrease in total energy was evident in all size groups. In *A. nitida*, there was a significant decrease in protein content from individuals exposed to 125–500 µm PE. Apparent, but not significant, changes in lipid content,	Bour et al. 2018

		carbohydrate content, and total energy were seen.	
Perinereis aibuhitensis (Clamworm) Four weeks	PS microspheres (size ranges of 8–12 µm and 32–38 µm) at 100 beads/mL and 1 000 beads/mL (nominal) Control group was exposed to 0.45 µm-filtered seawater without microplastic	The presence of microplastics increased mortality in *P. aibuhitensis*, with 8–12 µm microbeads having a significantly higher effect than the other treatments. For example, exposure to 8–12 µm microspheres at 100 beads/mL led to an average survival of 38% compared to over 80% in the control. Segment regeneration was size-dependent, with the slowest rate being observed in worms exposed to 8–12 µm (smaller size) microspheres at 1 000 beads/mL. Regeneration was 8.3 ± 1.4% for this group, compared to 20.7 ± 2.5% in the control group. In addition, worms exposed to a lower concentration of microplastics displayed a higher percent of segment regenerated.	Leung and Chan 2018
Hyalella azteca (Amphipod) *Asellus aquaticus* (Isopod) *Sphaerium corneum* (Bivalve) *Lumbriculus variegatus* (Worm) *Tubifex* spp. (Worm) 28 days	Irregular PS fragments (20–500 µm) mixed with sediment at 0.1%, 1%, 5%, 10%, 20%, 30% and 40% sediment dw Control group was exposed to sediment without microplastic	In *H. azteca*, *A. aquaticus*, *S. corneum*, and *Tubifex* spp., microplastics had no significant effect on mortality at all test concentrations. In *Lumbriculus variegatus*, no effects were found on reproduction (measured as reproduction factor). No differences in growth were seen in *A. aquaticus*, *S. corneum*, *H. azteca*, *L. variegatus*, and *Tubifex* spp. In *H. azteca*, there were no differences in feeding activity at all concentrations.	Redondo-Hasselerharm et al. 2018

		In *L. variegatus and Tubifex* spp., microplastic exposure had no negative effect on egestion. No microplastics were found in the body and fecal pellets of *H. azteca*.	
Chironomus tepperi (Sediment dwelling midge) Five day growth assay and 10 day emergence assay	Blue/white PE microplastics (size ranges of 1–4 µm, 10–27 µm, 43–54 µm, and 100–126 µm) at 500 particles/kg sediment Control group was exposed to unspiked sediment; additional control assays using moderately hard water with and without Tween-20 (surfactant) were also conducted to ensure that larvae were appropriately sensitive and Tween-20 did not affect results	Using a five-day growth assay, survival rates of midges exposed to microplastics were size-dependent, and the effects were found to be more pronounced with smaller particle sizes. Survival rate was the lowest in the 10–27 µm (57% survival) treatment group in comparison to the control (92% survival). Exposure to the highest tested concentration did not have any significant effect on survival. A size-dependent effect was also seen in larvae growth, where exposure to smaller microplastics led to significant decreases in body length. Exposure to 10–27 µm also led to the smallest body length (7.6 ± 2.4 mm) compared to the control (12.9 ± 3.1 mm). No significant changes were seen for the 100–126 µm group. The length of larvae head capsule was not affected by exposure to any treatment, with the exception of 10–27 µm, which had a significant reduction in mean head capsule length. SEM imaging also revealed reductions in the size of the head capsule and mouth of this group. It is hypothesized that the 10–27 µm particles had the greatest effects since they are	Ziajahromi et al. 2018

		the ideal size for consumption and to avoid egestion. Using a 10-day emergence assay, it was found that exposure to microplastics negatively affected the total number of emerged adults. There was a significant reduction in emergence rate for all microplastic size ranges. For the 10–27 µm group, the emergence rate was only 17.5%, compared to 92% in the control.	
Caenorhabditis elegans (Nematode) Two days	For *C. elegans*, concentrations of 0.5 mg/m^2, 1.0 mg/m^2, 5.0 mg/m^2 and 10.0 mg/m^2 were used For *C. elegans*, nematode growth medium agar seeded with *Escherichia coli* OP50 was used for the control group	In *C. elegans*, PA, PE, PP, and PVC microplastics had significant effects on their survival, with the exception of PVC at 0.5 mg/m^2. PS particles displayed a significant size-dependent effect on lethality, with the 1.0 µm particles causing strong lethality and the 5.0 µm particles causing moderate lethality. In addition, exposure to 5.0 mg/m^2 microplastics led to reductions in average body length and reproduction (embryo number and brood size). Microplastic exposure also led to decreased intestinal calcium levels and increased *gst-4* expression. In *C. elegans*, 1.0 µm PS particles showed the highest toxicity, highest accumulation in the intestines, lowest Ca^{2+} level in the intestine, and greatest expression of *gst-4* of the different sizes tested.	Lei et al. 2018a
Gammarus pulex (Amphipod) 28 days	Irregular PS fragments (20–500 µm) mixed with sediment at 0.1%, 1%, 5%, 10%, 20%, 30% and 40% sediment dw[a]	In *G. pulex*, microplastics had no significant effect on mortality at all test concentrations.	Redondo-Hasselerharm et al. 2018

	Control group was exposed to sediment without microplastic	*G. pulex* had a significant reduction in growth following exposure to high microplastic concentrations (10–40%) compared to controls. The EC_{50}[b] value was determined to be 3.57% sediment dw (±3.22) and the EC_{10}[c] value was 1.07%. There were no differences in feeding activity at all concentrations. In addition, *G. pulex* had microplastics present in the body and fecal pellets at all concentrations following a 24-hour depuration time. Uptake by *G. pulex* was found to be proportional to the concentration of microplastic in the sediment.	

[a] Dry weight
[b] Median effective concentration
[c] 10% effect concentration

Appendix E: Additional information on toxicological studies

Table E-1: Ingestion toxicity studies

Species, Route and Exposure Duration	Microplastic Tested	Concentration	Summary of Effects	Source
Rats Dietary 90 days (7 d/week)	Nonwoven, spunbond polymer fabric made of PE and PET (milled to fine powder) Particle sizes and counts were not reported, although based on typical diameter range of spunbond fibres, particles were likely in the range of 1 to 50 µm (Welle et al. 2018)	Test diet was prepared by mixing ground test material in basal diet at target levels of 0%, 0.5%, 2.5% or 5%	No toxicologically relevant treatment-related effects were observed in any of end points evaluated in the feeding study i.e., no treatment-related adverse effects on blood parameters, organ weights or histopathology of the liver NOEL[a] not identified by authors but can be considered the highest dose, which is equal to 2 500 mg/kg bw/day (assuming 5% food factor for rats) (WHO 2019)	Merski et al. 2008
Mice Oral gavage 28 days (7 d/week)	Fluorescent PS 5 µm and 20 µm in diameter	1.46 x 10^6 items of 5 µm particles at 0.1 mg/day 2.27 x 10^4 items of 20 µm particles at 0.1 mg/day	PS accumulation in the liver, kidney and gut of exposed mice for both 5 µm and 20 µm particle sizes (Translocation to the liver and kidney reportedly occurred and particles could be detected one week after cessation of exposure.)	Deng et al. 2017
Mice Oral gavage	Virgin PS 5 µm and 20 µm in diameter	1 × 10^5 items of 5 µm particles at 0.01 mg/day	Inflammation and lipid droplets were observed in the livers of treated mice at highest dose	Deng et al. 2017

28 days (7 d/week)		2×10^3 items of 20 µm particles at 0.01 mg/day 1×10^6 items of 5 µm particles at 0.1 mg/day 2×10^4 items of 20 µm particles at 0.1 mg/day 5×10^6 items of 5 µm particles at 0.5 mg/day 1×10^5 items for 20 µm at 0.5 mg/day	Incidence or severity data not reported Energy metabolism: Both sizes of PS induced a decrease in ATP level and significant decrease in LDH[b] activity in a dose-dependent matter Lipid metabolism: Decreases in all treatments for the levels of total cholesterol and triglycerides Biomarkers of oxidative stress: Increased GPx activity (more so in 5 µm group) and SOD; Decrease in catalase activity in almost all the treatment groups Potential for neurotoxicity: Decreased acetylcholinesterase activity in liver after exposure to two sizes of PS microplastics, but more so in 5 µm group	
Mice Oral gavage 28 days (three times /week)	PS 1 µm, 4 µm and 10 µm in diameter	Mixture of 1 µm (4.55×10^7 particles), 4 µm (4.55×10^7 particles), and 10 µm (1.49×10^6 particles) PS in CMC[c] at a volume of 10 mL/kg/bw	No evidence of occurrence of inflammation and/or oxidative stress following exposure of mice to PS microparticles Little presence of particles in cells of the jejunum and duodenum No particles were found in other organs (liver, spleen and kidney)	Stock et al. 2019

Mice Drinking water Six weeks (continuous exposure)	Virgin and fluorescent PS 5 µm in diameter	1.456×10^6 particles/L of 5 µm particles at 100 µg/L 1.456×10^7 particles/L of 5 µm particles at 1 000 µg/L	Accumulation of 5 µm PS in gut with 1 000 µg/L exposure Gut microbiota dysbiosis (change in the composition of the gut microbiota in the cecal contents of the mice) at both doses Intestinal barrier dysfunction Alterations in amino acid and bile acid metabolism with 1 000 µg/L exposure	Jin et al. 2019
Mice Drinking water Five weeks (continuous exposure)	PS 0.5 µm and 50 µm in diameter	1.456×10^{10} particles/L of 0.5 µm at 100 µg/L in drinking water 1.456×10^{10} particles/L of 0.5 µm at 1 000 µg/L in drinking water 1.456×10^4 particles/L of 50 µm at 100 µg/L in drinking water 1.456×10^4 particles/L of 50 µm at 1 000 µg/L in drinking water	Altered hepatic lipid metabolism Altered gut microbiota composition	Lu et al. 2018
Mice Drinking water 90 days (continuous exposure)	PE and organo-phosphorus flame retardants (OPFRs) (TCEP and TDCPP) or PS and OPFRs	2 000 µg/L PS (3.7×10^8 items/L) and 10 µg/L OPFRs 2 000 µg/L PS (3.7×10^8 items/L) and 100 µg/L OPFRs 2 000 µg/L PE (3.7×10^8 items/L) and 10 µg/L OPFRs	Increased oxidative stress, increased neurotoxicity, enhanced disruption of amino acid metabolism and energy metabolism from co-exposure No microplastic-only control group; it is unclear what component of the treatment contributed to the effects	Deng et al. 2018

		2 000 µg/L PE (3.7 x 10^8 items/L) and 100 µg/L OPFRs		

[a] No observed effect level
[b] Lactate dehydrogenase
[c] Carboxymethylcellulose

Table E-2: Inhalation toxicity studies

Species, Route and Exposure Duration	Microplastic Tested	Concentration	Summary of Effects	Source
Rats Nose-only inhalation 90 days (6 h/d, 5 d/wk)	PP fibres (GM[a] diameter of 1.2 µm and length of 11.6 to 14.7 µm)	13.0, 28.1 or 59.6 mg/m^3 (12.1, 20 or 48.1 fibres/cm^3)	Dose-related increase in incidence and severity of fibre-containing macrophages and microgranulomas, with bronchiolization at high concentration. Reversible at two lower concentrations. LOEC[b] = 13 mg/m^3 $LOEC_{adj}$[c] = 2.3 mg/m^3	Hesterberg et al. 1992
Rats Inhalation in chamber air 12 weeks (6 h/d, 5 d/wk)	Freshly generated PUF particulates (94% <5 µm and 83% <3 µm)	8.65 mg/m^3	No effect on body weight, survival time, behaviour or tumour incidence. Intra-alveolar granulomas and peribronchial and perivascular lymphocyte infiltration. LOEC = 8.65 mg/m^3 $LOEC_{adj}$ = 1.54 mg/m^3	Thyssen et al. 1978
Rats Inhalation in chamber air 30 exposure days (6 h/d, 5 d/wk)	Freshly generated PUF particulates (median diameter 0.7 µm)	3.6, 20.5 mg/m^3	No effect on mortality or weight. Hemorrhage congestion edema at high concentration. No increase in pneumonitis or lymphocytic infiltration. Dose-related increase in tracheal	Laskin et al. 1972

			hyperplasia. Increase in tracheal squamous metaplasia only at low concentration. Increase in bronchio-alveolar changes (centrilobular emphysema and macrophages) only at high concentration. No increase in bronchial hyperplasia or squamous metaplasia. Lung and lymph macrophages contained particles. Squamous cell carcinoma observed in 1 rat in each treatment group. LOEC = 3.6 mg/m^3 $LOEC_{adj}$ = 0.64 mg/m^3	
Hamsters Inhalation in chamber air 30 exposure days (6 h/d, 5 d/wk)	Freshly generated PUF particulates (median diameter 0.7 µm)	3.6, 20.5 mg/m^3	No increase in mortality. Weight loss only at low concentration. Hemorrhage congestion edema at high concentration. No increase in pneumonitis or lymphocytic infiltration. Histological changes limited to bronchial hyperplasia. LOEC = 3.6 mg/m^3 $LOEC_{adj}$ = 0.64 mg/m^3	Laskin et al. 1972
Rats Nose-only inhalation Four weeks (20 exposure days), 6 h/d, 5 d/wk	Uncoated nylon fibre-shaped particulates (mean length and diameter of 9.8 and 1.6 µm, respectively)	4.0, 15 and 57 fibres/cm^3 (0.6, 2.7 and 19.6 mg/m^3)	No effect on body weight, lung weight, or clinical observations. Reversible increase in total cell counts in BALF in 57 fibres/cm^3 group (with an	Warheit et al. 2003

			increase in neutrophil fraction). Absence of evidence of pulmonary inflammation, biomarkers of lung injury, and cell proliferation. Nylon particulates contained in BALF and nasal lymphoid macrophages; higher and more persistent at high concentration. No impact on phagocytic abilities of macrophages. No significant changes in cell proliferation rates. NOEC[d] = 15 fibres/cm^3 (2.7 mg/m^3) $NOEC_{adj}$[e] = 2.7 fibres/cm^3 (0.48 mg/m^3)	
Guinea pigs Inhalation in chamber air 325 days	Nylon and Orlon (PAN) particulates (dimensions not stated)	2 g pulverized 3 times/d	Nodular subpleural foci within areas of emphysema in interalveolar septa. Foci consisted of edema, reticular fibres, and granulomas containing histiocytes and fibroblasts. Lesions contained inhaled particles. LOEC = 6 g/day	Pimentel et al. 1975
Rats Nose-only inhalation Five days (6 h/day)	Acrylic ester copolymer, with and without a nanoparticle fraction (MMAD[f] of 1.2 µm and median diameter of 0.4 µm for both test	3.4 and 10.6 mg/m^3 for both test compounds	No treatment-related effect on body weight, clinical observations, hematological parameters, BALF parameters (total	Ma-Hock et al. 2012

	compounds, but size distribution varied in the two aerosol types)		and differential cell counts or biochemical indicators of lung injury) or lung and lymph node histology. NOEC = 10.6 mg/m^3 $NOEC_{adj}$ = 2.7 mg/m^3	
Rats Intratracheal Single instillation	PVC particulates (<5 µm)	25 mg suspended in 1 mL saline	No effect on mortality. Reversible increase in activity of lung succinic dehydrogenase and adenosine triphosphatase and lysosomal enzymes. Vascular and inflammatory changes, hyperplasia, interstitial fibrosis, and granulomas in areas of lungs corresponding to particulate deposition; effects were reversible as particulate was cleared. LOEL[g] = 25 mg	Agarwal et al. 1978
Rats Intratracheal Single instillation	PVC particulates as suspension or emulsion (various groups with mass median diameters ranging from 13 to 130 µm); one group exposed to a copolymer with vinyl acetate	2 mg in 0.2 mL saline	Small foci of granular material with mild inflammation, in alveoli and alveolar ducts. No fibrosis; no lymphatic changes. LOEL = 2 mg	Pigott and Ishmael 1979
Rats Intratracheal Single instillation	PVC particles as produced or washed (to remove adsorbed additives); median size of about 2 µm	10 or 50 mg/kg	No effect on body weight. BALF: elevated LDH, total protein level, total cell count, and neutrophils in 50 mg/kg groups at 2 and 7 days after instillation, but	Xu et al. 2004

			decreased to control levels at later timepoints; most effects persisted longer in washed PVC than non-washed. Histology: High-dose PVC groups had thickened aveolar walls accompanied by clusters of inflammatory cells and particles at 2 days post-instillation, with increased inflammation at 7 days in the washed PVC. Foci on lung surface at 28 days (less obvious at 90 days) and increase in macrophages (with no fibrosis) at 90 days.	
Rats Intratracheal Single instillation	Milled nylon tow (i.e., uncut nylon strands) (average width and length of 2 µm and 14 µm, respectively)	10 mg/kg bw of particulates in saline	Significant increase in breathing rate. Suppurative pneumonia around bronchioles; histiocytic inflammation in alveoli near fibres; no fibrosis. Significant increase in polymorphonuclear leukocyte cell count in BALF. Significant increase in chemiluminescence but not cell count for alveolar macrophage. Significant increase in albumin (indicator of blood-gas barrier	Porter et al. 1999

			deterioration) and matrix metalloprotease activity (indicator of inflammation). LOEL = 10 mg/kg	
Rats Intratracheal Single instillation	PS microspheres (64, 202 or 535 nm)	1 mg in 0.5 mL saline	BALF: Significant increase in total cells for 64 and 202 nm particles. Increase in protein in 64 and 535 nm particles, and increase in LDH activity (suggestive of cell death) in 64 nm particles. LOEL = 1 mg	Brown et al. 2001
Rats Intratracheal Single instillation	PU particles from aged (PUF I) or freshly-prepared (PUF II) foam (aerodynamic diameter of ≤10 µm for 93.5% of particles and ≤5 µm for 52% of particles)	20 mg/mL in saline	Early lymphocytic infiltration and macrophage activity in lungs, later accompanied by alveolar wall thickening, epithelization, and fibrosis, which at 18 and 24 months progressed to scarring and perifocal emphysema. Hyperplasia in bronchial epithelium and benign intrabronchial adenomas from PUF II.	Stemmer et al. 1975
Pregnant rats Intratracheal Repeat dose: instillation every other day, GD[h] 5 to 19	20 nm PS	2 974 µg total (equivalent to 952 µg/dose); 2.4×10^{13} particles. In 300 µL saline	Significant increase in reabsorption sites in exposed rats (both acute and repeat). Evidence of particle translocation from lung: repeat study – placenta, whole pup, and fetal liver; acute study – maternal heart, spleen,	Fournier et al. 2018 (abstract only; no full-text)

Acute: single instillation on GD 19			placenta, fetal heart, fetal liver, and whole pup.	

[a] Geometric mean
[b] Lowest observed effect concentration
[c] Lowest observed effect concentration, adjusted for continuous exposure
[d] No observed effect concentration
[e] No observed effect concentration, adjusted for continuous exposure
[f] Mass median aerodynamic diameter
[g] Lowest observed effect level
[h] Gestational day

Economic Study of the

CANADIAN PLASTIC INDUSTRY, MARKETS AND WASTE

Summary Report to Environment and Climate Change Canada

Environment and Climate Change Canada
Environnement et Changement climatique Canada

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Disclaimer

The assumptions and parameters used in the plastics waste management value chain modelling are based on a review of literature, industry reports and national statistics, as well as consultations completed with industry stakeholders. The Canadian Plastics Industry Association (CPIA) and the Chemistry Industry Association of Canada (CIAC) were consulted to ensure representation of the plastic resin industry. Stewardship organizations such as the Canadian Stewardship Services Alliances (CSSA) and Éco Entreprises Québec were consulted to gather information on residential packaging plastic waste collection and associated costs. Several provincial ministries, government agencies such as RECYC-QUÉBEC, and industry associations were consulted to inform the current state of recycling within their sector or region. To the extent possible, information gathered was cross-checked with additional sources of information such as data from Statistics Canada Waste Management Information Survey (WMIS) and reports such as the 2016 Post-consumer Plastics Recycling in Canada report from More Recycling (More Recycling, 2018)*. For greenhouse gas emissions life cycle data from previous studies conducted in Europe and from recognized lifecycle databases has been leveraged to provide greenhouse gas emissions factors for key steps of the value chain.*

Given the national scope of the study, the complexities of interactions between sector- and resin-level analysis, and the limited timespan within which this study was conducted, limitations and uncertainties remain in the results presented in these reports. First, the model developed by the authors to build the 2016 baseline and 2030 scenario projections does not reflect the specificities of all products containing plastics, given that a key source of information, the Supply and Use Table from Statistics Canada, was built using a limited number of product categories (286 product categories within the Canadian economy). Second, the model does not reflect all possible feedback loops (e.g. re-use/repair impact on actual new product demand). Third, imports and exports of sorted plastic wastes were excluded from the models used for the 2016 baseline (as it was difficult to allocate imports and exports to specific resins or sectors given available statistical data) and for the 2030 projections (as it was difficult to forecast import/export evolution). Finally, the recycling rates presented in this study are measured in relation to the output of recyclers in Canada, after factoring in all intermediate losses (sorting and reprocessing).

Consequently, numerical values appearing in this report represent average value estimates and should only be interpreted as such. The actual values of a specific product within a given product category might be different (higher or lower) and therefore no specific product or sector conclusion should be made without consideration of this limitation and undertaking additional research procedures.

Minor discrepancies may occur between stated totals and the sums of component items, as totals are calculated using component item values prior to rounding. Minor discrepancies between summary tables and figures presented may occur, in particular between task reports as their supporting methodology differed, in line with the overall goal of their respective tasks. General alignment has, however, been confirmed, with a few exceptions at intermediary steps of the recycling value chain. Assumptions and calculations have been made as transparent as possible to enable the future refinement of the model once new specific data points and research become available.

Executive summary

A unique view on plastics in Canada

ECCC commissioned this Economic Study of the Canadian Plastic Industry, Markets and Waste in July 2018. The scope of the study, encompassing most plastics types used across all key sectors, is a unique attempt to shed light on the entire plastics value chain in Canada, from raw material production and products manufacturing to use and end-of-life.

The authors leveraged a wide selection of primary and secondary sources to complete the four task reports that constitute the backbone of the results presented in this summary report (Deloitte, 2019a) (Deloitte, 2019b) (Deloitte, 2019c) (Deloitte, 2019d). In addition to national statistics, the authors reviewed over 220 documents and industry reports and conducted more than 130 interviews.

This report first presents an overview of the plastics value chain beginning with raw material production (virgin plastics resins) before moving into plastics products manufacturing and their end-use in key sectors, and concluding with an analysis of their end-of-life management. The report then describes 2030 scenarios, highlighting potential paths for the plastics value chain, in particular relating to end-of-life performance. The report then presents a high-level economic, environmental and social impact assessment to discuss the scenarios and their feasibility. Finally, the report introduces a review of policy measures that could be implemented to support the growth of the secondary plastics markets in Canada.

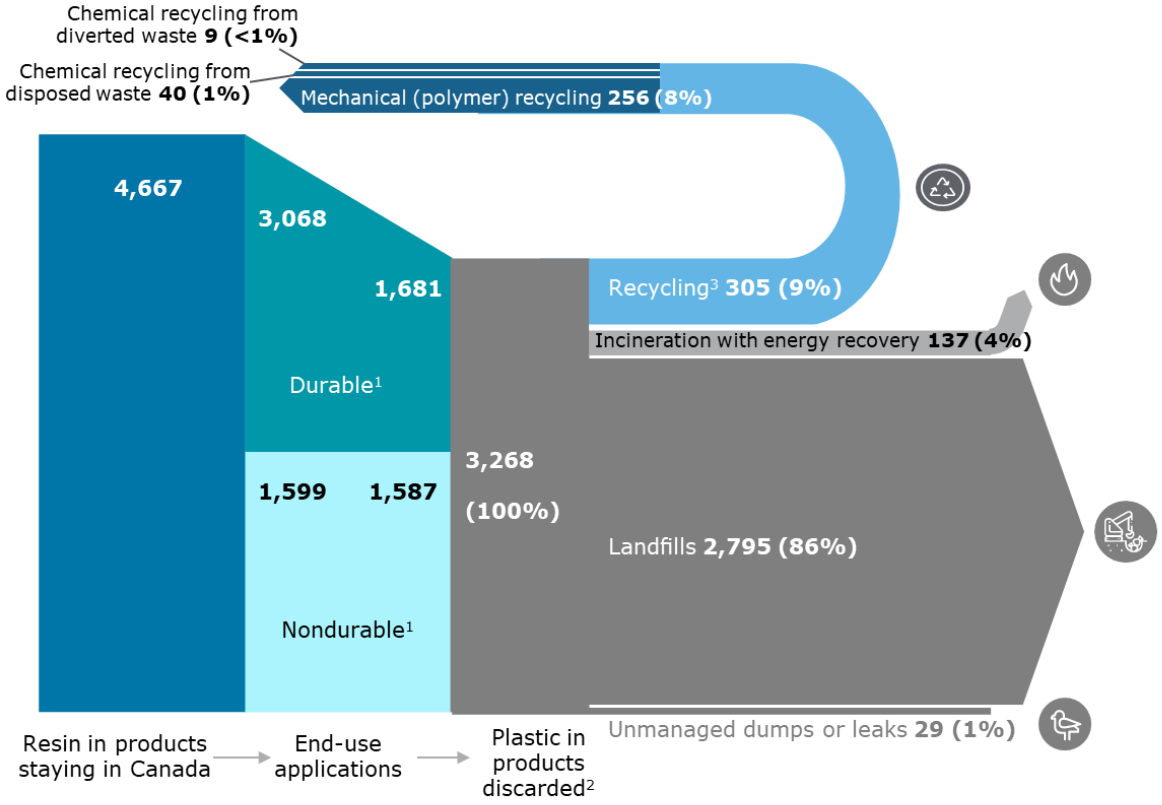
Plastics resins and products: CA$35 billion in sales in Canada

With total sales estimated at CA$35 billion, plastic resin (CA$10 billion) and plastic product (CA$25 billion) manufacturing in Canada accounts for over five percent of the sales in the Canadian manufacturing sector, and employs 93,000 people across 1,932 establishments. Present in almost every modern product, global demand and production of plastics is growing. In Canada, plastic products are in demand in most sectors of the economy, with approximately 4,667 kilotonnes (kt) of plastics introduced to the domestic market on an annual basis (more than 125 kg per capita). Three categories (packaging, construction and automotive) show a particular appetite for plastic, accounting for 69 percent of plastic end-use.

Canada's CA$7.8 billion lost opportunity: 87 percent of plastics waste ends up in landfills or the environment

The Canadian plastics economy is mostly linear, with an estimated nine percent of plastic waste recycled, four percent incinerated with energy recovery, 86 percent landfilled, and one percent leaked into the environment in 2016 (Figure 1). Thus, plastics material not recovered (i.e., 2,824kt of resins sent to landfill or leaked into the environment) represented a lost opportunity of CA$7.8 billion for Canada in 2016, based on the value of virgin resin material.

Figure 1: Canadian resin flows in thousands of tonnes per annum, 2016



[1] Durable applications with an average lifetime >1 year will end up as waste only in later years; given market growth and increase share of plastics in durable applications (e.g., construction, cars) plastics waste generated today is less than what is being put in the market that same year. On the contrary nondurable applications go almost straight to waste.

[2] 1,587 thousand metric tons of mixed plastic waste from nondurable applications plus 1,681 thousand metric tons of mixed plastic waste from production in previous years.

[3] Output recycling rate, after taking into account process losses.

The main generating sectors for plastic waste are packaging (47 percent of total plastic waste), automotive (9 percent), textiles (7 percent), and electrical and electronic equipment (EEE 7 percent). The construction sector, while an important end-use market (accounting for 26 percent of plastic put on the market), is not yet a large plastic waste generator (5 percent), given the fairly recent incorporation of plastics in construction (in the 1980s and 90s) that remains 'stocked' in houses and buildings; this situation could change in future years with construction renewal. Under a business as usual situation, the linear profile of the Canadian plastics economy is not going to improve given forecasted trends in waste streams and economic drivers.

By 2030, it is estimated that Canada's lost opportunity related to unrecovered plastics could rise to CA$11.1 billion, under a business as usual scenario following the same end uses and value recovery performance as the current baseline (Figure 1).

Given current market prices, structures, business models and the low cost of disposal, there is limited direct economic incentive for plastics recycling and value recovery in Canada

Domestically recycled "secondary" plastics output accounted for approximately CA$350 million in sales in Canada in 2016. In comparison with the sales of its primary resin competitor, it is 30 times smaller. The recycling industry focuses on polyethylene terephthalate (PET), high-density polyethylene (HDPE) and polypropylene (PP) and is predominantly located in large end-markets providing easier access to plastic waste feedstock, such as in Ontario, Quebec and British Columbia.

The Canadian virgin "primary" resin domestic output accounts for CA$10 billion annually and is driven by global oil prices and investment in large scale industrial facilities in locations allowing access to advantaged petrochemical feedstock, such as in Alberta or Ontario. Canadian virgin resin production focuses on high-volume resins such as polyethylene. The virgin resin industry has a very high international trade exposure, with 77 percent of its output exported, and 71 percent of the domestic resin demand fulfilled through imports. The United States (US) is the main trading partner, accounting for more than 80 percent of import and export of the industry.

Primary and secondary plastics compete against each other in the same market, based on price and quality of the resins. This competition is difficult for the recycling industry, which struggles with quality due to uneven feedstock composition, and on prices. Secondary plastics producers enjoy lower upfront investment than their virgin competitors do; however, during periods of low oil prices which bring downward prices for virgin resins, secondary resins producers are more exposed than their virgin counterparts as their cost structure is more labor-intensive. This is one reason why many secondary plastics producers ceased operations in 2016 in North America, as oil prices were low.

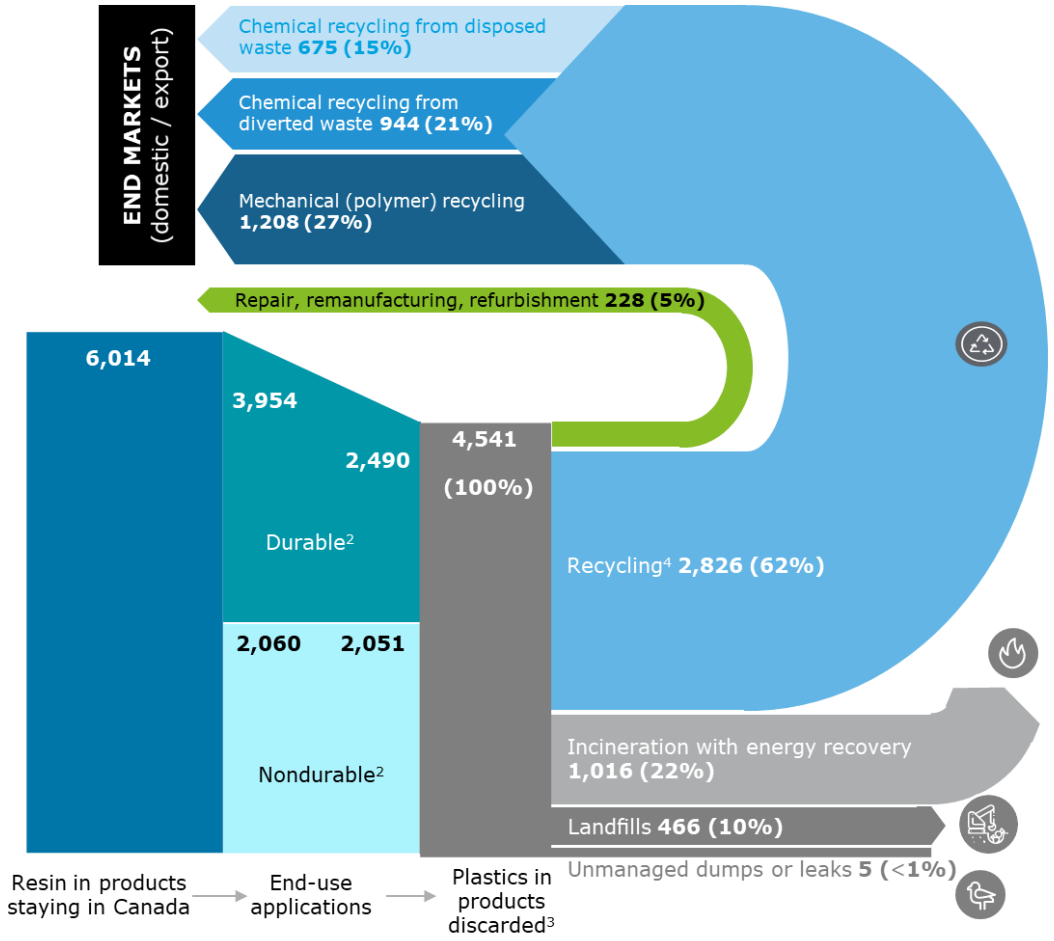
Overall, value recovery options are only as strong as their weakest link in the value chain and face competition from low-cost alternatives such as landfilling. Key barriers to the recovery of plastics include a combination of factors, such as: low diversion rates (only 25 percent of all plastics discarded are collected for diversion); process losses in the sorting (e.g., shredded residues containing plastic sent to landfill) and reprocessing stages; and the near-absence of high volume recovery options for hard-to-recycle plastics (e.g., plastics waste coming from the white goods, EEE or automotive sectors).

Mechanical recycling, which is currently the dominant value recovery option, only reprocessed eight percent of total plastics waste in Canada in 2016. Economic incentives are still limited, coupled with other factors including collection and processing costs, poor product design, and low participation in recycling programs. Several chemical recycling technologies exist that could allow the market to process monomers, petrochemical feedstock or fuels; however, these technologies require further investment to confirm their full-scale commercial viability in the Canadian plastic waste context.

A zero plastic waste economy would deliver significant benefits to Canada

An ambitious 2030 scenario was developed to model the potential costs and benefits of achieving zero plastic waste (2030_{T90}). This scenario used a 90% landfill diversion rate as a proxy for zero plastic waste and assumed that: i) plastics production and end use applications increased but followed the same patterns as in 2016, ii) mechanical recycling was quadrupled from its business as usual level; iii) chemical recycling was significantly scaled up, taking into account readiness levels and associated learning curves and iv) energy from waste was leveraged to deal with the remaining volumes and hard-to-recycle plastics.

Figure 2: Canadian resin flows in thousands of tonnes per annum, 2030_{T90} scenario1



[1] Scenario based on a multi-stakeholder push to boost recycling, including investment in new facilities, regulatory measures to encourage recycling, significant progress on technologies and favorable end-markets demand.

[2] Durable applications with an average lifetime >1 year will end up as waste only in later years; given market growth and increase share of plastics in durable applications (e.g., construction, cars) plastics waste generated today is less than what is being put in the market that same year. On the contrary nondurable applications go almost straight to waste.

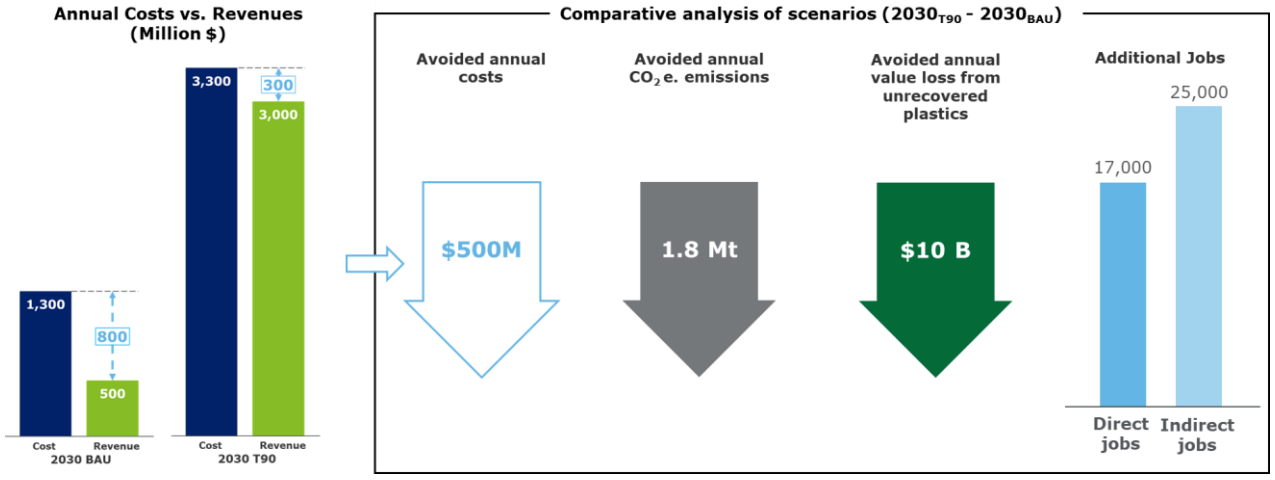
[3] 2,051 thousand metric tons of mixed plastic waste from nondurable applications plus 2,490 thousand metric tons of mixed plastic waste from production in previous years.

[4] Output recycling rate, after taking into account process losses

This scenario is not a prediction or a recommendation: it is an illustration of what zero plastic waste could look like given current product designs and emerging value recovery technologies. Changes in plastic production and design would open the door to higher value recycling and recovery options.

However, even without such changes, a preliminary comparative analysis (Figure 3) shows that 2030_{T90} would deliver significant benefits to Canada in comparison to business as usual (2030_{BAU}): CA$500 million of annual costs avoided, 42,000 direct and indirect jobs created, and annual greenhouse gas emissions savings of 1.8Mt of CO_2e.

Figure 3: Comparative analysis of scenarios



This analysis indicates that zero plastic waste cannot be achieved without concurrent, strategic interventions by government, industry stakeholders and the public across each stage of the plastic lifecycle and targeted at sectors

Business-as-usual or incremental changes are not an option to reach the target and the modelled 90 percent plastic waste recovery. Achieving 90 percent plastic waste recovery will require significant investment to diversify and expand the capacity of current value recovery options including mechanical recycling as the most mature technology, but also chemical recycling and waste-to-energy. It will also require significant improvements to current plastic waste diversion rates, which vary depending on sector specific approaches. An international benchmark demonstrated the need for a systemic approach, acting in several areas simultaneously, as no single public or private sector action can shift the system.

Five sets of interventions (e.g. policies, measures and calls-to-action) were identified as having been effective in other jurisdictions and could be used to achieve zero plastic waste in Canada:

Set 1: Create viable, domestic, secondary end-markets

- Create stable, predictable demand for recycled plastics that is separate from virgin markets (e.g., requirements for recycled content, taxes/fees on virgin resins)
- Improve the quality of recovered plastics at both the point of collection and in materials processing
- Improve access to domestic supply of recycled content
- Support innovation in product designs and uses for secondary plastics

Set 2: Get everybody onboard to collect all plastics

- Create sector-specific requirements for collection (e.g., extended producer responsibility, performance agreements)
- Restrict disposal (e.g., landfill taxes or bans)
- Require/incentivize collection (e.g., industry targets, deposit refund)
- Develop more consistent requirements and rules across Canada (e.g., common curbside recycling)
- Improve public information on collection and recyclability

Set 3: Support and expand all value-recovery options

- Support development of innovative value-recovery options, such as advanced mechanical and chemical recycling
- Focus primarily on improving mechanical recycling
- Increase the ease and speed at which new value recovery facilities can be developed by removing policy barriers and investing in innovation

Set 4: Increase efficiency throughout the value chain

- Facilitate collection and value-recovery by creating requirements for the reusability and recyclability of product design (e.g., standards and public procurement)
- Improve performance by investing in sorting and separation
- Educate and engage actors and consumers throughout the value chain

Set 5: Extend plastics lifetime to reduce and delay waste generation

- Leverage opportunities to extend the lifetime of durable goods, which account for approximately 51 percent of total plastics waste, but have a very low recycling rate (two percent) compared to that of non-durable goods (15 percent)
- Introduce measures that contribute to increased reuse, repair and remanufacturing (in particular with higher value durable goods such as EEE or white goods) such as standard requirements for reparability or reusability, and tax exemption to reduce and delay waste generation from durable goods in Canada

Table of contents

List of acronyms

ECCC	Environment and Climate Change Canada
EEE	Electrical and Electronic Equipment
EPR	Extended producer responsibility is a policy approach under which producers are given a significant responsibility – financial and/or physical – for the treatment or disposal of post-consumer products
EPRA	Electronic Products Recycling Association
HDPE	High Density Polyethylene
HS	Harmonized System codes
ICI	Industrial, Commercial, and Institutional sector
kt	kilotonne
LCA	Life Cycle Assessment
LDPE	Low Density Polyethylene
MRF	Material Recovery Facility
MSW	Municipal Solid Waste
PA	Polyamide
PE	Polyethylene
PET	Polyethylene Terephthalate
POP	Persistent Organic Pollutant
PP	Polypropylene
PS	Polystyrene
PU / PUR	Polyurethane
PVC	Polyvinyl Chloride
RRRDR	Remanufacturing, Refurbishment, Repair and Direct Re-use
StatCan	Statistics Canada
SUT	Supply and Use Table

Glossary of terms

Chemical recycling	Chemical recycling can be defined as a process changing the chemical structure of plastic waste, converting it into shorter molecules, ready to be used for producing new plastics or fuels
Depolymerisation	Depolymerisation refers to chemolytical processes that break down polymers and produce mainly the monomers from which they have been produced or other oligomers (short chains of monomers). These can then be used as building blocks for the production of new polymers
Diversion rate	See R1/COLL in Section Model parameters
Feedstock	Any bulk raw material that is the principal input for an industrial production process
Leakage	Materials that do not follow an intended pathway and 'escape' or are otherwise lost to the system. Litter is an example of system leakage.
Mechanical recycling	Operations that recover after-use plastics via mechanical processes (grinding, washing, separating, drying, re-granulating, compounding), without significantly changing the chemical structure of the material
Output recycling rate	See R3/COLL in Section Model parameters
Recycling	A general term covering the process chain of collection, sorting, reprocessing of end-of-life materials into raw material that can be used as an input into new product manufacturing
Remanufacturing	Remanufacturing and comprehensive refurbishment are intensive, standardized industrial processes that provide an opportunity to add value and utility to a product's service life
Repair, refurbishment and arranging direct use	Repair, refurbishment and arranging direct use are maintenance processes that typically occur outside of industrial facilities and provide an opportunity to extend the product's useful life
Reprocessing yield	See R3/R2 in Section Model parameters
Resin	A natural or synthetic solid or viscous organic polymer used as the basis of plastics, adhesives, varnishes, or other products
Re-use	Action or practice of using something again, whether for its original purpose or to fulfill a different function
Reverse logistics	Process of moving goods from their typical final destination for the purpose of capturing their value, or for their proper disposal
Sorting	Waste sorting is the process by which waste is separated into different elements. In the context of this study, it refers to the separation of plastic material in recovery (or "sorting") facilities
Sorting yield	See R2/R1 in Section Model parameters
Value recovery rate	Share of plastic that is ultimately value recovered whether through chemical or mechanical recycling from diverted and disposed waste or through thermal recovery, divided by plastics in waste collected. This rate is equal to (R3+D-CHEM+D-EFW)/COLL (see Section Model parameters)

White goods	In this study, white goods refer to appliances (large or small), which are machines in home appliances used for routine housekeeping tasks such as cooking, washing laundry, or food processing and preservation

Model parameters

COLL	Plastics in waste collected, either to be sent to a sorting facility (R1) or to disposal (D1) (Deloitte, 2019a)
D	Total plastics in waste sent to disposal. Some recovery can occur whether through chemical recycling (D-CHEM) or incineration with energy recovery (D-EFW). The rest either is incinerated without energy recovery (D-INC) or landfilled (D-LANDF) (Deloitte, 2019a)
D1	Plastics in waste sent to disposal (Deloitte, 2019a)
D2	Plastics in waste sent to disposal by MRFs. Represents the fraction rejected by the sorting facilities (Deloitte, 2019a)
D3	Plastics in recycling waste sent to disposal. Represents the fraction rejected by the recyclers (Deloitte, 2019a)
D-CHEM	Chemically recycled plastic from disposed waste (Deloitte, 2019c)
D-EFW	Plastics in disposed waste incinerated with energy recovery (Deloitte, 2019c)
DELT	The in-use delta measures the difference between the plastic products generation for a given product category in a given year and the estimated plastic waste generation of that same product category for the same year, before taking into account any additional re-use (see R-DELT below) (Deloitte, 2019a)
D-INC	Plastics in disposed waste incinerated without energy recovery (Deloitte, 2019c)
D-LANDF	Plastics in disposed waste sent to landfill (Deloitte, 2019a)
E2	Plastics in bales and sorted waste exported (Deloitte, 2019c)
GEN	Quantity of plastics in products generated in Canada (Deloitte, 2019a)
I2	Plastics in bales and sorted waste imported (Deloitte, 2019c)
LEAK	Plastics leaked permanently into the environment (Deloitte, 2019a)
QUANT	Quantity of plastics discarded, represents the plastic entering waste streams (Deloitte, 2019a)
R1	Plastics in waste diverted and sent to domestic MRFs (Deloitte, 2019a)
R1/COLL	Diversion rate, or the share of plastic diverted from direct disposal and sent to a sorting facility, divided by COLL. This rate is assessed per sector (Deloitte, 2019a)
R2	Plastics in bales and sorted waste sent to domestic recyclers (Deloitte, 2019a)
R2/COLL	Output sorting rate, or the share of plastic sorted by sorting facilities and sent to a reprocessing facility, divided by COLL. This rate is assessed per sector (Deloitte, 2019a)
R2/R1	Sorting yield, or the amount of plastics MRFs were able to sort out and send to reprocessing facilities, divided by the total amount of unsorted plastic received. This yield is affected by the quality of input waste material, contamination, type of plastics received, sorting technologies and equipment etc. It illustrates the efficiency of the sorting operations, and is assessed per waste stream category or sector (Deloitte, 2019a)
R3	Recycled plastic from diverted waste (Deloitte, 2019a)
R3/COLL	Output recycling rate, or the share of plastic that is ultimately reprocessed whether through chemical or mechanical recycling from diverted waste, divided by COLL. This rate does not include D-CHEM (Deloitte, 2019a)

R3/R2	Reprocessing yield, or the amount of recycled materials (flakes or pellets of recycled resins, monomers etc.) reprocessing facilities were able to produce and send to end-users, divided by the total amount of sorted plastics waste received from MRFs. It illustrates the recycling efficiency of the reprocessing operations, and is assessed per resin and technology (chemical or mechanical) (Deloitte, 2019a)
R3-CHEM	Chemically recycled plastic from diverted waste (Deloitte, 2019c)
R3-MECH	Mechanically recycled plastic from diverted waste (Deloitte, 2019a) (Deloitte, 2019c)
R-DELT	Direct re-use is a way to extend the expected end-of-use of products by a certain amount of time. As such, the re-use delta models the fact that a re-used product enters the waste stream later than an average non-re-used product (Deloitte, 2019a)
RRR	Plastics in repaired, remanufactured and refurbished products. Remanufacturing and comprehensive refurbishment take place within industrial or factory settings and result in quasi-new products, with a full-service life identical to a new product for which production can thus be avoided (Deloitte, 2019a) (Deloitte, 2019b)

List of tables in the report

List of figures in the report

1. The case for a zero plastic waste Canada

1.1 Plastic waste, a triple bottom-line challenge

Plastics are part of the everyday lives of most Canadians. Since the 1950s, global plastics production has increased more than any other manufactured material due to their low cost, durability and utility. However, the current ways in which plastics are managed throughout their lifecycle is threatening ecosystems, human health and livelihoods, and costing billions of dollars a year in lost economic value and other damages. In addition, the amount of plastic designed to be used once and then thrown away leads to a significant waste of resources and energy.

1.2 Canada is taking action

The Government of Canada has committed to work with its partners to move towards zero plastic waste with a vision of keeping all plastics in the economy and out of landfills and the environment. This represents an opportunity to grow Canada's economy while protecting the environment and reducing plastic waste, marine litter and greenhouse gas emissions.

1.3 Purpose of this report

Environment and Climate Change Canada commissioned this study to characterize plastic production, use and management in Canada and to identify the potential benefits, impacts, challenges and opportunities of transitioning to a zero plastic waste economy.

1.4 Scope and limitations of this report

This study is the first of its kind in Canada, presenting an entire lifecycle view (from production of virgin resins to the end-of-life of plastic waste) of most key plastics, both thermoplastics and thermosets. Thermoplastics are plastics that can be heated, cooled and reshaped repeatedly, while thermosets are plastics that can only be shaped once because their polymerization creates a three-dimensional network that cannot be remelted or solubilized.

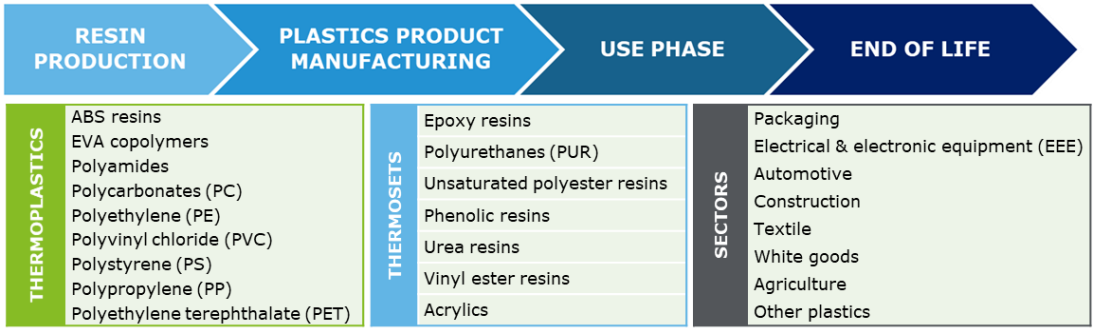
The lifecycle of plastics in the Canadian economy was broken down into four stages: resin production, plastic product manufacturing, use phase and end of life.

For each stage, a baseline economic assessment was conducted, looking at domestic production, import and export.

The various plastics products produced or traded in the Canadian economy were grouped into eight end-use sectors, defined for the purpose of this study: packaging, construction, automotive, electrical and electronic equipment, textiles, white goods, agriculture and other plastics. Together, these products covered an estimated 88 percent of plastics contained in products reaching the Canadian market annually.

Figure 4 illustrates the scope of the study from a lifecycle, resin and sector point of view.

Figure 4: Lifecycle stage, resins and sectors included in the scope of this study

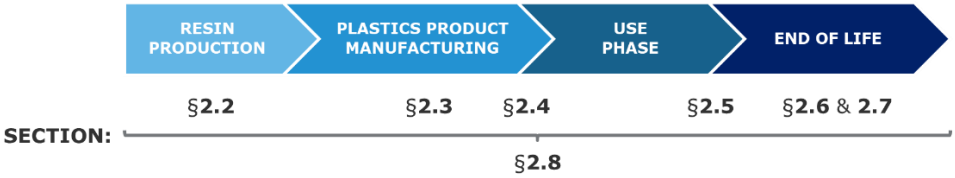


Unless stated otherwise, 2016 is the baseline year for the data presented in this report.

Scenario projections were also made for 2030, based on the situation in 2016 and several assumptions. An overview of the methodology followed to produce this study, as well as definitions of sectors and terms used is provided in Section 5 of this report.

2. Canada throws away 87 percent of plastics, valued at CA$7.8 billion

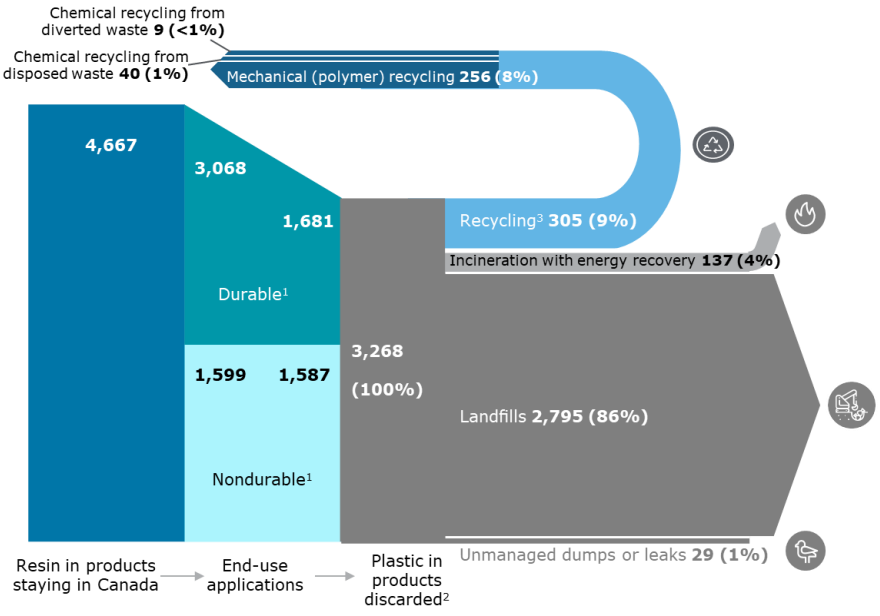
This section presents the key takeaways of the lifecycle of plastics in the Canadian economy. While Section 2.1 introduces the key findings from the overall lifecycle, Sections 2.2 to 2.7 explore in more detail the specific life cycle stages, and Section 2.8 concludes with end-use sector specificities.



2.1 The Canadian plastics economy is designed to be linear and to throw away plastic

In 2016, an estimated 3,268 kilotonnes (kt) of plastics were discarded as waste in Canada, out of the 4,667kt of plastics introduced to the market through both domestic and imported products. Only nine percent of these plastics were ultimately recycled (mechanically or chemically) and four percent were incinerated for energy recovery. The rest was landfilled (86 percent) or lost to the environment (unmanaged dumps or leaks; 1 percent), representing a value loss of CA$7.8 billion, based on the original value of the raw material (Figure 5).

Figure 5: Canadian resin flows in thousands of tonnes per annum, 2016



[1] Durable applications with an average lifetime >1 year will end up as waste only in later years; given market growth and increase share of plastics in durable applications (e.g., construction, cars) plastics waste generated today is less than what is being put in the market that same year. On the contrary nondurable applications go almost straight to waste.

[2] 1,587 thousand metric tons of mixed plastic waste from nondurable applications plus 1,681 thousand metric tons of mixed plastic waste from production in previous years.

[3] Output recycling rate, after taking into account process losses.

2.2 Canadian resin production geared towards virgin resin

RESIN PRODUCTION

With a production value of approximately CA$10.1 billion in 2017, virgin resin production accounts for the vast majority of the resins used by plastic producers and manufacturers. The industry is concentrated, mostly in Ontario and Alberta, with 87 companies producing 4,800kt of plastics resins and employing 4,000 people.

Virgin resins
CA$10B
in sales
4,000
Employees
87
Companies

Domestic production is specialized in high-volume thermoplastic resins, which represented 4,281kt in 2017, with a value of CA$8.2 billion. Polyethylene accounts for the majority of this production, with approximately 3,700kt produced in 2017. Other major thermoplastics include polyvinyl chloride (PVC, 210kt), polyethylene terephthalate (PET, 144kt), polyamide (PA, 95kt), polystyrene (PS, 80kt) and ethylene vinyl acetate (EVA, 53kt). Conversely, thermoset resins production in Canada represented 532kt in 2017, with a value of CA$1.9 billion. Four types of thermoset resins comprise the majority of production, including urea resins (204kt), phenolic resins (150kt), polyurethanes (123kt) and unsaturated polyesters (55kt).

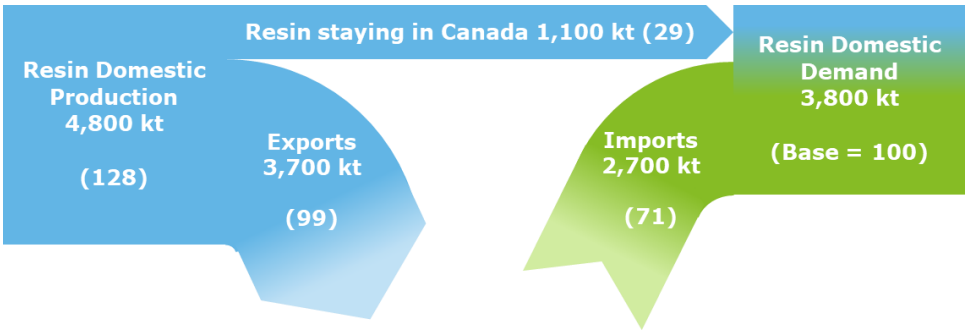
Virgin plastic resin production is dependent primarily on oil or natural gas for its source of chemical raw materials. The abundance of new, inexpensive energy sources resulting from shale gas development has precipitated unprecedented investment in new virgin resin production capacity. These investments are often vertically integrated and use the latest and most efficient technologies. This is expected to lead to an increase in the production of virgin resins in the near future, while potentially resulting in lower virgin resin prices (see blue box).

Price of virgin resins

Recently the prices of virgin plastic resins have experienced significant fluctuations. One reason for this was fluctuations of oil prices, with a sharp fall early in 2014 followed by a gradual recovery. The price of plastic resins (aggregate index) has followed a pattern that is very close to that of oil prices.

The virgin resin industry has a high level of international trade exposure, with 77 percent of the domestic production exported and 71 percent of the domestic demand fulfilled by imports (Figure 6). The US is a key trading partner, controlling more than 80 percent of the import and export share of the industry.

Figure 6: Virgin plastic resin production, demand and international trade in Canada (2016, kt) and relative share (base 100)



In comparison, the secondary market for recycled plastic resins is much smaller. In 2016, it is estimated that approximately 256kt of post-consumer plastics (mostly PET, PE and PP) were mechanically recycled in Canada, i.e., slightly more than five percent of the domestic virgin resin production. Representing approximately CA$350 million in annual revenues and 500 employees across its ten largest facilities, mostly located in Ontario, Québec, and British Colombia, the sector is however not as well documented as its virgin counterpart as it lacks some basic statistical and trade information (e.g., no specific import/export data for recycled resins).

Recycled resins
CA$350M
in sales
500
Employees
60
Major facilities

2.3 Plastic product manufacturing, a first step before integration into more complex finished products

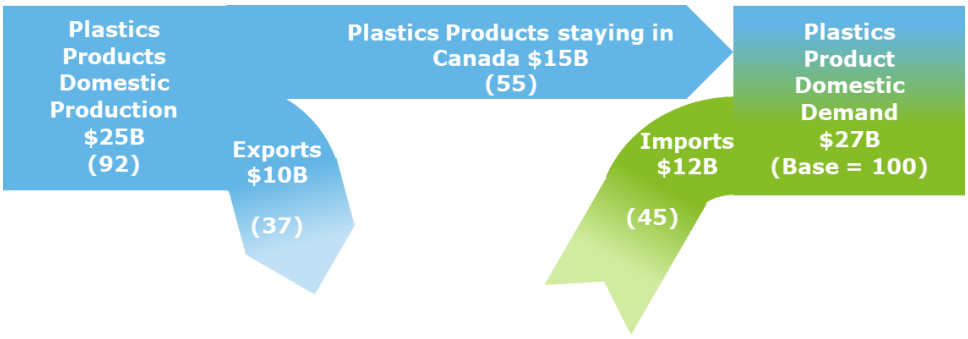
PLASTICS PRODUCT MANUFACTURING

Plastics products manufacturing
$25B
in sales
89,000
Employees
1,845
Establishments

Plastic product manufacturing is a growing sector of the Canadian economy. In 2017, sales from Canadian plastic manufacturers, sustaining 89,000 jobs, reached CA$25 billion. While this amount represents only four percent of the sales of the manufacturing sector, plastic manufacturing is its fastest growing segment experiencing an average annual growth rate of 5.5 percent between 2012 and 2017. The industry has a large pool of small and medium companies, operating approximately 1,845 establishments throughout the country, especially in Ontario, Quebec and Alberta.

Plastic product manufacturing has a high level of international trade exposure; in 2017, exports reached CA$10.2 billion, almost 40 percent of domestic output, and imports reached CA$12.3 billion, fulfilling approximately 45 percent of the domestic demand (Figure 7).

Figure 7: Canadian plastics products production, demand and international trade (2017, CA$) and relative share (base 100)



The sector demonstrates growing labour productivity with an average annual growth rate of 2.6 percent over the last five years. A large share of Canadian producers (63 percent) participate in the export market, which likely increases producers' competitiveness. However, the sector also faces challenges given the limited scale of production establishments, low investments in research and development, currency and commodity risks, and lack of skilled workers; similar to other sectors, plastic manufacturing also faces the challenge of future technological changes. Finally, as two inputs, price of plastic resins (26 percent) and labour (24 percent), account for half of the total costs of plastics manufacturing, sharp fluctuations in the price of oil can influence the price of plastic products.

A first step in the value chain of more complex products

93 percent of domestic demand for plastics products is driven by companies, which mainly use plastics products as intermediary components to incorporate into their final products or for their packaging.

Companies that mainly use plastics products as intermediary components to incorporate into their final products drive 93 percent of domestic demand. Among the top products sold by the plastic products manufacturing industry are motor vehicle plastics parts (CA$4.3 billion), plastic packaging material and unlaminated film and sheet (CA$5.5 billion), and plastic pipe and pipe fitting and unlaminated profile shape (CA$1.6 billion). Typically, these products will be further integrated into more complex finished products (e.g., cars, homes), or used as packaging of other goods.

Again, the US is the key trading partner, accounting for more than 90 percent of exports, and is responsible for over 60 percent of imports of the industry.

2.4 Few plastics products are designed with their Canadian use phase and end-of-life in mind

USE PHASE

Covering 88 percent of all resins, this study tracked products containing plastics throughout the Canadian economy, taking into account both domestic production, imports and exports of intermediate and final products. This led to the estimate that approximately CA$13 billion worth of resins, i.e., 4,667kt of plastics, were introduced to the Canadian market in 2016. As resins follow the import and export of intermediate (e.g., plastic motor vehicle parts) and final products (e.g., cars), few products containing plastics are designed with their Canadian use phase and therefore their Canadian end-of-life in mind (Figure 8).

Figure 8: Flows of resins in products containing plastics (2016 extrapolation based on 2014 Supply and Use Tables, CA$) and relative share (base 100)

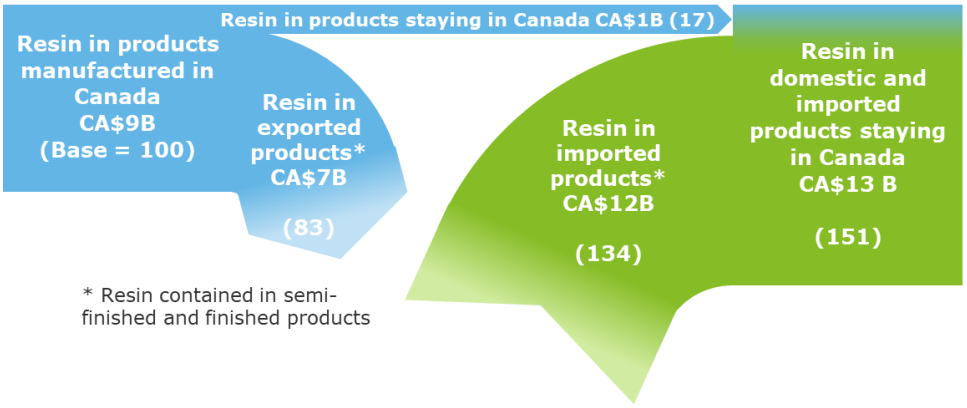
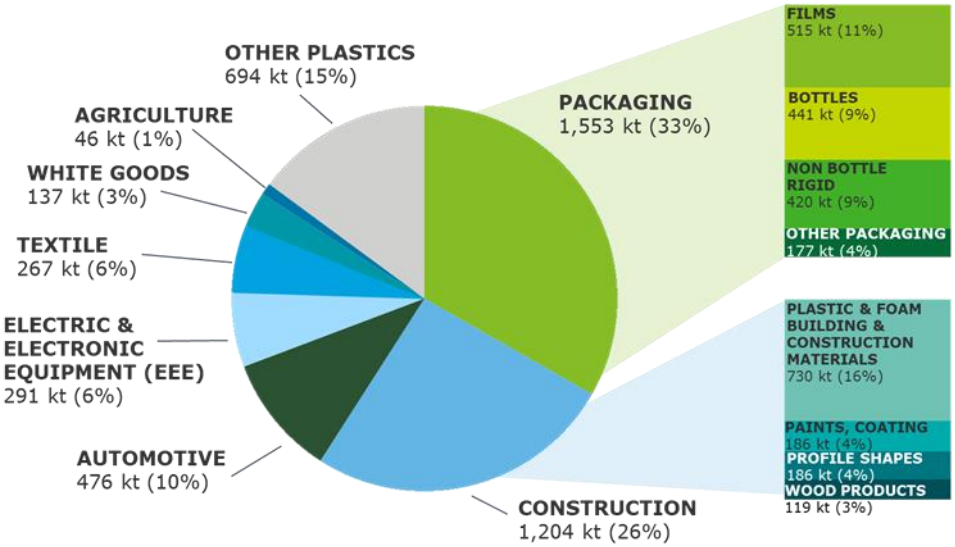


Figure 9 illustrates the end-use markets for plastics in those products staying in Canada. Three sectors (see blue box on the right) account for nearly 70 percent of plastic use: packaging, construction, and automotive.

Figure 9: End-use markets for plastic products in Canada (kt, 2016)



Sectors

Products containing plastics were grouped into eight "sectors" developed for the purpose of this study: packaging, construction, automotive, electric and electronic equipement, textile, white goods, agriculture and other plastics.

Section 5.2 provides details on the products grouped into each sector.

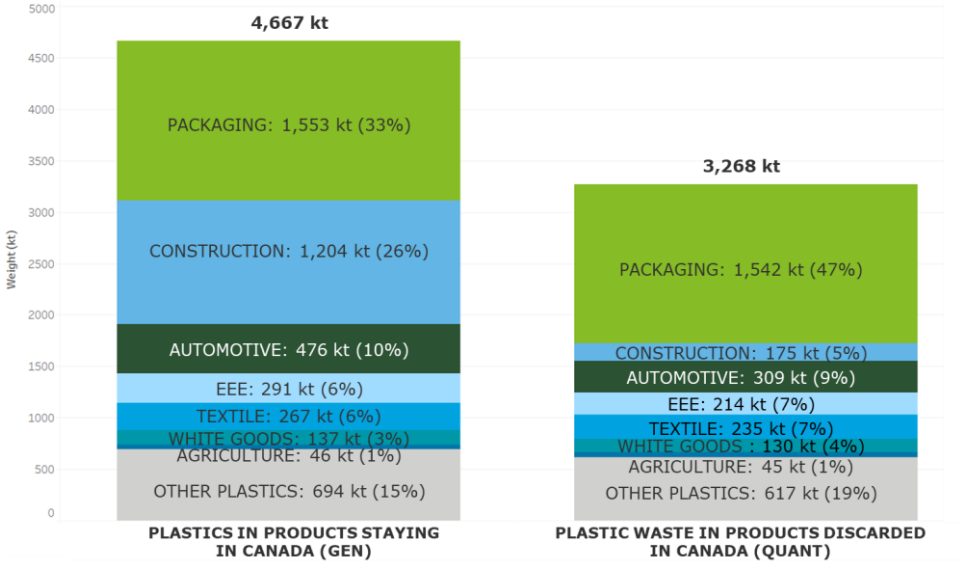
2.5 Packaging applications driving plastic waste, at least for now

END OF LIFE

Durable applications with an average lifetime over a year will end up as waste only in later years. Given the market growth and increased share of plastics in durable applications (e.g., construction, cars), plastics waste generated today is less than what is being introduced to the market that same year. Conversely, nondurable applications go almost straight to waste.

This means that, while packaging accounted for 33 percent of plastics introduced to the market in 2016, it accounted for 47 percent of all plastic waste discarded that same year (Figure 10). In coming years, the profile and quantity of plastics waste will progressively adjust to reflect the quantity of plastic waste from durable applications introduced to the market today, in particular with an increasing plastic waste stream coming from the construction sector, in which products have the longest average lifetime (between 15 and 25 years).

Figure 10: Plastics entering the market and plastics discarded in Canada (kt, 2016)



More plastic waste to come

In 2016, 43 percent more plastics entered the market in Canada (4,667kt) than plastic waste discarded in the same year (3,268kt).

2.6 Only 25 percent of plastic waste is collected for diversion

Once discarded in various products, plastic waste can be either collected for direct disposal (i.e., to be sent to landfills) or collected for diversion (i.e., diverted from direct disposal and sent to a sorting facility). The collection of plastic waste for diversion (e.g., through curbside collection, recycling depots, deposit-refund systems, etc.) is highly dependent on the end-use sector. As illustrated in Table 1, only 25 percent of all plastics discarded are collected for diversion (i.e., 807kt collected out of the 3,268kt discarded).

Table 1: Diversion rate broken down by sector, 2016

Sector	Plastics discarded[1] (kt)	Diversion rate[2] (%)	Plastics diverted[3] (kt)
Construction	175	11	19
EEE	214	16	34
Packaging	1,542	23	347
Textile	235	5	11
Automotive	309	100	308
White goods	130	64	83

Sector	Plastics discarded[1] (kt)	Diversion rate[2] (%)	Plastics diverted[3] (kt)
Agriculture	45	9	4
Other plastics	617	0	0
Total	**3,268**	**25**	**807**

[1] Quantity of plastics discarded representing the plastic entering waste streams (QUANT)

[2] Diversion rate is the share of plastic diverted from direct disposal and sent to a sorting facility divided by plastics waste available for collection (R1/COLL)

[3] Plastic diverted from direct disposal and sent to a sorting facility (R1)

[1,2,3] See Section 5.3 for more details on the plastic waste management model and its underlying assumptions.

There are several contributing factors to the low diversion rate for end-of-life plastics in Canada. Some of the most important contributing factors are included in Table 2.

Table 2: Contributing factors to the low diversion rate in Canada

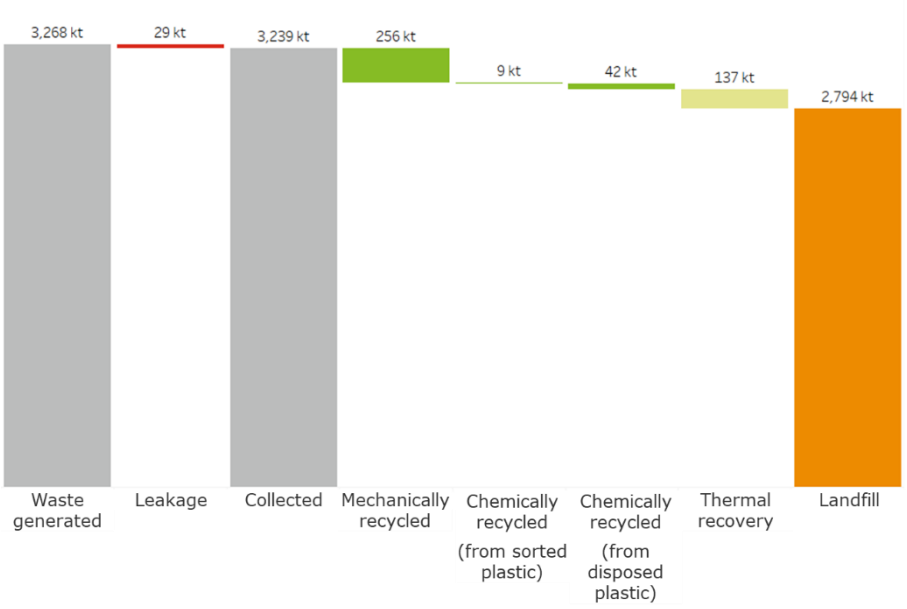
Area	Factors
Product design	• Continued poor adherence to available "design for recyclability" standards on behalf of many brand owners reduces the amount of end-of-life plastic waste that can be diverted to the recycling stream
Collection mechanisms	• Improper sorting at the consumer and collection level (e.g., increasing reliance on single-stream collection systems) results in the contamination of collected plastics. Additional sorting and quality control are thus necessary at material recovery facilities, and additional technologies to remove or mask a moving target of contaminants at plastic recycling plants • The realities of the geography of Canada, in which plastic consumption is distributed over a wide area (e.g. end-of-life agricultural plastics) limits the ability to establish comprehensive and cost-effective collection systems
Collection from ICI	• There are low levels of end-of-life plastics collection from the industrial, commercial and institutional sectors, which generally fall outside of established municipal collection systems
Infrastructure	• Current lack of robust infrastructure for chemical recycling or thermal recovery of end-of-life plastics limits the potential diversion routes for hard-to-recycle plastic material
Regulatory	• There is a lack of robust government intervention (as compared to other international jurisdictions) to force a greater level of diversion (e.g., landfill bans)
Economic and price signals	• Low virgin resin prices establish the ceiling at which recycled resins can be sold, impacting the amount of end-of-life plastic products that can be cost-effectively diverted for recycling • The cost of separating end-of-life plastics from certain waste streams is prohibitive (most notably for construction), especially when compared to other available management options (e.g., low landfill tipping fees) • General lack of markets for several recycled plastic resins from end-of-life plastics (e.g. polystyrene, plastic film, construction and demolition waste) limits most plastic recyclers from accepting/managing those materials

The above factors combine to form a system in Canada that does not provide the necessary incentives or outlets to divert plastics away from the disposal route for some end-of-life plastic streams and generators. Automotive and white goods are noticeable exceptions, as metals from end-of-life vehicles and large appliances provide additional incentive to collect these products. However, even in the case of end-of-life vehicles and white goods (although they are diverted), the plastics that are contained in these materials eventually end up in shredder residue, which in Canada is disposed of in landfills (although often used beneficially as daily landfill cover). After collection, plastic waste has access to various value-recovery options, presented in Section 2.7.

2.7 Canadian value recovery options, focused today on mechanical recycling, are slowly expanding

There are various value recovery options for plastic waste, as illustrated in the waste management hierarchy (see box on the right). In 2016, three recovery options (i.e., mechanical, chemical and thermal) enabled the diversion of 13 percent of plastic waste from landfills in Canada. Figure 11 highlights that mechanical recycling is the first option for recovery, accounting for eight percent of plastic waste, followed by thermal recovery (four percent) and chemical recycling (one percent). This section describes the key elements of each option.

Figure 11: Waterfall view of total plastic waste over the lifecycle (kt, 2016)

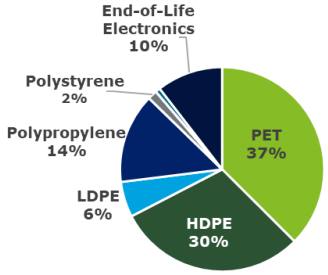


Remanufacturing, refurbishment, repair and direct reuse (RRRDR) is the first option in the waste management hierarchy, but the least present in Canada. Initiatives to reuse or repair certain products containing plastics (e.g., textiles, electronics, construction) exist, but remain fragmented and small scale in nature (and therefore are not included in Figure 11). The impact of these initiatives is difficult to quantify given that some (e.g., repair and direct reuse) temporarily reduce waste by keeping products in service for longer, while others (e.g., remanufacturing) provide a new lifetime to the material. Overall, several factors limit the development of RRRDR approaches in the plastic value chain, including:

- The dominance of linear business models, through which products are manufactured, distributed, consumed and then disposed of, with limited options for RRR. However, more circular oriented models are emerging, such as the function economy (through which companies sell a service rather than a product);
- There can be a negative tradeoff for the consumer/user, for whom it is generally cheaper to dispose and buy new products than to repair the old one;
- Products are often not repairable by design (voluntarily or not);
- The lack of mechanisms in place for reverse logistics, which jeopardizes the economic viability of RRR activities by adding collection and transport costs; and
- The replacement of plastics parts (e.g., casings, shells, or hulls) to provide new 'look and feel', even if a product is remanufactured.

Mechanical recycling is currently the main value recovery option utilized in Canada. The vast majority of post-consumer mechanical recycling economic activity occurs at approximately 10-11 facilities across Canada, which typically (but not exclusively) produce resins and/or flakes of multiple resins. These facilities primarily recycle PET, HDPE, LDPE and polypropylene, which almost exclusively originate from packaging. The main challenges faced by mechanical recycling operations include the continued low prices of virgin resins, low bale quality received from some municipalities resulting in higher operating costs and lower profitability, the prevalence of poor design decisions (from a recyclability standpoint) on behalf of brand owners, and increasing costs to transport bales from various municipalities to the recycling facility.

Resins mechanically recovered (volumes, 2016)



Improvements that could increase the amount of mechanically recycled post-consumer plastics in Canada include:

- Facilitating greater adherence to "design for recyclability" guidelines by brand owners to reduce the quantity of end-of-life plastics that cannot be recycled for technical and/or economic reasons;
- Ensuring a continued market for post-consumer resins, irrespective of potential reductions in the price of virgin resin (e.g., by mandating post-consumer content in some plastic products);
- Encouraging municipalities to enter long-term contracts with Canadian recyclers, thereby ensuring raw material availability for these recyclers and the resulting stability to invest in plants and equipment; and
- Fostering a collection and separation system that reduces the contamination of post-consumer plastic bales.

Despite these potential improvements, there are limits to the increase in plastic waste that the system can manage. The fact remains that some end-of-life plastics cannot be cost-effectively recycled mechanically (i.e., the post-consumer resin that is produced would have to be priced much higher than virgin resins). In other instances, there is simply no market (or the market is not sufficient) to sell the post-consumer resins that are produced. This was a major contributing factor to the Canadian export of certain end-of-life plastic streams overseas for processing, as there was a very small or non-existent North American market for these resins.

Chemical recycling of plastic waste is the process of converting plastic waste into shorter molecules, for use in the production of new plastics or fuels. From a circular economy perspective, the utilization of chemical recycling technologies to produce new plastic resins would be preferred. However, at present the companies that operate these types of facilities in Canada are generally managing small quantities of post-consumer plastics. Conversely, chemical recycling facilities that are producing fuels from end-of-life plastics are managing much higher quantities of plastic waste. Although still in the emerging phase, chemical recycling is recognized as being a potential outlet for end-of-life plastics that cannot be mechanically recycled due to technical, economic or market considerations. Developing technologies are creating a new market and offering innovative outputs for plastic waste. Further, they offer an additional source for plastic producers or for other industries if the recycling process includes a polymerization phase or a dissolution. Chemical recycling could bring new solutions to the sorting issue by accepting "lower quality" or mixed input, such as shredder residues from the automotive, EEE, or white goods sectors. Furthermore, actors in the private and public sector view chemical recycling as an opportunity to respond to societal expectations in terms of "closed-loop" economy. Enhancing or investing in these technologies could help address mixed plastics treatment on a large scale through projects with greater acceptability (versus waste-to-energy plants). Six companies in Canada have commercialized or are nearing commercialization of chemical recycling processes using waste plastics as feedstock. However, several of these technologies still need to be scaled up, or demonstrate commercial viability.

Composting is an option that has been explored in Canada, but very little post-consumer plastic is managed through industrial composting facilities, with biodegradable and compostable plastics often considered a nuisance by industrial composting operations. There is no labelling requirement, standardized chemistry or standardized degradation time for biodegradable plastics, and even certified compostable plastics are not accepted by many composting facilities in Canada due to the differences between the certification requirements and their operating conditions.

Incineration with energy recovery (also called waste-to-energy or thermal recovery) is the second most prevalent value recovery option for managing plastic waste in Canada, with 137kt treated in 2016. The vast majority of these plastics are thermally recovered at Canada's five waste-to-energy plants, but other facilities such as steel and cement manufacturing plants could use plastic for energy (volumes used in these applications are estimated to be low). Plastics are valuable fuels because they are made with petroleum and generate energy when incinerated. Waste-to-energy (as well as cement) facilities accept all kinds of plastics, including currently unrecyclable resins such as thermosets and mixed plastics, and thus offer interesting avenues for treating waste from certain sectors. However, due to the substances released during incineration (e.g., dioxins, furans, heavy metals, and volatile organic compounds) waste-to-energy facilities typically have significant public opposition to their construction and/or expansion. It may therefore be difficult to expand upon Canada's current infrastructure of waste-to-energy plants to provide more outlets for increased value recovery of hard-to-recycle plastics. All five of Canada's current waste-to-energy facilities are operating at full capacity and generally are not allowed to accept waste materials from outside of their jurisdiction. Currently, there are no known new waste-to-energy plants being considered in Canada.

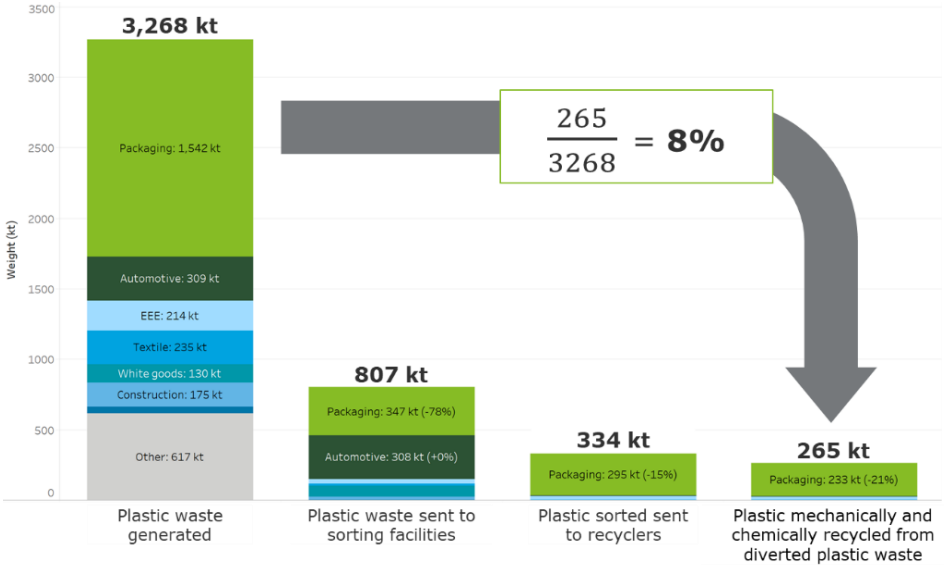
Following Section 2.8 presents sector specificities concerning plastic waste management in Canada.

2.8 Value recovery performance, drivers and challenges vary greatly by sector

The overall value recovery rate (which includes mechanical and chemical recycling from disposed and diverted waste, as well as thermal recovery) for plastics reached 13 percent in 2016 in Canada. However, the situation varies greatly between the eight sectors defined for the purpose of this study (see Section 5.2).

Figure 12, while focused on only one value recovery option (i.e., recycling from diverted plastic waste), illustrates already some of those major differences and in particular the specific role of packaging which accounts for 88 percent of all plastics resins recycled.

Figure 12: Plastic at different stages of the waste life cycle, per sector (kt, 2016)



Source: (Deloitte, 2019a). Please refer to model introduced in Section 5 to identify data in recycling value chain

Further, and based on a comparative analysis of their performance rates and yields (see Table 3), the eight sectors were clustered into four distinct groups: plastics from packaging; plastics in other products targeted by extended producer responsibility (EPR) systems; plastics collected but discarded; and untargeted plastics. Key characteristics of each group are presented in this section.

Table 3: Diversion rate, recycling rate and value recovery rate, per sector, 2016

Sector	Plastics discarded[1] (kt)	Diversion rate[2] (%)	Recycling rate[3] (%)	Value recovery rate[4] (%)	Plastics recovered[5] (kt)
Packaging	1,542	23	15	21	327
EEE	214	16	13	15	33
Agriculture	45	9	5	10	5
Automotive	309	100	0	0	0
White goods	130	64	0	5	7
Construction	175	11	1	6	11
Textile	235	5	0	7	17
Other plastics	617	0	0	7	43
Total	**3,268**	**25**	**8**	**13**	**442**

[1] Quantity of plastics discarded representing the plastic entering waste streams (QUANT)

[2] Diversion rate is the share of plastic diverted from direct disposal and sent to a sorting facility divided by plastics waste available for collection (R1/COLL)

[3] Output recycling rate is the share of plastic that is ultimately reprocessed whether through chemical or mechanical recycling from diverted waste, divided by plastics waste available for collection (R3/COLL). This rate does not include chemical recycling from disposed waste (D-CHEM).

[4] Value recovery rate, or share of plastic that is ultimately value recovered (whether through chemical or mechanical recycling from diverted and disposed waste or through thermal recovery), divided by plastics in waste collected. This rate is equal to (R3+D-CHEM+D-EFW)/COLL

[5] Quantity of plastics recovered through chemical or mechanical recycling from diverted and disposed waste or through thermal recovery (R3+D-CHEM+D-EFW)

[1,2,3,4,5] See Section 5.3 for more details on the plastic waste management model.

Plastics from packaging

Plastics from packaging (e.g., films, bottle, non-bottle rigid) represents 1,542kt or 47 percent of all plastic waste generated in Canada in 2016. Overall, it is the first source (74 percent) of value recovered plastics with 327kt. Its 21 percent value recovery rate is supported by the highest recycling rate among all sectors, 15 percent. Packaging is targeted by several EPR and other programs, such as deposit-refund systems for beverage plastic bottles, which are the main drivers for its fairly high diversion rate (23 percent). However, this diversion rate is also limited due to multiple root causes, including (but not limited to) lack of collection infrastructure away from home and lack of acceptance of many products by curbside collection. Most plastics used in packaging (e.g., PET, PE, PP) have a high recyclability and are the focus of attention for recyclers given the relative high value of these resins on the secondary market. However, the dominance of single-use products, the variety of packaging design and materials (multi-laminate), the presence of additives or pigments also affects contamination of waste streams and overall profitability of plastics packaging value recovery.

Packaging

- First source of plastic waste generated (1,542kt in 2016)
- Highest value recovery rate: 21% (of which 15% through recycling)
- Pros: high recyclability and large utilization of thermoplastics, widely deployed curbside collection of recyclable goods, EPR and deposit-refund programs, efficient sorting technologies, relatively high value of resin recycled
- Cons: lack of collection infrastructure away from home, not all products accepted by curbside collection, dominance of single-used products, multiplicity of packaging design and materials, presence of additives/pigments

Plastics in other products targeted by extended producer responsibility (EPR) schemes

In addition to the EPR systems applicable to packaging, several additional mandatory or voluntary EPR schemes exist in Canada, in particular for the EEE and agriculture sectors. They allow for partial collection and recycling of plastics waste within the targeted sector.

The Electronic Products Recycling Association (EPRA) operates programs across Canada to collect targeted electrical and electronic equipment products (e.g., computers, printers, display devices like television sets, audio/video systems and phones) and to send them towards recycling streams. Although plastic contained in EEE (mainly EPS, PP and ABS resins) is not specifically targeted by EPRA, it is nonetheless sorted and recycled through shredding operations and categorized within the mixed plastic stream (lower quality). Out of the 214kt of EEE plastic waste generated annually, 33kt or 15 percent are recovered (mainly through mechanical recycling: 26kt). This material is usually exported to Asia, although the number of countries that are still willing to accept shredded mixed plastic waste from EEE waste recyclers is becoming rapidly smaller.

EEE

- Half of plastics from EEE waste are targeted by a nationwide EPR system
- Value recovery rate: 15% (29% for EEE products targeted by EPR system)
- Pros: EPR system in place, wide access rate
- Cons: 33% diversion rate, low quality recycled plastics (shredded mixed material), shrinking end-market (Asia)

The agriculture sector has deployed five known voluntary EPR schemes on various product categories in several provinces in Canada. They target plastics used for grain and seed transportation bags, fertilizer and pesticide packaging, as well as agricultural films – mainly HDPE, LDPE and woven PP. Discussion with one stewardship organization indicated these EPR schemes are expanding year after year. In 2016, out of the 45kt of plastic waste generated by the agriculture sector, approximately 4kt is collected for diversion (nine percent), 2kt recycled (five percent), 2kt incinerated (waste-to-energy), and 40kt sent to landfills.

Agriculture

- Approximately 45kt of plastic waste generated annually
- Value recovery rate: 10%
- Pros: use highly recyclable plastics (PE, PP); several EPR system implemented
- Cons: scope and geography of EPR systems to be widened

Plastic collected but discarded

In the automotive and white goods sector (e.g., large appliances such as fridges or stoves as well as small household appliances such as food processors or electric kettles), the recycling of plastic is almost non-existent. Diversion rates are, however, very high (100 percent for automotive, 64 percent for white goods) as products are collected for recycling. However, they are usually sent to a shredder where only the material of interest (generally the metal content) is sorted and sent to recyclers. It is indeed more cost-effective and less labour-intensive to crush and shred vehicles or appliances for metal recycling than to dismantle parts, including plastic parts.

Automotive

- 1.6 million end-of-life vehicles are retired annually, or 309kt of plastic
- Almost no value recovery for plastics occurs. Shredded plastic used as cover/capping material for landfills
- Pros: opportunity for sorting (transit through auto-recyclers)
- Cons: low recyclability (plastic blends, contamination), no incentive to sort (cost, labour), absence of end-market for recycled resin

In the automotive sector, the quasi-absence of end markets for the plastic contained in cars, which are often blends or potentially contaminated by automotive fluids and additives, reduces the incentives for recyclers to explore this avenue.

In the white goods sector, the low presence of appliance manufacturers in Canada (whether to implement closed loop recycling, remanufacturing or re-use of spare parts) has an effect on the economic cost of disassembly. In addition, there are limited end markets for mixed shredded plastics. Combined, those two factors limit recycling of plastics from white goods.

White goods

- Recycling of white goods targets metal parts and not plastic
- Almost no recycling for plastics occurs, and shredded plastic is used as cover/capping material for landfills
- Value recovery rate: 5% through waste-to-energy
- Pros: diversion stream in place (large appliances)
- Cons: plastic parts of low interest for recyclers, high cost of disassembly, very small presence of appliance manufacturers in Canada

Thus, in these two sectors, plastic shows a good collection rate, but is turned into shredder residue and sent to landfills, usually as daily cover material. Despite this poor performance, the existing collection channels (through which the products and their plastic content get collected for diversion) represent an opportunity, with the right market signals, for increased recycling.

Untargeted plastics

Plastic waste from the last three sectors (e.g., construction, textile and other plastics) shows very low collection rate, sorting and reprocessing yields, either across the board or at one specific step of the value chain, leading to an overall quasi-null recycling rate. This situation stems from different reasons, including (but not limited to): hard to recycle plastics (e.g., blends, thermosets), contamination (e.g., problematic additives, dusts), and the absence of incentives to sort/recycle.

While construction sector is still a relatively small plastic waste generator (175kt or five percent), its share will progressively increase to reflect its current share of plastics introduced to the market in Canada (1,204kt or 26 percent, see Figure 10). As such, this sector will likely play an increasingly important role in the overall performance of plastics value recovery in Canada.

Most value recovery in those three sectors occurs through incineration with energy recovery.

3. A zero plastic waste economy would deliver significant benefits to Canada

3.1 A zero plastic waste economy by 2030

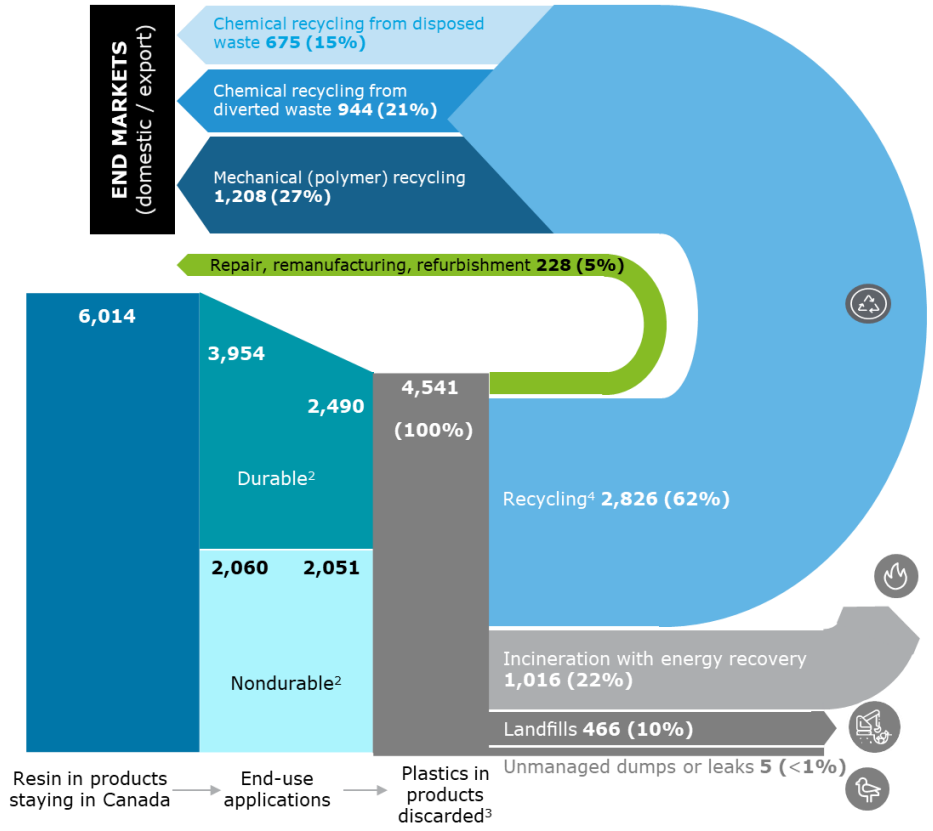
To illustrate a different future for plastic management in Canada, the authors developed two plastic waste management scenarios at the **2030 horizon**[1]:

- A business as usual scenario (2030_{BAU}), taking into account a generic market growth for all sectors and keeping the same performance parameters as the 2016 baseline (Figure 5 in Section 2); and
- An ambitious scenario (2030_{T90}) in which the overall system performance leads to the diversion from landfill of 90 percent of the discarded plastic waste (Figure 13).

The ambitious scenario is not a prediction or a recommendation: it is an illustration of what zero plastic waste could look like given current product designs and emerging value recovery technologies. It was developed to model the potential costs and benefits of achieving zero plastic waste if the plastic production and end use applications remain unchanged from 2016. Changes in plastic production and design would open the door to a very different scenario with higher value recycling and recovery options.

[1] While this Task 5 report presents only results associated with the 90 percent diversion scenario and its comparison with the business as usual one, another scenario illustrating a 50 percent diversion rate was also developed in Task 2 report (Deloitte, 2019b).

Figure 13: Canadian resin flows in thousands of tonnes per annum, 2030_{T90} scenario[1]



[1] Scenario based on a multi-stakeholder push to boost recycling, including investment in new facilities, regulatory measures to encourage recycling, significant progress on technologies and favorable end-markets demand.

[2] Durable applications with an average lifetime >1 year will end up as waste only in later years; given market growth and increase share of plastics in durable applications (e.g., construction, cars) plastics waste generated today is less than what is being put in the market that same year. On the contrary nondurable applications go almost straight to waste.

[3] 2,051 thousand metric tons of mixed plastic waste from nondurable applications plus 2,490 thousand metric tons of mixed plastic waste from production in previous years.

[4] Output recycling rate, after taking into account process losses

While this ambitious scenario (2030_{T90}) represents a promising and potentially achievable future, it is based on systemic and far-reaching assumptions, which are presented in the next section.

3.2 A path towards a 90 percent diversion of plastic waste

A 2030 scenario based on a 90 percent diversion of plastics waste from landfill (2030_{T90}) can only be envisioned when coupled with a series of major systemic changes compared to business as usual, at all stages of the plastics value chain. To achieve the required increase to diversion rates for plastics waste in Canada, significant improvements in the quantities managed by the various value recovery options are required. The 2030_{T90} scenario was developed by first pushing mature technologies like mechanical recycling, then projecting chemical recycling development, and finally resorting to waste-to-energy. Technical, economic and market limits on the quantity of end-of-life plastics that can be mechanically recycled were considered first. Chemical recycling growth potential was then estimated, given its attractiveness from several viewpoints (i.e., circular economy, management of hard-to-recycle plastic waste, public perception) and the presence in Canada of several entrepreneurial firms that have developed market-ready and/or proven chemical recycling technologies.

Key assumptions underlying the 2030_{T90} scenario are presented in three tables. First, Table 4 presents the key end of life assumptions for 2030_{T90}.

Table 4: Key end of life assumptions for 2030_{T90}

End of life of plastic waste	Change from 2030_{BAU} to 2030_{T90}	Key assumptions and rationale
Plastics leakage into the environment	From 1 percent to 0.1 percent	Plastic leakage (i.e., permanent litter) reduced ten-fold because of increased awareness from consumers and initiatives from public/private sector actors to reduce litter.
Repair, remanufacturing and refurbishment (RRR)	From <1 percent to 5 percent	RRR levels rapidly scaled in sectors in which RRR activities already exist in other jurisdictions (e.g., white goods and EEE sectors).
Mechanical recycling	From 7 percent to 27 percent	Mechanical recycling quadrupled due to improved (or maintained in the context of increased volumes) sorting and reprocessing yields, and scale-up of the number of facilities. This is the target scenario proposed by industry associations in Canada.
Chemical recycling	From 1 percent to 36 percent	Chemical recycling facilities scaled up following increased recycling activity, based on technologies currently developed in Canada (e.g., monomer recycling for PET/PA, building block recycling for PS/PE, pyrolysis to generate liquid feedstocks/fuels from disposed waste).
Incineration with energy recovery	From 4 percent to 22 percent	Incineration with energy recovery, while not a preferred option to recover plastic waste, is scaled (as a necessary recourse) to meet the 90 percent diversion target. This increase could be supported by additional facilities and by having existing industrial facilities (e.g., cement kilns) accept more plastics.

Second, the end of life assumptions above are based on additional assumptions regarding the entire recycling value chain (Table 5). Those assumptions represent significant efficiency improvements at each key step of the value chain and take into account an analysis of the value recovery technologies and their readiness level. In particular, chemical recycling technologies, which in Canada range from pilot to larger scale commercial, were significantly factored in to be able to reprocess the increased projected volume and diversity of resins present in the Canadian mix.

Table 5: Key recycling value chain assumptions for 2030_{T90}

Recycling value chain step	Change from 2030_{BAU} to 2030_{T90}	Key assumptions and rationale
Diversion rate	From 25 percent to 77 percent	Multi-stakeholder (consumer, industry, government) push to collect more plastics waste for diversion. Sector assumptions pushed to their maximum given sector specificities, including a major a push from 23 percent to 90 percent in packaging.
Sorting yield	From 40 percent to 82 percent	Increased sorting of plastics within diverted waste, in particular for waste from sectors that do not currently focus on plastics, such as automotive, white goods, and textile.
Reprocessing yield	No change (maintained at 79 percent)	Maintained reprocessing yield (chemical and mechanical) in the context of an additional amount of sorted plastic waste, including harder-to-recycle resins.
End-markets	A viable and stable domestic end-market for secondary plastics is developed	End-markets exist for all secondary plastic products and their by-products at a viable price point, which means either favourable virgin resin price and/or the development of a viable decoupled secondary plastics market. The quality of recycled plastics is broadly comparable to virgin resins.

Third, the significant expansion of all value-recovery options assumes support for the development of new facilities. The model projects the need to add 167 facilities for a total estimated investment of between CA$4.6 billion and CA$8.3 billion for 2030_{T90}, broken-down by facility types (Figure 14).

Figure 14: Additional capacity and investment estimates, 2030_{T90}

This estimate is based on:

- Additional waste processing capacities required in future scenarios compared to the current situation (2016 baseline);
- Average size of waste processing facilities; and
- Investment cost proxies, specific to four key step of the waste processing system: sorting, recycling of diverted waste (based on mechanical recycling estimates), chemical recycling from disposed waste and incineration with energy recovery. Landfilling capacities in 2016 were estimated to be sufficient for 2030 requirements under the scenarios considered.

Assumptions and values used for these estimates (Table 6) are based on recent investments for the various facilities and their feedstock composition.

Table 6: Capacity and investment requirement key assumptions

Type of facility	Facility average capacity (in kt of plastics waste)	Investment cost (low-high range, $/t[2])
Sorting	45 kt/y	750–1,200
Mechanical / chemical recycling from diverted waste	35 kt/y	400–1,200
Chemical recycling from disposed waste	30 kt/y	1,000–1,300
Energy from waste	106 kt/y	1,400–2,000

Source: (Deloitte, 2019b).

3.3 Benefits of a zero plastic waste economy by 2030

While the significant investment required to manage plastic waste under the 2030_{T90} scenario is reflective of the challenge Canada is facing, a comparative analysis between this scenario and business as usual demonstrates benefits from an economic, social and environmental point of view, as illustrated by Figure 15. These benefits should be considered in light of the investments required, as presented in Section 3.2.

Figure 15: Comparative analysis of scenarios

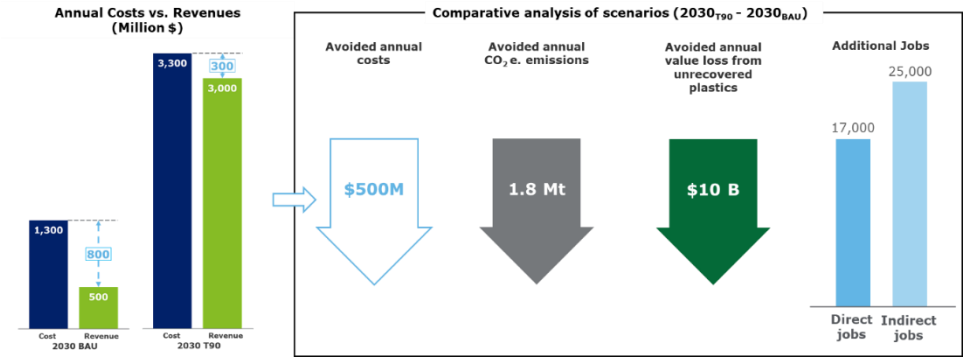


Table 7 presents the assumptions made for each area of the comparative analysis.

Table 7: Assumptions supporting the comparative analysis of scenarios

Comparison element	Change from 2030_{BAU} to 2030_{T90}	Key assumptions and rationale
Operating Costs	From CA$1,300 million to CA$3,300 million	Average costs per tonne of plastic going through each step of the recycling value chain were estimated based on available proxies and multiplied by material flows projected for both scenarios.
Revenues	From CA$500 million to CA$3,000 million	Price per tonne of recycled plastics along the value chain was estimated based on available proxies and reference points, and multiplied by material flows projected for both scenarios.
Direct jobs	From 10,000 to 27,000 direct jobs	Additional jobs in collection, sorting, and reprocessing counterbalance the losses in less labour intensive landfilling operations.
Indirect jobs	Same multiplier	Multiplier effect of 1.5 times each direct job.

[2] Per tonne of plastic waste (conversions have been made when facilities capacity was initially provided in tonne of MSW).

Comparison element	Change from 2030_{BAU} to 2030_{T90}	Key assumptions and rationale
CO_2 emissions across full lifecycle	From +0.2 to -1.6 Mt CO_2 equivalent (CO_2 e)	Avoided emissions through substitution of virgin resins with recycled plastic, offsetting direct emissions from other steps of the value chain, such as incineration with energy recovery.
Value loss from unrecovered plastics	From CA$11.1 billion to CA$1.4 billion	Value of unrecovered plastic (plastic sent to landfill or leaked into the environment) based on virgin resin prices.

3.4 Scenario implications for plastics markets

Achieving the 90 percent scenario would have impacts on the primary and secondary plastic markets. The increased quantity of recycled material (e.g., resin polymer, building blocks, monomers or feedstocks) could reach approximately 45 percent of plastics resin domestic demand. However, given the importance of international trade in the domestic plastics resins production sector, in particular with the US (see Section 2), it is difficult to forecast the final destination or usage of that recycled material. This material could be used to close the loop domestically by displacing imports or primary production, but it could also be exported, depending on several factors such as price, quality and demand for recycled material.

4. Canada needs an integrated approach to plastic management

Drawing the portrait of a 2030 scenario where 90 percent of plastic diversion is attained demonstrated that this goal could be realistic and drive significant benefits; however, this will require a concerted effort across several stakeholders in the public and private sector. It also demonstrated that Canadian society must implement radical changes to its current plastic management throughout the full lifecycle.

There is no single public or private sector action that can shift the system; international benchmarks from ten European jurisdictions, and examples from US and Australian case studies demonstrated that a systemic approach is needed, acting in several areas concurrently. A wide range of policies and approaches can be used to achieve these objectives, and this final section highlights those that have been effective in other jurisdictions.

4.1 Create a viable domestic secondary end-market

The main challenges of a secondary market are the lack of demand, low prices of secondary resins that compete with virgin resins, and the lack of supply. Thus, one of the most important actions that can be taken to encourage recycling is to create a reliable domestic market for collectors/processors/recyclers that is uncoupled from primary resin prices. As highlighted in Table 8, this could be accomplished by developing product-based quotas or requirements for secondary material content.

Table 8: Measures to support the creation of a viable domestic secondary end-market

Measure	Rationale
Product-based quotas or requirements for secondary material content	Creating a guaranteed stable domestic demand for secondary materials and subsequently increasing investment in plastics recycling/diversion. This could be thought of as the "first domino" that must be toppled to create cascading impacts on secondary plastics infrastructure investment and use. Certain products (bottles, certain packaging) that do not have difficult performance requirements (flame retardant, food-safe) could use secondary plastics of sufficient purity without significant issue.
Tax or fee on virgin resins	Introducing a tax or fee on virgin resins would make secondary plastic more economically appealing to manufacturers. However, the high volatility of oil price and the significant investment in virgin resin production would make that tax/fee hard to adjust in time to reach the desired effect. Further, it could lead to increased consumer prices.

Deployment of such measures could be progressive, beginning by targeting certain categories for which it is already technically and economically feasible. This requirement may be difficult to implement for imported products.

The creation of a reliable domestic market for collectors/processors/recyclers that is uncoupled from primary resin prices cannot be rolled out alone and should be accompanied with policies to:

- Improve the quality of recovered plastics at both the point of collection and in materials processing;
- Improve access to domestic supply of recycled content; and
- Support innovation in product design and use of secondary plastics.

4.2 Get everybody onboard to collect all plastics

Reaching a zero plastic waste goal will require major concerted efforts from all stakeholders of the value chain, including producers, retailers, consumers, recycling actors, and the public sector. As mentioned above, the recycling burden in Canada is currently concentrated within a few plastic product categories (e.g., packaging) and actors (e.g., residential consumers), resulting in the collection of only 25 percent of plastics. To trigger the systemic engagement of all parties, policy makers must consider several measures at different levels, such as illustrated in Table 9.

Table 9: Measures to support collection of plastics

Measure	Rationale
Requirements/incentives to participate in recycling	Widening recycling obligations/incentives to industries, commerce and institutions (ICI) is a first step to mobilizing the country towards a zero plastic waste goal. For example, policy makers can introduce differentiated recycling targets for plastic products (e.g., reduction targets for plastics in vehicles, rather than undifferentiated targets for all materials in vehicles), and deposit refund systems (through which an incentive is created to return/recycle a product).
Create sector requirements and mechanisms to support compliance	Approaches such as extended-producer responsibility (EPR) or performance agreements have the capacity to engage the entire value chain to rethink plastic usage. The most effective programs would target specific products and include standardization requirements, secondary material use requirements, and set trackable recycling targets.
Restricting disposal (e.g., landfill taxes or bans)	Whether they selectively target a specific product/sector or are broader, landfill restrictions or bans send a strong signal along the value chain, and require collective efforts. Providing significant lead-time between announcement and enforcement is necessary to ensure industry/governments have sufficient time to adapt and develop new infrastructure.
Directive or restrictions (e.g. bans) on specific products (e.g., *Single-Use Plastics Directive* in Europe)	These measures prevent the generation of problematic wastes in the first place. Although not always an option (e.g., automobiles), certain single-use plastics can be replaced with reusable alternatives, and taking action against certain single-use products could reduce the volume of plastic waste that must be managed.
Increased public awareness	Promote public awareness to enhance recycling program participation.

These measures have the potential to divert significant quantities of plastic waste from landfills. However, installed capacity to properly manage this influx of plastic waste is currently missing in Canada. Thus, prior to implementing the above-listed measures, policy-makers should consider the following:

- To ensure effectiveness, EPR programs should target specific products and include standardization requirements, secondary material use requirements, and set trackable recycling targets;
- When voluntary standards are in place (e.g., list of approved glues, labels, additives for specific applications), they appear to have no impact; regulators should ensure these standards are capable of achieving waste reduction; and
- Actions to expand the capacity of recovery options.

4.3 Support and expand all value-recovery options

The current value recovery options in place do not allow the recycling of all plastics. In order to reach the goal of 90 percent of plastic waste diverted from landfill, an estimated 167 new facilities will be required to collect, sort and treat this additional material, while diversifying treatment pathways (chemical and thermal in addition to mechanical). Government and policy makers at all levels have a key role to play to facilitate this expansion by removing policy barriers, investing in innovation to bring technology to scale and encouraging knowledge sharing, as shown in Table 10.

Table 10: Measures to support value-recovery

Measure	Rationale
Create grant or loan programs to develop collection, sorting, or reprocessing facilities	Facilitating access to investments.
Set product or waste stream targets for collection, recovery, and/or recycling	Leading jurisdictions have utilized targets for certain waste streams to encourage and support plastics recycling initiatives.
Undertake measures that make landfilling more expensive, or otherwise ban the landfilling of plastics	Increased materials diverted through recycling facilities.
Ensure consistent and clear standards and labelling to help establish further integrated North American recycling/reprocessing capacity	Ensuring consistent and clear standards to ensure that cross-border/inter-provincial trade benefits more efficiently the Canadian/US recycling sector.
Use taxes (lower VAT rate) or other financial instruments to stimulate demand for recycled plastics	Alleviating certain barriers such as uncertain return on investment, limited resilience to shocks, and resistance to change.
Identify emerging technologies that can be applied to overcome barriers to the recycling of certain problematic waste streams	Understanding the costs of these new technologies could help inform future policy decisions and strategies for handling plastics that contain additives of concern.
Develop waste-to-energy options to treat hard-to-recycle plastics	Supporting or developing high-volume alternatives (e.g., waste to energy, industrial use such as cement kilns) for those specific waste streams that are very low value and/or highly contaminated.

As plastic waste treatment capacity grows, it will require stable flows of materials to reach economic viability. Policy makers must concurrently implement approaches that will increase the amount of plastic waste diversion (upstream – see Section 4.2) while ensuring that secondary plastics markets exist (downstream – see Section 4.1).

4.4 Increase efficiency throughout the recycling value chain

With only 13 percent of plastics being diverted from landfill, efficiency increases are needed at all steps of the value chain. Losses are recorded at the collection (incorrect sorting at the consumer level), sorting (ability of MRF to sort waste with a low contamination rate and limited losses), and reprocessing (losses in the process, contamination of input material) stages. For Canada, increasing efficiency throughout the value chain means improving the productivity and accuracy of sorting, increasing the quantity of waste recycled, and decreasing the amount of mismanaged plastic waste.

In addition to the measures presented in the sections above, policy makers could take action at several levels, as presented in Table 11.

Table 11: Measures to support efficiency throughout the recycling value chain

Measure	Rationale
Product design guidelines	Facilitating downstream collection and value recovery by creating requirements for product design (e.g., systematic use of recyclable resins, lower use of additives, easy to disassemble products). Eco-designed products could be supported through standards and preference in public procurement. These guidelines would also facilitate reuse / repair / remanufacturing.
Investment in sorting	Increasing the efficiency of recycling by investing in new sorting technology, enabling more accurate sorting of different plastic streams.
Education	Educating and engaging actors and consumers throughout the value chain to increase awareness of recycling.

These efficiency improvements are necessary to achieve zero plastic waste in Canada, since several management avenues such as advanced mechanical recycling or chemical recycling function better with a low level of contaminants.

4.5 Extend lifetime to delay waste generation

By design, many durable products cannot be repaired. Yet, the longer products containing plastics remain in use, the later these plastics will enter waste streams. Furthermore, extending product use life (including through remanufacturing) should lead to reduced demand for new products.

Although it will be difficult to reverse the trend towards single-use and disposable products, Canadian policy-makers can advocate for better quality products with longer average lifetimes. This can be supported through several approaches, as presented in Table 12.

Table 12: Measures to support product lifetime extension

Measure	Rationale
Discourage planned obsolescence	Create and communicate standards for product quality that would extend the effective life of the product by increasing the minimum legal warranty period for a given category of products or by introducing a "right to repair" that requires manufacturers to provide repair information, tools, and replacement parts to independent repair shops as well as product owners.
Encourage reuse, repair, remanufacturing and refurbishment	Explore financial incentives such as tax benefits/exemption to support repair activities and reuse of specific plastic product categories (often, disposing and buying new is cheaper than repairing, especially for low and medium-value items).
Education	Support communication campaigns that encourage repair and reuse, including labels (e.g., similar to energy star, specific labels could be developed to indicate product longevity).

4.6 Cross-cutting insights for successful implementation

The aforementioned approaches should be implemented in a concerted and systematic way, acting in several areas concurrently. However, international benchmarks from European, US and Australian case studies have demonstrated that no "one size fits all" approach exists. Due to the diverse nature of plastic applications, each sector is unique and will require a different and well-thought-out combination of efforts.

Further, policy-makers need to aim for greater harmonization at the national level. The present approach to recycling in Canada (e.g., collection schemes such as EPR, fees and tax on landfilling, provincial legislation and regulation) is fragmented and can lead to confusion. A concerted approach would bring clarity to the various stakeholders.

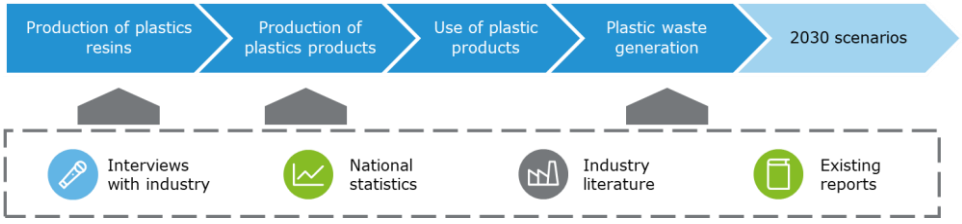
Finally, it would be beneficial to implement nation-wide monitoring of waste management and value recovery activities in order to track progress and competitiveness of the recycling industry against international benchmarks.

5. Methodology annex

5.1 Approach and scope of the study

In the absence of data covering the entire plastic value chain in Canada, a model was built to consolidate and connect the different data and information available. Figure 16 introduces the key steps of the overall approach.

Figure 16: Overall approach of the study



Source: (Deloitte, 2019a)

The resins profiled in this study (Table 13) include all key thermoplastics (plastics that can be heated, cooled and reshaped repeatedly) and thermosets (plastics that can only be shaped once due to their polymerization, which creates a three-dimensional network that cannot be remelted or solubilized).

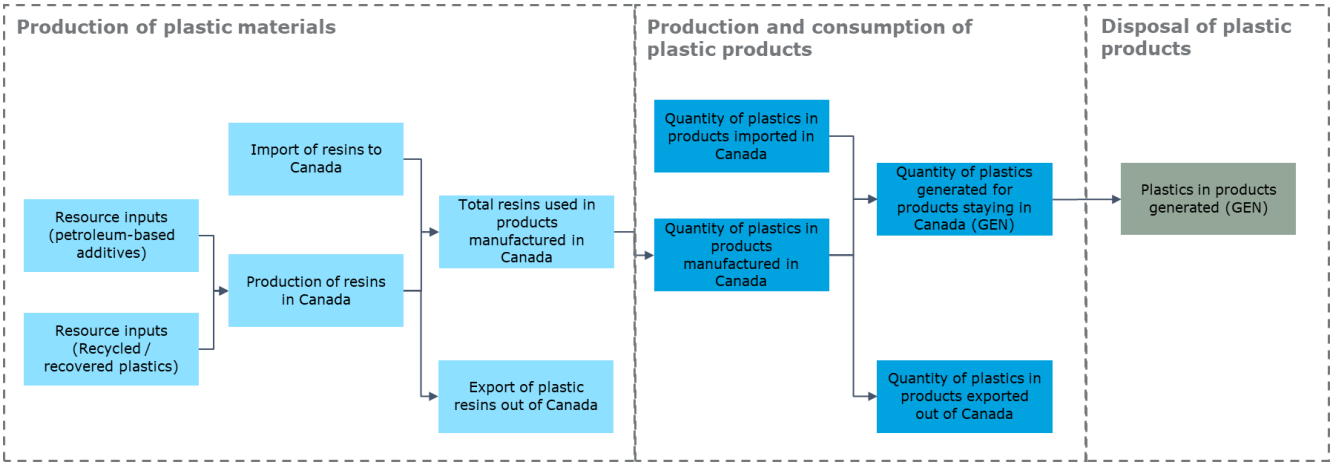
Table 13: Thermoplastic and thermosets resins profiled

Category	Resin Type	
Thermoplastics	ABS resins	Polyvinyl chloride (PVC)
	EVA copolymers	Polystyrene (PS)
	Polyamides (PA)	Polypropylene (PP)
	Polycarbonates (PC)	Polyethylene terephthalate (PET)
	Polyethylene (PE)	
Thermosets	Epoxy resins	Urea resins
	Polyurethanes (PUR)	Vinyl ester resins
	Unsaturated polyester resins	Acrylics
	Phenolic resins	

Source: (Deloitte, 2019a)

The approach taken to build the model (Figure 17) follows the plastic value chain in three phases: the production of resin and plastic materials, the production and consumption of plastic products, and plastic products' end-of-life.

Figure 17: Data flow chart of plastic products in Canada



First, a model to represent the 2016 baseline was developed, in which the various plastics products produced and traded in the Canadian economy were grouped into eight end-use sectors, defined for the purpose of this study as packaging, construction, automotive, electrical and electronic equipment, textile, white goods, agriculture and other plastics (see Section 5.2 for a description of each sector). Second, a plastic waste management model was developed to illustrate the end-of-life of plastic waste (see Section 5.3 for the detailed plastic waste management model developed for this study). Third, the models were extrapolated based on available proxies and assumptions to develop scenarios to 2030.

5.2 Sectors description

This study highlights eight sectors (also called "categories" below) that represent significant sources of plastic waste generation in Canada. Products have been grouped within those sectors based on their Supply and Use Product Classification (SUPC) code (i.e., the "MPGXXXXXX/Product Name" in the tables below).[3]

The supply and use tables include close to 500 products (i.e., unique SUPC codes). Our model considers only products related to physical goods manufactured and/or imported in Canada (SUPC codes starting with MPG). This means that other SUPC categories are excluded from our analysis, since they are not relevant in our material flow analysis (see Statistics Canada for more details on the SUPC categories[4]):

- ENExxxxxx: energy, utilities and fuels, etc.
- MPSxxxxxx: services, margins and commissions, software, etc.
- IMGxxxxxx, IMSxxxxxx: imputed codes
- FICxxxxxx: fictive materials and services, transportation margins
- NGSxxxxxx: services provided by government sector
- NNPxxxxxx: services provided by non-profit institutions serving households

[3] For more details on the SUPC codes and the concordance with other StatCan data please follow this link.
[4] For more details on the SUPC codes categories, please follow this link

In order to focus the analysis on the most material products containing plastics, a cut-off rule was applied to their plastic resin value to exclude products with a low contribution to the overall quantity of plastics generated in Canada from the analysis. The threshold chosen was CA$40 million, applied on the absolute value of the resin content in products staying in Canada. The application of this criterion was adjusted at the product level depending on various considerations, leading to the following exceptions:

- Grouping of similar products that would otherwise be excluded due to the threshold:
  - Food and non-alcoholic beverages (codes starting by MPG311 followed by 3 digits) were grouped into the MPG311XXX codes ($52 million of plastic resins staying in Canada)
- Inclusion of products that would otherwise be excluded due to the threshold, and similar to other products in existing categories and subcategories, to increase our model coverage of the economy notably for some categories and resins (polyurethane, acrylics):
  - MPG312110 / Bottled water, soft drinks and ice and MPG3121A1 / Wine and brandy were added to the Packaging – Bottles subcategory
  - MPG339905 / Signs was added to Other – Other goods
  - MPG325203 / Artificial and synthetic fibres and filaments was added to Textile
  - MPG337901 / Mattresses and foundations
- Exclusion of specific products:
  - MPG326201 / Tires, MPG326202 / Rubber and plastic hoses and belts and MPG325202 / Rubber and rubber compounds and mixtures: rubber related products were out of scope for this study
  - MPG325105 / Basic organic chemicals, n.e.c., MPG325101 / Petrochemicals, MPG3241A8 / Lubricants and other petroleum refinery products: excluded due to lack of information on the plastics used in these products
  - The cut-off rule used led to the exclusion of more than a hundred of products (codes starting with MPG), including for example:
    - MPG332500 / Builders, motor vehicle and other hardware,
    - MPG333402 / Heating and cooling equipment (except household refrigerators and freezers)
    - MPG336601 / Ships
    - MPG336900 / Other transportation equipment and related parts
    - MPG333300 / Commercial and service industry machinery
    - MPG335102 / Lighting fixtures
    - MPG334401 / Printed and integrated circuits, semiconductors and printed circuit assemblies
    - MPG336602 / Boats and personal watercraft
    - MPG333101 / Agricultural, lawn and garden machinery and equipment
    - MPG323001 / Printed products
    - MPG334A05 / Medical devicesMPG336401 / Aircraft
    - MPG336401 / Aircraft
    - MPG336403 / Aircraft parts and other aerospace equipment

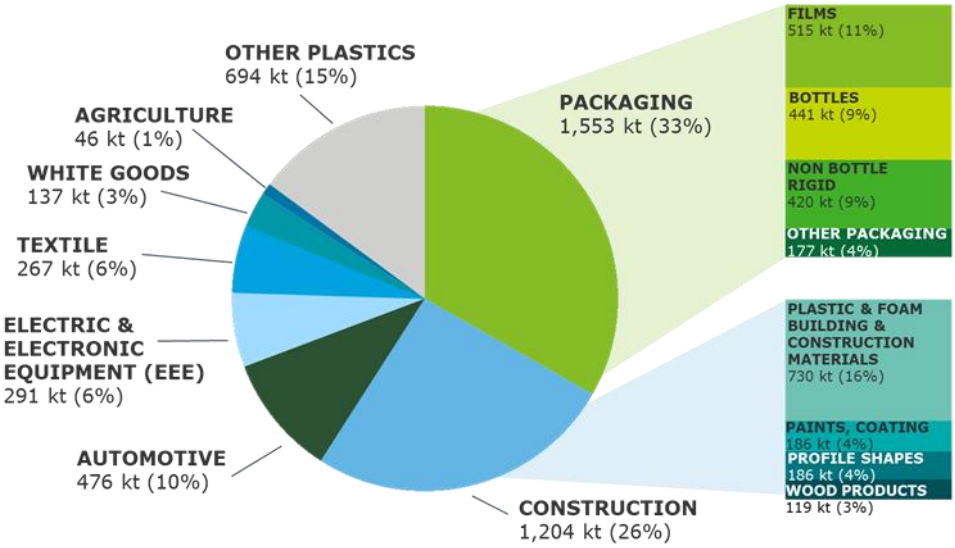
Overall, the products that were included in our model account for **88 percent of the value of plastic resins in products remaining in Canada**.

When the SUPC code was not precise enough, an additional review of Harmonized System (HS) products falling under the SUPC code was applied to assess where the code should be categorized. This additional review was conducted using HS 2017 to SUPC 2013 concordance table provided by StatCan. In very few instances, trade data related with one SUPC code was split between two customized product categories to reflect clearly distinct sector affiliation and waste management fate (e.g., MPG 335901/Batteries was split between the automotive sector for car batteries and the EEE sector for primary cells and batteries).

For some sectors, it was deemed necessary to create subcategories to provide a more granular view of key products and to reflect differences in waste management within sectors. This decision was based on information gathered on key products for each sector and their respective waste management. For example, the fate of plastic bottles was considered to be different from that of plastic films in the packaging category. Likewise, the existence of extended producer responsibility systems applicable to select products within a given sector triggered the creation of distinct sub-categories within the sector (e.g., EEE sector).

Sectors and their respective subcategories are detailed in Figure 18.

Figure 18: End-use markets for plastic products in Canada (kt, 2016)



Source: Deloitte

Packaging

Plastic packaging is commonly used to protect, preserve, store and transport products, and is the main category in terms of the end market for plastic products. It regroups films (including plastic bags), bottles and other items for sectors including food and beverage, healthcare, consumer packaged goods, and cosmetics and personal care among countless other applications.

Table 14: Main subcategories and products, category "packaging"

Category	Subcategory	Product
Packaging	Packaging – Film	MPG326102 / Plastic films and non-rigid sheets MPG326101 / Plastic bags
	Packaging – Bottles	MPG326109 / Plastic products, n.e.c. MPG326106 / Plastic bottles MPG312110 / Bottled water, soft drinks and ice MPG3121A1 / Wine and brandy
	Packaging – Non-bottle rigid	MPG326109 / Plastic products, n.e.c. MPG311XXX / Miscellaneous food products MPG325601 / Soaps and cleaning compounds MPG325400 / Pharmaceutical and medicinal products MPG325602 / Perfumes and toiletries
	Packaging – Other packaging	MPG326105 / Foam products (except for construction) MPG322209 / Other converted paper products MPG322201 / Paperboard containers MPG335901 / Batteries

Construction

Plastic has a variety of uses in the construction industry due to its strength and durability, despite being lightweight. This includes resins used in paints and coatings, profile shapes (e.g., windows and doors) and pipes, insulation board and foam, plastics used in reconstituted wood and plywood, and other generic products used in construction. Thermoplastics are often used in flooring and window covering applications. Resins and adhesives produced by this industry are used in the creation of polyvinyl chloride (PVC) pipes, flooring, insulation, roofing, windows and doors.

Note there is a large portion of plastic from the construction sector that is 'stocked' in buildings, and will likely enter waste stream more than 30 years later.

Table 15: Main subcategories and products, category "construction"

Category	Subcategory	Product
Construction	Construction – Generic	MPG326103 / Plastic and foam building and construction materials
	Construction – Paints, coatings	MPG325500 / Paints, coatings and adhesive products
	Construction – Profiles shapes & pipe fitting	MPG326104 / Plastic profile shapes MPG332A02 / Metal valves and pipe fittings
	Construction – Reconstituted wood products, plywood & veneer	MPG321203 / Reconstituted wood products MPG321201 / Veneer and plywood MPG321202 / Wood trusses and engineered wood members

Automotive

Plastic in the automotive sector accounts for between 8 and 10 percent of the vehicle weight and is constantly increasing as automobile manufacturers are replacing steel and aluminum parts with plastic parts that help to make automobiles lighter and more fuel efficient. Motor vehicle manufacturers typically use plastic and resin inputs in the creation of automotive parts (e.g., bumper, tanks and fluid containers) and interior components (e.g., seats, dashboard).

Table 16: Main subcategories and products, category "automotive"

Category	Subcategory	Product
Automotive	Vehicles – Generic	MPG326107 / Motor vehicle plastic parts
		MPG336360 / Motor vehicle interior trim, seats and seat parts
		MPG336390 / Other miscellaneous motor vehicle parts
		MPG336370 / Motor vehicle metal stamping
		MPG336320 / Motor vehicle electrical and electronic equipment and instruments
		MPG336120 / Medium and heavy-duty trucks and chassis
		MPG336330 / Motor vehicle steering and suspension components
		MPG336350 / Motor vehicle transmission and power train parts
		MPG336111 / Passenger cars
		MPG336112 / Light-duty trucks, vans and sport utility vehicles (SUVs)
		MPG335901 / Batteries

Electric and electronic equipment (EEE)

Plastics in the Electric and electronic equipment (EEE) sector include two subcategories:

- Products such as computers, phones, printers, and audio-video devices were grouped into an "Electronic Products Recycling Association" (EPRA[5]) subcategory as they are most likely targeted by an EPR scheme in Canada.
- Products such as electric wire, cables and other components were grouped into a "generic" subcategory and are most likely not covered by an EPR scheme in Canada.

Table 17: Main subcategories and products, category "EEE"

Category	Subcategory	Product
EEE	EEE – EPRA	MPG335903 / Wiring devices
		MPG334201 / Telephone apparatus
		MPG334100 / Computers, computer peripherals and parts
		MPG334209 / Other communications equipment
	EEE – Generic	MPG335902 / Communication and electric wire and cable
		MPG335909 / Other electrical equipment and components

[5] For more information, please visit EPRA website.

Textile

The plastic from textiles is comprised of artificial fibres such as polyester and nylon. The category also includes textiles for furniture, and fibres from carpets, rugs and mats.

Table 18: Main subcategories and products, category "textile"

Category	Subcategory	Product
Textile	Textile – Generic	MPG31B001 / Men's, women's, boys' and girls' clothing
		MPG31A002 / Fabrics
		MPG31A004 / Other textile furnishings
		MPG31A005 / Textile products, n.e.c.
		MPG31A003 / Carpets, rugs and mats
		MPG31B005 / Footwear
		MPG325203 / Artificial and synthetic fibres and filaments

White goods

The white goods sector refers to large appliances such as fridges and stoves, as well as small household appliances such as food processors and electric kettles.

Table 19: Main subcategories and products, category "white goods"

Category	Subcategory	Product
White goods	White goods – Generic	MPG335204 / Major appliances
		MPG335203 / Small electric appliances

Agriculture

The agricultural sector accounts for the plastic used for the transportation of grains and seeds, fertilizer and pesticide packaging, and agricultural films. Due to the lack of a specific category focusing on agricultural plastics, the model used a portion of the plastic films and non-rigid sheets category. This portion was estimated based on the amount of agricultural plastic waste generated in Canada (CleanFarms estimate) extrapolated to obtain the quantity of agricultural plastic products staying in Canada.

Table 20: Main subcategories and products, category "agriculture"

Category	Subcategory	Product
Agriculture	Agriculture – Generic	MPG326102 / Plastic films and non-rigid sheets

Other plastics

The "other plastics" sector aggregates the diversity of product categories that could not be categorized elsewhere. This heterogeneous category includes plastics such as chemical products and resins, plastics used in medical, dental and personal care, toys, household furniture, sporting goods, mattresses, and industrial machinery.

Table 21: Main subcategories and products, category "other plastics"

Category	Subcategory	Product
Other	Other – Miscellaneous chemical, resins, organic chemicals, petrochemicals	MPG325900 / Chemical products, n.e.c. MPG325201 / Plastic resins
	Other – Other goods	MPG339100 / Medical, dental and personal safety supplies, instruments and equipment MPG339909 / Other miscellaneous manufactured products MPG339903 / Toys and games MPG337102 / Household furniture MPG339902 / Sporting and athletic goods MPG339901 / Jewellery and silverware MPG327A02 / Glass (including automotive), glass products and glass containers MPG339905 / Signs MPG337901 / Mattresses and foundations
	Other – Machinery	MPG333200 / Other industry-specific machinery MPG333102 / Logging, mining and construction machinery and equipment MPG333909 / Other miscellaneous general-purpose machinery

Table 22 provides an overview of the main products containing plastics included in the categories or "sectors" developed for this study.

Table 22: Description of sectors for end-market products containing plastic

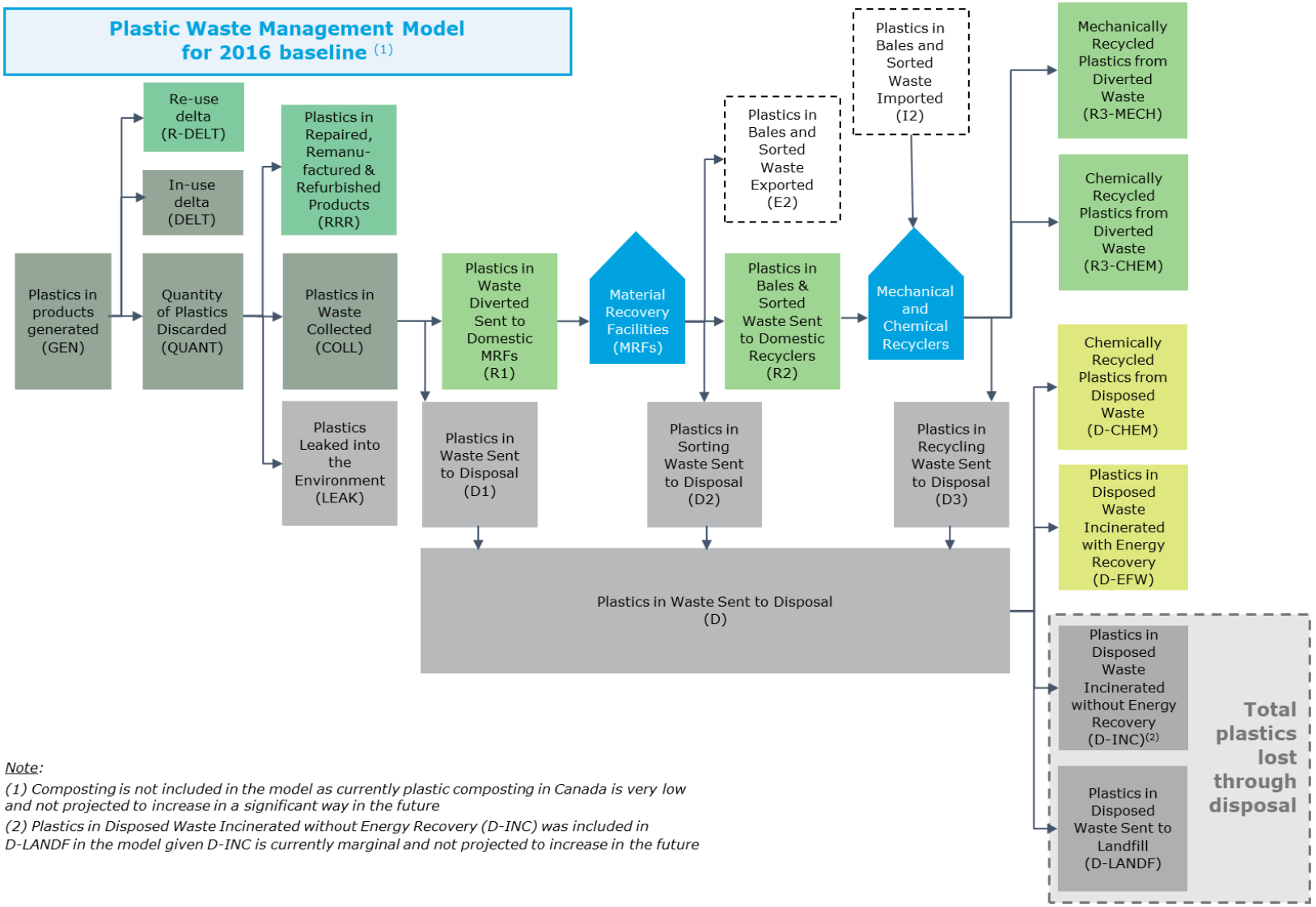
Sector	Type of plastic products
Packaging	Includes films (e.g., plastic bags), bottles and other items, for sectors such as food and beverage, healthcare, consumer packaged goods, and cosmetics and personal care.
Construction	Includes resins used in paints and coatings, profile shapes (e.g., windows and doors) and pipes, insulation board and foam, plastics used in reconstituted wood and plywood, and other generic products used in construction.
Automotive	Comprises plastic parts such as the bumper, tanks and fluid containers, and the plastic components inside the passenger compartment, seats and dashboard.
Electric and electronic equipment (EEE)	Parts in electronics such as computers, phones, printers, audio-video devices, and items such as electric wire, cables and other components.
Textile	Artificial fibres such as polyester and nylon. Also includes textile for furniture, and fibres from carpets, rugs and mats.
White goods	Plastic contained in large appliances such as fridges and stoves, and small household appliances including food processors and electric kettles.

Sector	Type of plastic products
Agriculture	Plastic used for grains and seeds transportation, fertilizer and pesticide packaging, and agricultural films.
Other plastics	This heterogeneous category includes plastics such as chemical products and resins, plastics used in medical, dental and personal care, toys, household furniture, sporting goods, mattresses, and industrial machinery.

5.3 Description of the plastic waste management model

Figure 19 presents a flow chart of the lifecycle of plastic waste in Canada, as modelled in this study, while Table 23 defines the terms used.

Figure 19: Flow chart of plastic waste in Canada



Note:

(1) Composting is not included in the model as currently plastic composting in Canada is very low and not projected to increase in a significant way in the future

(2) Plastics in Disposed Waste Incinerated without Energy Recovery (D-INC) was included in D-LANDF in the model given D-INC is currently marginal and not projected to increase in the future

Table 23: Legend of terms used in the flow chart of plastic waste in Canada

Acronym	Definition	Key assumptions and hypothesis	Reference
GEN	Quantity of plastics in products generated in Canada.	Approach taken to build this model leveraged StatCan's Supply and Use Tables (SUT) to assess the generation (i.e., arrival on the Canadian market) of products containing plastic. There are two main sources for those products: domestically manufactured products using plastic resins and net imports of finished or semi-finished goods containing plastic.	(Deloitte, 2019a)
DELT	The in-use delta measures the difference between the plastic products generation for a product category in a given year and the estimated plastic waste generation of that same product category for the same year, before taking into account any additional re-use (see R-DELT below).	The in-use delta is based on the average product category lifetime, the past annual sector market growth during that product category lifetime, and the evolution of the average plastic content in that product category over its lifetime. The in-use delta impacts the automotive, construction and EEE sectors the most, due to relatively long product lifetimes.	(Deloitte, 2019a)
R-DELT	Direct re-use is a way to extend the expected end-of-use of products by a certain amount of time. As such, the re-use delta models the fact that a reused product enters the waste stream later than an average non-reused product.	The re-use delta is modelled in a similar way to the in-use delta (DELT). Its calculation is based on an average additional product lifetime of 50 percent, the past annual sector market growth during that lifetime, and an estimation of the applicable re-use rate within each sector.	(Deloitte, 2019a)
QUANT	Quantity of plastics discarded represents the plastic entering waste streams.	It is equal to the quantity of plastics in product generated in Canada (GEN) minus the in-use and re-use deltas.	(Deloitte, 2019a)
RRR	Plastics in repaired, remanufactured and refurbished products (RRR). Remanufacturing and comprehensive refurbishment take place within industrial or factory settings and result in quasi-new products, with a full-service life identical to a new product, for which production is avoided.	Currently, it is not certain that RRR activities occur on a large scale in Canada for products containing plastics. Accordingly, RRR is not quantified in the 2016 baseline model. However, RRR is considered in 2030 scenarios of the Task 2 report, in which they have a direct impact on plastic waste diversion.	(Deloitte, 2019a) and (Deloitte, 2019b)

Acronym	Definition	Key assumptions and hypothesis	Reference
LEAK	Plastics leaked permanently into the environment.	Litter can be split into fractions, the first of which is temporary and eventually captured by municipal waste collection (e.g. when cleaning streets and parks). In the model, this fraction is included in the plastics in waste sent to disposal (D1). The second fraction of plastics littered is never collected and considered to be permanently lost into the environment. This second fraction, also called plastics leaked into the environment (LEAK) is estimated in the model. Global estimates of plastic leakage into the environment were prepared by Jambeck et al. in 2015. In this study, the authors estimated that approximately 10,000 tonnes of plastic waste were mismanaged in coastal areas and nearly 29,000 tonnes across Canada.	(Deloitte, 2019a)
COLL	Plastics in waste collected, which are either sent to a sorting facility (R1) or to disposal (D1).	Plastics in waste collected is equal to the after-use quantity (QUANT) minus the plastic leaked into the environment (LEAK) and plastics in repaired, remanufactured and refurbished products (RRR). It is also equal to R1 + D1.	(Deloitte, 2019a)
R1	Plastics in waste diverted and sent to domestic MRFs.	It is calculated using a diversion rate based on information gathered in the Task 3 report and additional references. R1 = R2 + D2 + E2	(Deloitte, 2019a)
D1	Plastics in waste sent to disposal.	It is calculated based on the current rates presented by StatCan and research from Cheminfo	(Deloitte, 2019a)
R2	Plastics in bales and sorted waste sent to domestic recyclers.	Calculated based on the sector-specific sorting yield (R2/R1). Yields were sourced from studies such as MORE (packaging), Ontario Electronic Stewardship (EEE), or estimations relying on literature reviews and benchmarks. Another equation involving R2 is: I2 + R2 = R3 + D3	(Deloitte, 2019a)
D2	Plastics in waste sent to disposal by MRFs. Represents the fraction rejected by the sorting facilities.	D2 is deducted using R1 and R2, given that we have D2 = R1 – R2 – E2. However, as E2 was not quantified in the model, we have D2 = R1 – R2.	(Deloitte, 2019a)
E2	Plastics in bales and sorted waste exported.	Documented in Task 3 report but not quantified in the model as some information was missing on a resin by resin basis.	(Deloitte, 2019c)
I2	Plastics in bales and sorted waste imported.	Documented in Task 3 report but not quantified in the model as some information was missing on a resin by resin basis.	(Deloitte, 2019c)

Acronym	Definition	Key assumptions and hypothesis	Reference
R3	Recycled plastic from diverted waste.	Based on the reprocessing yield (R3/R2), which refers to the efficiency of recycling operations. It is a combination of chemical and recycling yields. With the exception of EEE waste, for which recycling efficiency was available, reprocessing yields were assessed at the resin level and derived from current recycling operations or sourced from other comparable jurisdictions (e.g., Europe) when no Canadian data was available. It is also equal to R3 MECH+ R3 CHEM	(Deloitte, 2019a)
R3-MECH	Mechanically recycled plastic from diverted waste.	Stemming from the diverted waste stream, these plastics are mechanically reprocessed into flakes or pellets, ready for incorporation as recycled resins by plastic products manufacturers or resin compounders. This currently represents the dominant output of municipal recycling programs across the country.	(Deloitte, 2019a)
R3-CHEM	Chemically recycled plastic from diverted waste.	Stemming from the diverted waste stream, these plastics are chemically converted into shorter molecules, ready to be used to produce new plastics or fuels. Given low contamination levels of input material, chemical recycling from diverted waste usually attempts to convert most of the received feedstock into the monomer state of the original polymer resin, in order to generate the highest possible revenue. By-products are usually other chemicals or fuels.	(Deloitte, 2019c)
D3	Plastics in recycling waste sent to disposal; represents the fraction rejected by the recyclers.	Based on the reprocessing yield.	(Deloitte, 2019a)
D	Total plastics in waste sent to disposal.	Some recovery can still occur whether through chemical recycling (D-CHEM) or incineration with energy recovery (D-EFW). The rest is either incinerated without energy recovery (D-INC) or landfilled (D-LANDF). D = D1 + D2 + D3.	(Deloitte, 2019a)
D-CHEM	Chemically recycled plastic from disposed waste.	Stemming from the disposed waste stream, mainly from municipal solid waste (MSW), these plastics are chemically converted into fuels such as methanol, ethanol, diesel, and other related chemicals. Given the relatively high contamination level of the input material, chemical recycling from disposed waste does not usually directly return to monomers as R3-CHEM sometimes does.	(Deloitte, 2019c)
D-EFW	Plastics in disposed waste incinerated with energy recovery.	Also called thermal recovery, this stream accounted for the vast majority of thermal treatment of plastics in Canada with 134.5 kt in 2016 (the other avenue being incineration without energy recovery). Most facilities use an energy recovery approach as plastics have relatively high caloric values relative to other waste materials and relative to some conventional fuels (e.g., PE, PP and PS have energy content 50 percent higher than coal). Most of the current treatment capacity originates from five waste-to-energy facilities, one treatment centre, and (to a lesser extent) cement plants.	(Deloitte, 2019c)

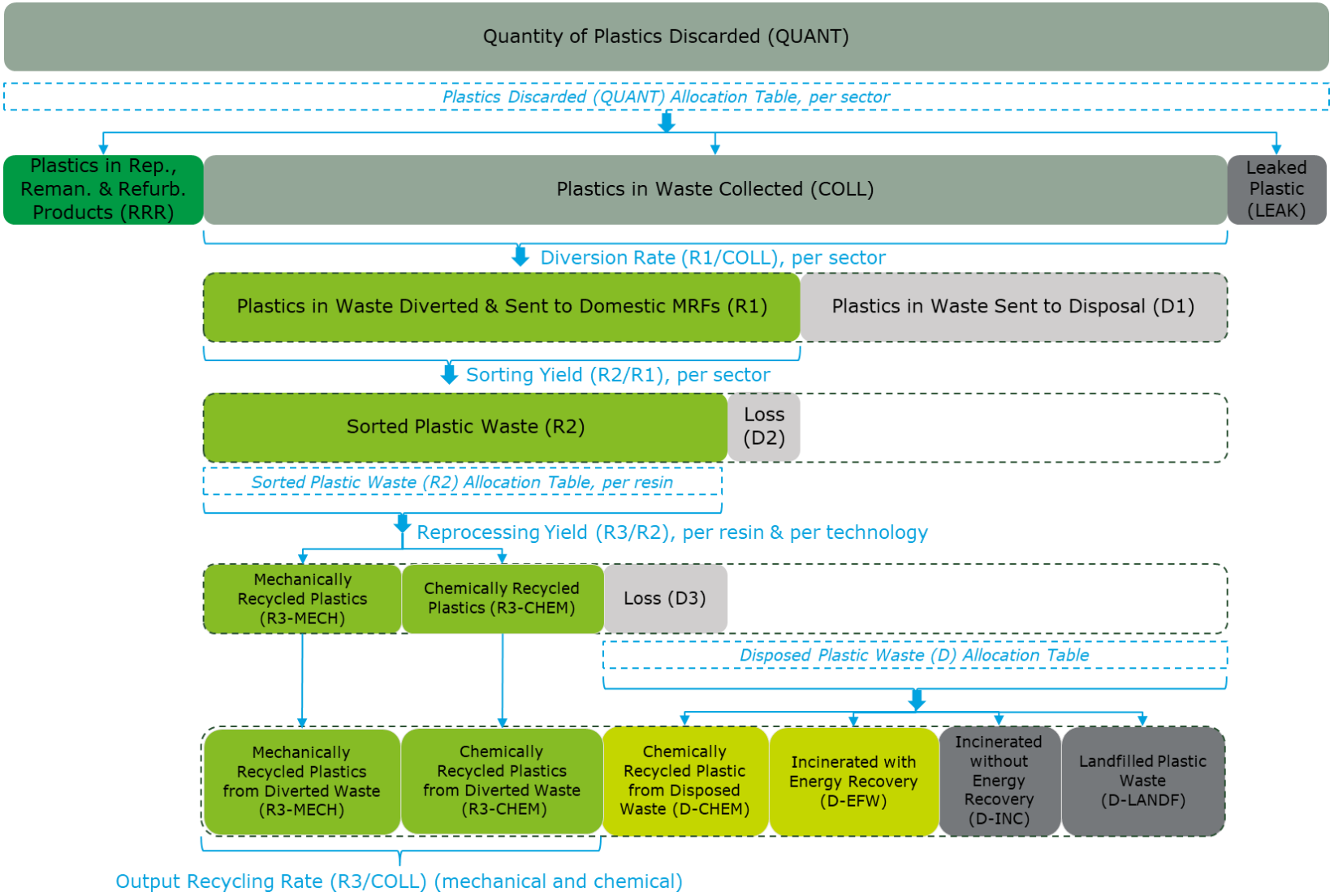
Acronym	Definition	Key assumptions and hypothesis	Reference
D-INC	Plastics in disposed waste incinerated without energy recovery.	Incineration without energy recovery accounted for less than two percent of thermal treatment for plastics in 2016. Only one site in Canada (Lévis, built in 1976) is known to incinerate municipal solid waste without energy recovery. Given the small amount, D-INC values have not been singled out in the model and were rather included in D-EFW.	(Deloitte, 2019c)
D-LANDF	Plastics in disposed waste sent to landfill.	Based on the material flow model. The amount landfilled is the difference between after-use quantities (QUANT) and each of the above life cycle stages. StatCan's information on disposal in Canada was also used as a benchmark and data validation source.	(Deloitte, 2019a)

The study has also defined some rates and yields for clear recognition of the performance level presented in the study. These are presented in Table 24 and illustrated in Figure 20.

Table 24: Acronyms of rates and yields used in the waste management model

Acronym	Definition
R1/COLL	Diversion rate, or the share of plastic diverted from direct disposal and sent to a sorting facility, divided by COLL. This rate is assessed by sector.
R2/COLL	Output sorting rate, or the share of plastic sorted by sorting facilities and sent to a reprocessing facility, divided by COLL. This rate is assessed by sector.
R3/COLL	Output recycling rate, or the share of plastic that is ultimately reprocessed, whether through chemical or mechanical recycling from diverted waste, divided by COLL. This rate does not include D-CHEM.
(R3+D-CHEM+ D-EFW)/COLL	Value recovery rate, or the share of plastic that is ultimately value recovered whether through chemical or mechanical recycling from diverted and disposed waste or through thermal recovery, divided by COLL.
R2/R1	Sorting yield, or the amount of plastics MRFs were able to sort out and send to reprocessing facilities, divided by the total amount of unsorted plastic received. This yield is affected by factors including the quality of input waste material, contamination, type of plastics received, and sorting technologies and equipment. It illustrates the efficiency of sorting operations, and is assessed by waste stream category or sector.
R3/R2	Reprocessing yield, or the amount of recycled materials (e.g., flakes or pellets of recycled resins, monomers) reprocessing facilities were able to produce and send to end-users, divided by the total amount of sorted plastics waste received from MRFs. It illustrates the recycling efficiency of reprocessing operations, and is assessed by resin and technology (chemical or mechanical).

Figure 20: Key steps of the waste management model



5.4 Key assumptions and limitations

Considering the range of resins included in this study (both thermoplastics and thermosets), the scope of this study is wider than most other studies conducted on plastic in other jurisdictions, which tend to focus on specific sectors (packaging in particular) and are usually limited to (a selection of) thermoplastics only. This has an influence on the calculated rates and yields presented in this study and should be considered when comparing performance between jurisdictions.

6. Bibliography

Deloitte. (2019a). *Economic study of the Canadian plastic industry, markets and waste- Task 1.* Government of Canada, Environment and Climate Change Canada Internal Report.

Deloitte. (2019b). *Economic study of the Canadian plastic industry, markets and waste-Task 2.* Government of Canada, Environment and Climate Change Canada Internal Report.

Deloitte. (2019c). *Economic study of the Canadian plastic industry, markets and waste-Task 3.* Government of Canada, Environment and Climate Change Canada Internal Report.

Deloitte. (2019d). *Economic study of the Canadian plastic industry, markets and waste-Task 4.* Government of Canada, Environment and Climate Change Canada Internal Report.

More Recycling. (2018). *2016 Post-Consumer Plastics Recycling in Canada.*

Deloitte.

www.deloitte.ca

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REFERENCES AND NOTES

1. Z. Kielan-Jaworowska, R. L. Cifelli, Z.-X. Luo, *Mammals from the Age of Dinosaurs: Origins, Evolution, and Structure* (Columbia Univ. Press, New York, 2004).
2. J. A. Lillegraven, G. Krusat, *Rocky Mount. Geol.* **28**, 39–138 (1991).
3. D. Sigogneau-Russell, *Acta Palaeontol. Pol.* **48**, 357–374 (2003).
4. Q. Ji, Z. X. Luo, C. X. Yuan, A. R. Tabrum, *Science* **311**, 1123–1127 (2006).
5. Z.-X. Luo, T. Martin, *Bull. Carnegie Mus. Nat. Hist.* **39**, 27–47 (2007).
6. A. O. Averianov, A. V. Lopatin, S. A. Krasnolutskii, S. V. Ivantsov, *Proc. Zool. Inst. Russian Acad. Sci.* **314**, 121–148 (2010).
7. Y.-M. Hu, J. Meng, J. M. Clark, *Vertebr. Palasiat.* **45**, 173–194 (2007).
8. T. Martin, A. O. Averianov, H.-U. Pfretzschner, *Palaeobiodivers. Palaeoenviron.* **90**, 295–319 (2010).
9. G. W. Rougier, A. S. Sheth, K. Carpenter, L. Appella-Guiscafre, B. M. Davis, *J. Mamm. Evol.* (2014).
10. C.-F. Zhou, S. Wu, T. Martin, Z. X. Luo, *Nature* **500**, 163–167 (2013).
11. T. B. Rowe, T. E. Macrini, Z. X. Luo, *Science* **332**, 955–957 (2011).
12. Z.-X. Luo, *Annu. Rev. Ecol. Evol. Syst.* **42**, 355–380 (2011).
13. J. Meng, Y. Hu, Y. Wang, X. Wang, C. Li, *Nature* **444**, 889–893 (2006).
14. Z.-X. Luo, *Nature* **450**, 1011–1019 (2007).
15. G. W. Rougier, S. Apesteguía, L. C. Gaetano, *Nature* **479**, 98–102 (2011).
16. P. G. Gill *et al.*, *Nature* **512**, 303–305 (2014).
17. M. Chen, Z.-X. Luo, *J. Mamm. Evol.* **20**, 159–189 (2012).
18. Materials and methods and supplementary text are available as supplementary materials on *Science* Online.
19. P. Hershkovitz, *New World Monkeys (Platyrrhini)* (Univ. Chicago Press, Chicago, 1977).
20. G. P. Wilson *et al.*, *Nature* **483**, 457–460 (2012).
21. T. Martin, *Zool. J. Linn. Soc.* **145**, 219–248 (2005).
22. Z.-X. Luo, Q. Ji, C. X. Yuan, *Nature* **450**, 93–97 (2007).
23. V. Weisbecker, D. I. Warton, *J. Morphol.* **267**, 1469–1485 (2006).
24. E. C. Kirk, P. Lemelin, M. W. Hamrick, D. M. Boyer, J. I. Bloch, *J. Hum. Evol.* **55**, 278–299 (2008).
25. N. MacLeod, K. D. Rose, *Am. J. Sci.* **293** (A), 300–355 (1993).
26. Z. Kielan-Jaworowska, P. P. Gambaryan, *Fossils Strata* **36**, 1–92 (1994).
27. F. A. Jenkins Jr., F. R. Parrington, *Philos. Trans. R. Soc. Lond.* **273**, 387–431 (1976).
28. C. Argot, *J. Morphol.* **247**, 51–79 (2001).
29. E. J. Sargis, *Evol. Anthropol.* **13**, 56–66 (2004).
30. J. Lessertisseur, R. Saban, in *Traité de Zoologie. Tome XVI (Fascicule I). Mammifères: Téguments et Squelette*, P.-P. Grassé, Ed. (Masson, Paris, 1967), pp. 587-675.

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SUPPLEMENTARY MATERIALS

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Materials and Methods
Supplementary Text
Figs. S1 to S8
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References (*31*–*80*)

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MARINE POLLUTION

Plastic waste inputs from land into the ocean

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Plastic debris in the marine environment is widely documented, but the quantity of plastic entering the ocean from waste generated on land is unknown. By linking worldwide data on solid waste, population density, and economic status, we estimated the mass of land-based plastic waste entering the ocean. We calculate that 275 million metric tons (MT) of plastic waste was generated in 192 coastal countries in 2010, with 4.8 to 12.7 million MT entering the ocean. Population size and the quality of waste management systems largely determine which countries contribute the greatest mass of uncaptured waste available to become plastic marine debris. Without waste management infrastructure improvements, the cumulative quantity of plastic waste available to enter the ocean from land is predicted to increase by an order of magnitude by 2025.

Reports of plastic pollution in the ocean first appeared in the scientific literature in the early 1970s, yet more than 40 years later, no rigorous estimates exist of the amount and origin of plastic debris entering the marine environment. In 1975, the estimated annual flux of litter of all materials to the ocean was 6.4 million tons [5.8 million metric tons (MT)], based only on discharges from ocean vessels, military operations, and ship casualties (*1*). The discharge of plastic from at-sea vessels has since been banned (*2*), but losses still occur. It is widely cited that 80% of marine debris originates from land; however, this figure is not well substantiated and does not inform the total mass of debris entering the marine environment from land-based sources.

Plastics have become increasingly dominant in the consumer marketplace since their commercial development in the 1930s and 1940s. Global plastic resin production reached 288 million MT in 2012 (*3*), a 620% increase since 1975. The largest market sector for plastic resins is packaging (*3*); that is, materials designed for immediate disposal. In 1960, plastics made up less than 1% of municipal solid waste by mass in the United States (*4*); by 2000, this proportion increased by an order of magnitude. By 2005, plastic made up at least 10% of solid waste by mass in 58% (61 out of 105) of countries with available data (*5*).

Plastics in the marine environment are of increasing concern because of their persistence and effects on the oceans, wildlife, and, potentially, humans (*6*). Plastic debris occurs on coastlines, in Arctic sea ice, at the sea surface, and on the sea floor (*7*, *8*). Weathering of plastic debris causes fragmentation into particles that even small marine invertebrates may ingest (*9*). Its small size also renders this debris untraceable to its source and extremely difficult to remove from open ocean environments, suggesting that the most effective mitigation strategies must reduce inputs.

We estimated the annual input of plastic to the ocean from waste generated by coastal populations worldwide. We defined mismanaged waste as material that is either littered or inadequately disposed. Inadequately disposed waste is not formally managed and includes disposal in dumps or open, uncontrolled landfills, where it is not fully contained. Mismanaged waste could eventually enter the ocean via inland waterways, wastewater outflows, and transport by wind or tides. Estimates of the mass of plastic waste carried by particular waterways range from <<1 kg per day (Hilo, HI) to 4.2 MT (4200 kg) per day (Danube River) (*10*, *11*). Because of their dependence on local watershed characteristics, these results cannot be easily extrapolated to a global scale.

Here we present a framework to calculate the amount of mismanaged plastic waste generated annually by populations living within 50 km of a coast worldwide that can potentially enter the ocean as marine debris. For each of 192 coastal countries with at least 100 permanent residents that border the Atlantic, Pacific, and Indian oceans and the Mediterranean and Black seas, the framework includes: (i) the mass of waste generated per capita annually; (ii) the percentage of waste that is plastic; and (iii) the percentage of plastic waste that is mismanaged and,

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therefore, has the potential to enter the ocean as marine debris (*12*) (data S1). By applying a range of conversion rates from mismanaged waste to marine debris, we estimated the mass of plastic waste entering the ocean from each country in 2010, used population growth data (*13*) to project the increase in mass to 2025, and predicted growth in the percentage of waste that is plastic. Lacking information on future

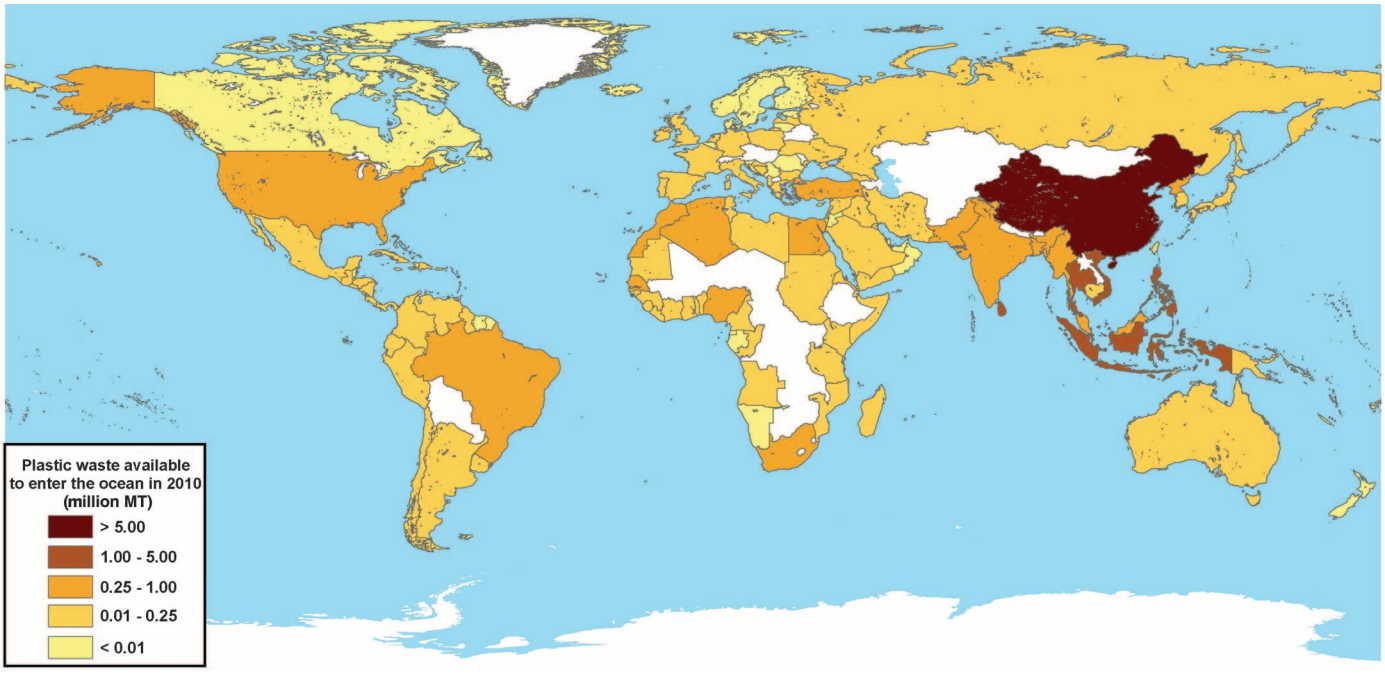

Fig. 1. Global map with each country shaded according to the estimated mass of mismanaged plastic waste [millions of metric tons (MT)] generated in 2010 by populations living within 50 km of the coast. We considered 192 countries. Countries not included in the study are shaded white.

Table 1. Waste estimates for 2010 for the top 20 countries ranked by mass of mismanaged plastic waste (in units of millions of metric tons per year). Econ classif., economic classification; HIC, high income; UMI, upper middle income; LMI, lower middle income; LI, low income (World Bank definitions based on 2010 Gross National Income). Mismanaged waste is the sum of inadequately managed waste plus 2% littering. Total mismanaged plastic waste is calculated for populations within 50 km of the coast in the 192 countries considered. pop., population; gen., generation; ppd, person per day; MMT, million metric tons.

Rank	Country	Econ. classif.	Coastal pop. [millions]	Waste gen. rate [kg/ppd]	% plastic waste	% mismanaged waste	Mismanaged plastic waste [MMT/year]	% of total mismanaged plastic waste	Plastic marine debris [MMT/year]
1	China	UMI	262.9	1.10	11	76	8.82	27.7	1.32–3.53
2	Indonesia	LMI	187.2	0.52	11	83	3.22	10.1	0.48–1.29
3	Philippines	LMI	83.4	0.5	15	83	1.88	5.9	0.28–0.75
4	Vietnam	LMI	55.9	0.79	13	88	1.83	5.8	0.28–0.73
5	Sri Lanka	LMI	14.6	5.1	7	84	1.59	5.0	0.24–0.64
6	Thailand	UMI	26.0	1.2	12	75	1.03	3.2	0.15–0.41
7	Egypt	LMI	21.8	1.37	13	69	0.97	3.0	0.15–0.39
8	Malaysia	UMI	22.9	1.52	13	57	0.94	2.9	0.14–0.37
9	Nigeria	LMI	27.5	0.79	13	83	0.85	2.7	0.13–0.34
10	Bangladesh	LI	70.9	0.43	8	89	0.79	2.5	0.12–0.31
11	South Africa	UMI	12.9	2.0	12	56	0.63	2.0	0.09–0.25
12	India	LMI	187.5	0.34	3	87	0.60	1.9	0.09–0.24
13	Algeria	UMI	16.6	1.2	12	60	0.52	1.6	0.08–0.21
14	Turkey	UMI	34.0	1.77	12	18	0.49	1.5	0.07–0.19
15	Pakistan	LMI	14.6	0.79	13	88	0.48	1.5	0.07–0.19
16	Brazil	UMI	74.7	1.03	16	11	0.47	1.5	0.07–0.19
17	Burma	LI	19.0	0.44	17	89	0.46	1.4	0.07–0.18
18*	Morocco	LMI	17.3	1.46	5	68	0.31	1.0	0.05–0.12
19	North Korea	LI	17.3	0.6	9	90	0.30	1.0	0.05–0.12
20	United States	HIC	112.9	2.58	13	2	0.28	0.9	0.04–0.11

*If considered collectively, coastal European Union countries (23 total) would rank eighteenth on the list

global infrastructure development, the projection represents a business-as-usual scenario.

We estimate that 2.5 billion MT of municipal solid waste was generated in 2010 by 6.4 billion people living in 192 coastal countries (93% of the global population). This estimate is broadly consistent with an estimated 1.3 billion MT of waste generated by 3 billion people in urban centers globally (*5*). Approximately 11% (275 million MT) of the waste generated by the total population of these 192 countries is plastic. We expect plastic waste to roughly track plastic resin production (270 million MT in 2010) (*3*), with differences resulting from the time lag in disposal of durable goods (lifetime of years to decades), for example. Scaling by the population living within 50 km of the coast (those likely to generate most of the waste becoming marine debris), we estimate that 99.5 million MT of plastic waste was generated in coastal regions in 2010. Of this, 31.9 million MT were classified as mismanaged and an estimated 4.8 to 12.7 million MT entered the ocean in 2010, equivalent to 1.7 to 4.6% of the total plastic waste generated in those countries.

Our estimate of plastic waste entering the ocean is one to three orders of magnitude greater than the reported mass of floating plastic debris in high-concentration ocean gyres and also globally (*14*–*17*). Although these ocean estimates represent only plastics that are buoyant in seawater (mainly polyethylene and polypropylene), in 2010 those resins accounted for 53% of plastic production in North America and 66% of plastic in the U.S. waste stream (*4*, *18*). Because no global estimates exist for other sources of plastic into the ocean (e.g., losses from fishing activities or at-sea vessels, or input from natural disasters), we do not know what fraction of total plastic input our land-based waste estimate represents.

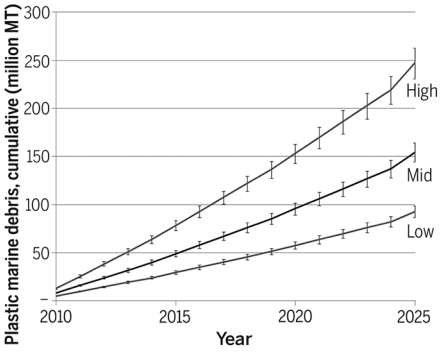

Fig. 2. Estimated mass of mismanaged plastic waste (millions of metric tons) input to the ocean by populations living within 50 km of a coast in 192 countries, plotted as a cumulative sum from 2010 to 2025. Estimates reflect assumed conversion rates of mismanaged plastic waste to marine debris (high, 40%; mid, 25%; low, 15%). Error bars were generated using mean and standard error from the predictive models for mismanaged waste fraction and percent plastic in the waste stream (*12*).

Our framework was designed to compute, from the best-available data, an order-of-magnitude estimate of the amount of mismanaged plastic waste potentially entering the ocean worldwide. It is also a useful tool to evaluate the factors determining the largest sources of mismanaged plastic waste. The amount of mismanaged plastic waste generated by the coastal population of a single country ranges from 1.1 MT to 8.8 million MT per year, with the top 20 countries' mismanaged plastic waste encompassing 83% of the total in 2010 (Fig. 1 and Table 1). Total annual waste generation is mostly a function of population size, with the top waste-producing countries having some of the largest coastal populations. However, the percentage of mismanaged waste is also important when assessing the largest contributors of waste that is available to enter the environment. Sixteen of the top 20 producers are middle-income countries, where fast economic growth is probably occurring but waste management infrastructure is lacking (the average mismanaged waste fraction is 68%). Only two of the top 20 countries have mismanaged fractions <15%; here, even a relatively low mismanaged rate results in a large mass of mismanaged plastic waste because of large coastal populations and, especially in the United States, high per capita waste generation.

Assuming no waste management infrastructure improvements, the cumulative quantity of plastic waste available to enter the marine environment from land is predicted to increase by an order of magnitude by 2025 (Fig. 2 and table S1). The predicted geographic distribution of mismanaged plastic waste in 2025 does not change substantially, although the disparity between developing and industrialized countries grows (table S2). For example, mismanaged plastic waste in the United States increases by 22%, whereas in the top five countries it more than doubles. The increase in these middle-income countries results from population growth, waste generation rates for 2025 that are consistent with economic growth (*5*), and a projected increase in plastic in the waste stream.

The analytical framework can also be used to evaluate potential mitigation strategies. For example, if the fraction of mismanaged waste were reduced by 50% (i.e., a 50% increase in adequate disposal of waste) in the 20 top-ranked countries, the mass of mismanaged plastic waste would decrease 41% by 2025. This falls to 34% if the reduction is only applied to the top 10 countries and to 26% if applied to the top 5. To achieve a 75% reduction in the mass of mismanaged plastic waste, waste management would have to be improved by 85% in the 35 top-ranked countries. This strategy would require substantial infrastructure investment primarily in low- and middle-income countries.

Alternatively, reduced waste generation and plastic use would also decrease the amount of mismanaged plastic waste. If per capita waste generation were reduced to the 2010 average (1.7 kg/day) in the 91 coastal countries that exceed it, and the percent plastic in the waste stream were capped at 11% (the 192-country average in 2010), a 26% decrease would be achieved by 2025. This strategy would target higher-income countries and might require smaller global investments. With a combined strategy, in which total waste management is achieved (0% mismanaged waste) in the 10 top-ranked countries and plastic waste generation is capped as described above, a 77% reduction could be realized, reducing the annual input of plastic waste to the ocean to 2.4 to 6.4 million MT by 2025 (table S3).

Sources of uncertainty in our estimates result from the relatively few measurements of waste generation, characterization, collection, and disposal, especially outside of urban centers. Even where data were available, methodologies were not always consistent, and some activities were not accounted for, such as illegal dumping (even in high-income countries) and ad hoc recycling or other informal waste collection (especially in low-income countries). In addition, we did not address international import and export of waste, which would affect national estimates but not global totals. Although national estimates are somewhat sensitive to the model predicting the percentage of mismanaged waste, the global estimate and ranking of top countries are not. The long-term projections are also sensitive to the model predicting growth of plastic in the waste stream; historical growth may not be a good indicator of future trends (*12*). The inclusion of the economic cost of implementation, as well as socio-cultural, environmental, and other factors that affect infrastructure development or behavioral change, would improve the evaluation of mitigation strategies (*19*).

We will not reach a global "peak waste" before 2100 (*20*). Our waste will continue to grow with increased population and increased per capita consumption associated with economic growth, especially in urban areas and developing African countries (see supplementary materials). Historically, waste management by burying or burning waste was sufficient for inert or biodegradable waste, but the rapid growth of synthetic plastics in the waste stream requires a paradigm shift. Long-term solutions will likely include waste reduction and "downstream" waste management strategies such as expanded recovery systems and extended producer responsibility (*21*, *22*). Improving waste management infrastructure in developing countries is paramount and will require substantial resources and time. While such infrastructure is being developed, industrialized countries can take immediate action by reducing waste and curbing the growth of single-use plastics.

REFERENCES AND NOTES

1. National Research Council (U.S.) Study Panel on Assessing Potential Ocean Pollutants, *Assessing Potential Ocean Pollutants: A Report of the Study Panel on Assessing Potential Ocean Pollutants to the Ocean Affairs Board, Commission on Natural Resources, National Research Council* (National Academy of Sciences, Washington, DC, 1975).
2. International Maritime Organization, "International Convention for the Prevention of Pollution from Ships (MARPOL), annex V prevention of pollution by garbage from ships"

(International Maritime Organization, London, 1988); www.imo.org/Environment/mainframe.asp?topic_id=297.
3. "Plastics – the facts 2013" (PlasticsEurope, Brussels, Belgium, 2013); www.plasticseurope.org/Document/plastics-the-facts-2013.aspx?FolID=2.
4. "Municipal solid waste generation, recycling, and disposal in the United States: Facts and figures for 2010" [U.S. Environmental Protection Agency (EPA), Washington, DC, 2011]; www.epa.gov/solidwaste/nonhaz/municipal/pubs/msw_2010_rev_factsheet.pdf
5. D. Hoornweg, P. Bhada-Tata, "What a waste: A global review of solid waste management" (The World Bank, Washington, DC, 2012); https://openknowledge.worldbank.org/handle/10986/17388
6. R. C. Thompson, C. J. Moore, F. S. vom Saal, S. H. Swan, *Philos. Trans. R. Soc. London Ser. B* **364**, 2153–2166 (2009).
7. D. K. A. Barnes, F. Galgani, R. C. Thompson, M. Barlaz, *Philos. Trans. R. Soc. London Ser. B* **364**, 1985–1998 (2009).
8. R. W. Obbard *et al.*, *Earth's Future* **2**, 315–320 (2014).
9. M. C. Goldstein, D. S. Goodwin, *PeerJ* **1**, e184 (2013).
10. H. S. Carson *et al.*, *Mar. Environ. Res.* **84**, 76–83 (2013).
11. A. Lechner *et al.*, *Environ. Pollut.* **188**, 177–181 (2014).
12. Materials and methods are available as supplementary materials on *Science* Online.
13. "Probabilistic projections of total population: Median and confidence intervals," (United Nations, Department of Economic and Social Affairs, New York, 2012); http://esa.un.org/unpd/ppp/Data-Output/UN_PPP2010_output-data.htm.
14. K. L. Law *et al.*, *Science* **329**, 1185–1188 (2010).
15. K. L. Law *et al.*, *Environ. Sci. Technol.* **48**, 4732–4738 (2014).
16. A. Cózar *et al.*, *Proc. Natl. Acad. Sci. U.S.A.* **111**, 10239–10244 (2014).
17. M. Eriksen *et al.*, *PLOS ONE* **9**, e111913 (2014).
18. "2013 resin review" (American Chemistry Council, Washington, DC, 2013).
19. R. E. Marshall, K. Farahbakhsh, *Waste Manag.* **33**, 988–1003 (2013).
20. D. Hoornweg, P. Bhada-Tata, C. Kennedy, *Nature* **502**, 615–617 (2013).
21. M. Braungart, *Nature* **494**, 174–175 (2013).
22. T. Lindhqvist, K. Lidgren, in *Ministry of the Environment, From the Cradle to the Grave - Six Studies of the Environmental Impact of Products* (Ministry of the Environment, Stockholm, Sweden, 1990), pp. 7–44.

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SUPPLEMENTARY MATERIALS

www.sciencemag.org/content/347/6223/768/suppl/DC1
Materials and Methods
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Data S1

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VIRAL REPLICATION

Structural basis for RNA replication by the hepatitis C virus polymerase

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Nucleotide analog inhibitors have shown clinical success in the treatment of hepatitis C virus (HCV) infection, despite an incomplete mechanistic understanding of NS5B, the viral RNA-dependent RNA polymerase. Here we study the details of HCV RNA replication by determining crystal structures of stalled polymerase ternary complexes with enzymes, RNA templates, RNA primers, incoming nucleotides, and catalytic metal ions during both primed initiation and elongation of RNA synthesis. Our analysis revealed that highly conserved active-site residues in NS5B position the primer for in-line attack on the incoming nucleotide. A β loop and a C-terminal membrane–anchoring linker occlude the active-site cavity in the apo state, retract in the primed initiation assembly to enforce replication of the HCV genome from the 3′ terminus, and vacate the active-site cavity during elongation. We investigated the incorporation of nucleotide analog inhibitors, including the clinically active metabolite formed by sofosbuvir, to elucidate key molecular interactions in the active site.

Hepatitis C virus (HCV) is a positive-sense, single-stranded RNA virus of the family *Flaviviridae* and genus *Hepacivirus* and is the cause of hepatitis C in humans (*1*). Long-term infection with HCV can lead to end-stage liver disease, including hepatocellular carcinoma and cirrhosis, making hepatitis C the leading cause of liver transplantation in the United States (*2*). Direct-acting antiviral drugs were approved in 2011, but they exhibited limited efficacy and had the potential for adverse side effects (*3*). The catalytic core of the viral replication complex, the NS5B RNA-dependent RNA polymerase (RdRp), supports a staggering rate of viral production, estimated to be 1.3×10^{12} virions produced per day in each infected patient (*4*). Because the NS5B polymerase active site is highly conserved, nucleotide analog inhibitors offer advantages over other classes of HCV drugs, including activity across different viral genotypes and a high barrier to the development of resistance (*5*, *6*). The nucleotide prodrug sofosbuvir was recently approved for combination treatment of chronic HCV (*7*, *8*).

One substantial obstacle for the rapid discovery of effective nucleotide-based drugs for HCV was the lack of molecular detail concerning substrate recognition during replication. NS5B contains several noncanonical polymerase elements, including a C-terminal membrane anchoring tail and a thumb domain β-loop insertion (*9*–*11*), that are implicated in RNA synthesis initiation (*12*). To gain insight into the mechanism of HCV RNA replication and its inhibition by nucleotide analog inhibitors, we determined atomic-resolution ternary structures of NS5B in both primed initiation and elongation states.

Because traditional approaches failed to yield ternary complexes (see the supplementary materials), we prepared multiple stalled enzyme-RNA-nucleotide ternary complex structures containing several designed features. First, we used NS5B from the JFH-1 genotype 2a isolate of HCV, which is extraordinarily efficient at RNA synthesis (*13*). Second, we exploited a conformational stabilization strategy that had been developed for structural analysis of G protein–coupled receptors (*14*). We hypothesized that a triple resistance NS5B mutant isolated under selective pressure of a guanosine analog inhibitor that exhibits 1.5 times the initiation activity of the wild type (*15*) might stabilize a specific conformational state along the initiation pathway. Indeed, this triple mutant exhibits a substantial structural rearrangement of the polymerase (*15*), which is consistent with the structural rearrangement observed in binary complexes of a β-loop deletion mutant bound to primer-template RNA (*16*). The triple mutant was able to incorporate native and nucleotide analog inhibitors with the RNA samples used in structure determination (fig. S1). The use of nucleotide diphosphate substrates rather than nucleotide triphosphates (fig. S2) generates stalled polymerase complexes in a catalytically relevant conformation. Ternary complexes could be obtained only with Mn^{2+}, which lowers the Michaelis constant (K_m) of the initiating nucleotide (*17*) and increases the activity of NS5B 20-fold relative to Mg^{2+} (*18*), and only with a nucleotide/Mn^{2+}/double-stranded RNA ratio of 1.0/0.6/0.2. These approaches designed to stabilize the incoming nucleotide allowed for soaking experiments targeting several distinct assemblies.

Hepatitis C virus NS5B initiates RNA synthesis by a primer-independent mechanism. Two slow steps in the catalytic pathway have been identified, including the formation of an initial dinucleotide

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Plastic waste inputs from land into the ocean

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Supplementary Materials for

Plastic waste inputs from land into the ocean

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This PDF file includes:

Other Supplementary Material for this manuscript includes the following:
(available at www.sciencemag.org/content/347/6223/768/suppl/DC1)

Data S1 (Excel file)

Methods

Estimating per capita waste generation rates and percentage of plastic in the waste stream in 2010

The World Bank generated the most recent and most comprehensive estimates of per capita waste generation rates and percentage of plastic waste for 145 countries in the year 2005 (*5*). Of the 192 coastal countries in our analysis, waste generation rates were reported for 128 countries, and percent plastic waste for 73 countries. To estimate these quantities for the remaining countries, we applied average values for each economic classification defined by the World Bank (HIC = high income; UMI = upper middle income; LMI = lower middle income; LI = low income) based upon 2010 gross national income per capita (GNI; from http://unstats.un.org/unsd/pocketbook/World_Statistics_Pocketbook_2013_edition.pdf). One exception is China, for which the 2010 value from a more recent World Bank study was used (*23*). This study reported a lower waste generation rate (1.1 kg/person/day) than would have been assigned using China's 2010 economic classification (1.2 kg/person/day). Waste generation rates likely increased from 2005 to 2010, thus our estimates are conservative.

To project the trend of plastic in the waste stream from 2005 onwards, we developed a model to predict the annual growth rate of the percent plastic in the waste stream using measured percentage of plastic in the municipal solid waste stream in the United States from 1960 (0.4%) through 2012 (12.7%), reported by the U.S. Environmental Protection Agency (*24*) (Fig S1). This proportional growth reflects increased plastic use due, in part, to the substitution of plastic for heavier materials (i.e., glass, metal). We fit three linear models (constant, first order and second order) to the curve of annual change in percent plastic versus time in the United States. The constant rate of increase (0.19% per year, standard error 0.0623%) was the best fit as determined by the lowest Akaike Information Criterion (AIC) score. This fractional increase, applied annually to each country in the study from 2005 onwards, is conservative compared to the growth in global plastic resin production (average ~5% per year from 1960 to 2011; *3*).

Estimating percentage of waste that is mismanaged

To quantify the percentage of mismanaged waste, we considered inadequate waste management practices separately from littering. We classified waste management practices for 81 coastal countries in which disposal methods were reported (*5*); we considered waste managed in landfills (high- and middle-income countries only) and in composting, recycling, and waste-to-energy programs to be "adequately managed". Dumps and landfills in low-income countries are described by the World Bank as, "Low-technology sites usually open dumping of wastes. High polluting to nearby aquifers, water bodies, settlements" (*5*). In addition, first-hand study of solid waste management in 14 developing countries by one of the authors (T. R. Siegler) supports the assertion that landfills in low-income countries are not adequately managed. Therefore, we considered landfills in low-income countries and all dumps to be "inadequately managed". The results were not substantially different if landfills in low-income counties were considered adequately managed or if those data were removed altogether.

We developed a logistic regression model to estimate the percentage of waste that is inadequately managed in each country. We modeled the ratio of adequate to inadequate waste management using data on waste disposal methods, economic classification and geographic region (as defined by the World Bank) for 81 countries for which we had complete data (*5*). We explored the effect of 2010 GNI and geographic region on the probability of inadequate management. We were also concerned about the variation in knowledge across the reporting countries on the fate of waste. In some cases the "Other" category of waste disposal methods accounted for as much as 94% of the total reported fates, although the median share of the reported fates in the Other category was 0.015%. We accounted for this by using the ratio of waste in the Other category to the total waste as a weight for the data in the regression, thus down-weighting data where there was significant uncertainty with respect to fate. Based on AIC scores, the best model used both GNI and region (Table S4). As expected, the probability of inadequate disposal of waste decreased with increasing income. Four of the regions had significantly different disposal behavior; two regions (Europe and Central Asia (ECA) and Latin America and the Caribbean (LCR)) had a lower inadequate management fraction than expected based on income alone, while two regions (East Asia and Pacific (EAP) and Middle East and North Africa (MENA)) had a higher inadequate management fraction than expected (Table S5). Using this fitted relationship we predicted the mean percentage of inadequately managed waste for the remaining countries, including a standard error.

Litter studies are difficult to synthesize because they are typically designed to evaluate counts of particular items and rarely report mass, and they vary substantially in methodology, which limits comparison between studies. We estimated percentage of waste littered using the only available national estimate of litter mass (*25*), which reported 4.17 million MT of litter generated in the United States in 2008, equivalent to approximately 2% of national waste generation (*24*). For each country we estimated 2% of the mass of total waste generated is littered. Although littering is ill-defined in the absence of formal waste management, in countries where waste management infrastructure is robust, litter can have a measurable impact (e.g., the United States and countries in the European Union).

Estimating the input of mismanaged plastic waste to the ocean

Some percentage of the total mismanaged plastic waste (inadequately managed plus litter) enters the ocean and becomes marine debris. To our knowledge no direct estimates of this conversion rate exist. The percent of mismanaged waste entering the ocean is highly variable and dependent on local factors such as weather conditions (e.g., rain storms flushing debris from waterways), topography and vegetation, and infrastructure that removes or traps mismanaged waste before it reaches the ocean, such as municipal street sweeping, beach cleaning and stormwater catchment devices.

To loosely bound the estimate of the mass of plastic waste that enters the ocean we used municipal water quality data from the San Francisco Bay (California) watershed. In the context of water quality assessment, litter and "trash" have been identified as contaminants of concern (*26*), driving initiatives to quantify capture rates by infrastructure at municipal or county levels. Total Maximum Daily Loads (TMDLs; the maximum quantity of a pollutant that can enter a waterway while still allowing the waterway to meet its water quality standards (Section 303(d) of

the Clean Water Act)) are developed for impaired waterways with water quality below applicable standards. Trash TMDLs have been developed for, or are under development for, 73 waterways in California, the Anacostia River watershed in Washington, DC and Maryland, and the Duck Creek in Mendenhall Valley, Alaska (*27-29*). The Trash TMDL, where defined and typically for trash greater than 5 mm in size, is set at zero.

Baseline and monitoring data were collected in 71 municipalities in the San Francisco Bay watershed to evaluate the effectiveness of measures designed to meet the zero trash TMDL (note that no such data exist for the Washington, DC/Maryland and Alaska regions) (*29*). The baseline trash loading rate (gallons), the quantity of trash (gallons) collected by street sweeping, storm drain catchment, and pump station cleaning, and the trash loading rate (gallons/year, defined as baseline minus the trash collected) were documented from each report. For each municipality, the percentage of trash that was not collected by street sweeping or catchment was also reported. The minimum, maximum and mean, computed over 71 municipalities, of the quantity of trash collected by street sweeping, catchments or pump stations, and the uncaptured residual, are given in Table S6. From these data an estimated 61% of trash (all materials littered in the watershed), was uncaptured by street sweeping or catchments, and thus available to enter waterways that ultimately drain to the Pacific Ocean. In our study, we assumed a more conservative range of conversion rates (15%, 25%, 40%) of mismanaged plastic waste to marine debris in order to estimate the mass of plastic that entered the ocean from land-based waste.

Projections from 2010 to 2025

To extend our estimates of the mass of mismanaged plastic waste to the year 2025, we utilized population projections for each country for 2015, 2020 and 2025 (*13*). We held 2010 per capita waste generation rates constant until 2025 when projected rates (given for 128 countries (*5*) and using averages by economic category for the remainder) were applied. We projected the percentage of plastic waste using the method described above, and used a business-as-usual approach assuming no improvements in waste management infrastructure (i.e., mismanaged waste fractions were constant). We chose this approach because of the inability to predict future infrastructure development, and because it provided a framework to examine the effect of potential mitigation strategies such as a reduction in mismanaged waste through infrastructure development.

To determine the size of coastal populations, gridded population density raster data was downloaded for use in ArcMap 10.1® for 2010 and 2015 (*30*). A 50 km buffer was drawn around the world's coastlines, and the gridded population raster data was clipped to this buffer. This allowed us to calculate a coastal (within 50 km of the coastline) population for each country. To project the coastal populations forward from 2015 we assumed that the coastal populations would increase at a rate equal to the total projected population increase for each country.

Because the fraction of inadequately managed waste and percent plastic in the waste stream were derived from predictive models, as described above, we used the standard error associated with these fits to generate error bars on the 2025 projections of the mass of mismanaged waste available to become marine debris. For each pentad with population growth data we randomly

generated 1000 values of both the mismanagement fraction and the plastic percentage from normal distributions with the mean and standard deviation defined using the mean and standard error associated with the respective predictive model. The error bars in Figure 1 describe the minimum and maximum value (from the 1000 scenarios) of the mass of mismanaged waste for a particular year.

Supplementary text

Comparison of global plastic input from mismanaged waste to ocean estimates of floating plastic debris

Cozar et al. (16) estimated the mass of floating plastic debris (7,000 – 35,000 tons) from data collected using surface-towed plankton nets. Plastic debris collected in these nets is typically microplastics, 0.33 mm – 5 mm in size. Eriksen et al. (17) reported 35,540 tons of floating microplastics from plankton net data, and 233,400 tons of "larger plastic items" (> 20 cm in size) from shipboard visual survey data. Both estimates of the mass of net-collected plastic debris, and the combined estimate from net plus visual survey data, are orders of magnitude smaller than our estimate of 4.8 to 12.7 million tonnes (5.3 to 14.0 million tons) of plastic entering the ocean in 2010 from land-based waste. Our estimate includes all plastic materials (including those that sink) in all size classes, whereas the published ocean estimates only compute the mass of floating plastic in a particular size class (or classes). In addition, we estimate the input of plastic waste in a single year (2010), while the ocean estimates represent an accumulation of floating plastic debris over an unknown time period (in part because the fragmentation and degradation rates of plastic in the ocean, and therefore the "age" of debris collected, are unknown).

Future projections

Our results indicate China had the largest mass of mismanaged plastic waste in 2010, similar to previously reported trends (*20,23*). By 2025, South Asia (e.g., India) is predicted to have a large increase in the mass of mismanaged plastic waste. In addition, two African countries (Nigeria and Senegal) showed large population growth and, therefore, increased mismanaged waste. Following projected trends through 2100 of large population growth, urbanization and increased waste generation, the forethought to develop infrastructure to adequately manage waste in African countries could mitigate increasing future inputs of plastic into the marine environment.

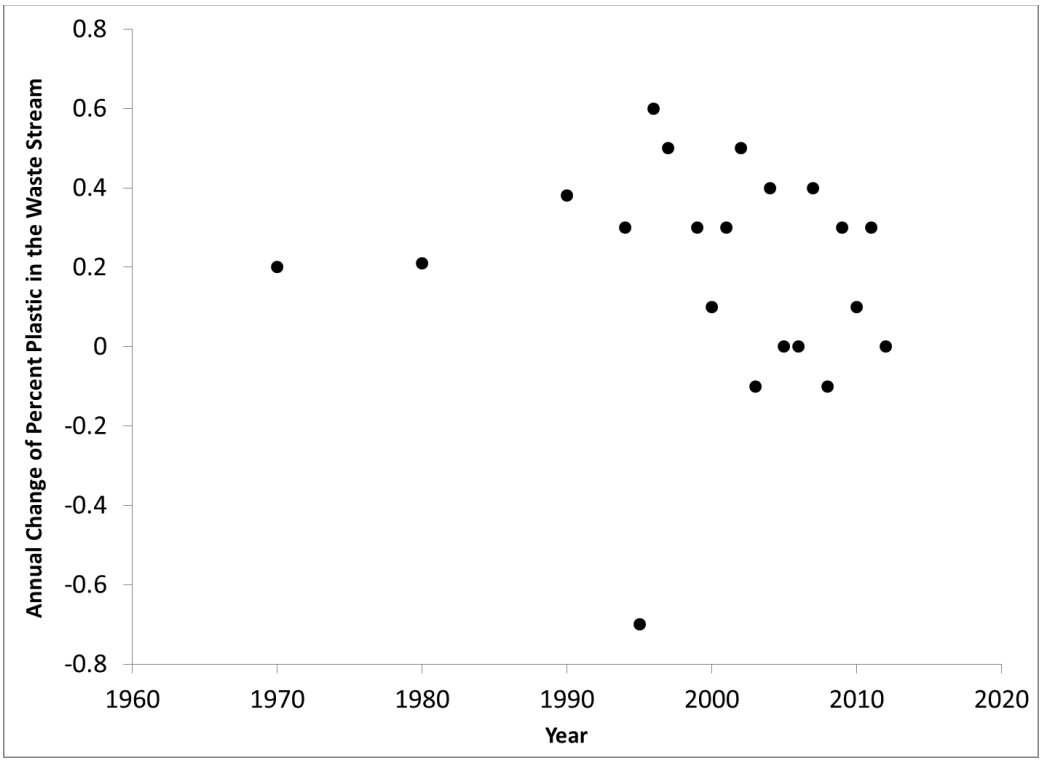

Figure S1: Annual change of percent plastic in municipal solid waste in the United States as a function of year, 1960 – 2012 (*24*), illustrating a mean annual increase of 0.19%.

Table S1: Annual and cumulative quantities (millions of metric tons (MMT)) of mismanaged plastic waste and plastic marine debris (assuming three different conversion rates) for 2010-2025.

Year	Mismanaged plastic waste [MMT/year]	15% marine debris (MMT)	25% marine debris (MMT)	40% marine debris (MMT)
2010	31.9	4.8	8.0	12.7
2015	36.5	5.5	9.1	14.6
2020	41.3	6.2	10.3	16.5
2025	69.9	10.5	17.5	28.0
Cumulative	618.7	92.8	154.7	247.5

Table S2. Top 20 countries ranked by mass of mismanaged plastic waste in 2010 and 2025, with percent increase in coastal population from 2010 to 2025. MMT, million metric tons

	Year 2010		Year 2025		
Rank	Country	Mismanaged plastic waste [MMT/year]	Country	Mismanaged plastic waste [MMT/year]	% pop. change since 2010
1	China	8.82	China	17.81	3.7%
2	Indonesia	3.22	Indonesia	7.42	11.9%
3	Philippines	1.88	Philippines	5.09	26.0%
4	Vietnam	1.83	Vietnam	4.17	13.3%
5	Sri Lanka	1.59	India	2.88	18.7%
6	Thailand	1.03	Nigeria	2.48	45.1%
7	Egypt	0.97	Bangladesh	2.21	18.5%
8	Malaysia	0.94	Thailand	2.18	5.4%
9	Nigeria	0.85	Egypt	1.94	25.0%
10	Bangladesh	0.79	Sri Lanka	1.92	9.0%
11	South Africa	0.63	Malaysia	1.77	23.6%
12	India	0.60	Pakistan	1.22	26.6%
13	Algeria	0.52	Burma	1.15	11.1%
14	Turkey	0.49	Algeria	1.02	18.4%
15	Pakistan	0.48	Brazil	0.95	10.6%
16	Brazil	0.47	South Africa	0.84	7.2%
17	Burma	0.46	Turkey	0.79	16.2%
18	Morocco	0.31	Senegal	0.74	44.3%
19	Korea, North	0.30	Morocco	0.71	14.1%
20	United States	0.28	North Korea	0.61	5.0%

Table S3: The effect of a variety of mitigation strategies on the amount of mismanaged plastic waste generated and the amount of plastic waste entering the ocean as marine debris in 2025 assuming three different conversion rates. MMT, million metric tons

	Mitigation strategy	Reduction	Mismanaged plastic waste [MMT/year]	15% marine debris (MMT)	25% marine debris (MMT)	40% marine debris (MMT)
	No intervention	0%	69.1	10.4	17.3	27.7
1	Reduce mismanaged waste by 50% in Top 20	41%	41.0	6.2	10.3	16.4
2	Reduce mismanaged waste by 50% in Top 10	34%	45.7	6.9	11.4	18.3
3	Reduce mismanaged waste by 50% in Top 5	26%	50.9	7.6	12.7	20.4
4	Reduce mismanaged waste by 85% in Top 35	75%	17.4	2.6	4.4	7.0
5	Cap at 1.7 kg/person/day and 11% plastic	26%	51.5	7.7	12.9	20.6
6	Top 10 = 0% combined with Strategy 5	77%	15.9	2.4	4.0	6.4

Table S4: Comparison of model quality, using the Akaike Information Criterion (AIC) scores, to predict the probability of inadequate waste management.

Rank	Models	AICs
4	Intercept only	5647.7
3	Intercept + GNI2010	3067.1
5	Intercept + Region	10,403.2
2	Intercept + GNI2010 + Region	2800.1
1	Intercept + GNI2010 + Region[1]	2344.5

[1]Full model with observations weighted for uncertainty

Table S5: Terms and significance for the best fit model for the probability of inadequate waste management. Coefficients correspond to the response variable on the logit scale.

Term	Estimate	Std. Error	z value	Pr(>|z|)
Intercept	1.7400	0.1233	14.1110	< 2e-16
GNI2010	-0.0002	0.0000	-18.1870	< 2e-16
Region EAP	0.3267	0.2885	1.1320	0.2575
Region ECA	-1.1300	0.1515	-7.4570	0.0000
Region LCR	-1.7130	0.1360	-12.6000	< 2e-16
Region MENA	-0.4626	0.1435	-3.2230	0.0013
Region OECD	-16.8900	337.5000	-0.0500	0.9601

Regions defined by World Bank: EAP = East Asia and Pacific; ECA = Europe and Central Asia; LCR = Latin America and the Caribbean; MENA = Middle East and North Africa; OECD = The Organization for Economic Co-operation and Development.

Table S6: Percentage of trash collected by infrastructure in the San Francisco Bay watershed (*29*), and the residual uncollected percentage that is available to enter the ocean as marine debris.

n = 71 municipalities	**% total trash collected by street sweeping**	**% total trash collected in stormwater catchments**	**% total trash collected in pump stations**	**% total trash uncollected**
Minimum	0%	1.2%	0%	36%
Maximum	61%	5.0%	16.5%	95%
Mean	34%	3.2%	1.5%	61%

References and Notes

1. National Research Council (U.S.) Study Panel on Assessing Potential Ocean Pollutants, *Assessing Potential Ocean Pollutants: A Report of the Study Panel on Assessing Potential Ocean Pollutants to the Ocean Affairs Board, Commission on Natural Resources, National Research Council* (National Academy of Sciences, Washington, DC, 1975).

2. International Maritime Organization, "International Convention for the Prevention of Pollution from Ships (MARPOL), annex V prevention of pollution by garbage from ships" (International Maritime Organization, London, 1988); www.imo.org/Environment/mainframe.asp?topic_id=297.

3. "Plastics – the facts 2013" (PlasticsEurope, Brussels, Belgium, 2013); www.plasticseurope.org/Document/plastics-the-facts-2013.aspx?FolID=2.

4. "Municipal solid waste generation, recycling, and disposal in the United States: Facts and figures for 2010" [U.S. Environmental Protection Agency (EPA), Washington, DC, 2011]; www.epa.gov/solidwaste/nonhaz/municipal/pubs/msw_2010_rev_factsheet.pdf

5. D. Hoornweg, P. Bhada-Tata, "What a waste: A global review of solid waste management" (The World Bank, Washington, DC, 2012); https://openknowledge.worldbank.org/handle/10986/17388

6. R. C. Thompson, C. J. Moore, F. S. vom Saal, S. H. Swan, Plastics, the environment and human health: Current consensus and future trends. *Philos. Trans. R. Soc. London Ser. B* **364**, 2153–2166 (2009). Medline

7. D. K. A. Barnes, F. Galgani, R. C. Thompson, M. Barlaz, Accumulation and fragmentation of plastic debris in global environments. *Philos. Trans. R. Soc. London Ser. B* **364**, 1985–1998 (2009). Medline doi:10.1098/rstb.2008.0205

8. R. W. Obbard, S. Sadri, Y. Q. Wong, A. A. Khitun, I. Baker, R. C. Thompson, Global warming releases microplastic legacy frozen in Arctic Sea ice. *Earth's Future* **2**, 315–320 (2014). doi:10.1002/2014EF000240

9. M. C. Goldstein, D. S. Goodwin, Gooseneck barnacles (*Lepas* spp.) ingest microplastic debris in the North Pacific Subtropical Gyre. *PeerJ* **1**, e184 (2013). Medline

10. H. S. Carson, M. R. Lamson, D. Nakashima, D. Toloumu, J. Hafner, N. Maximenko, K. J. McDermid, Tracking the sources and sinks of local marine debris in Hawai'i. *Mar. Environ. Res.* **84**, 76–83 (2013). Medline doi:10.1016/j.marenvres.2012.12.002

11. A. Lechner H. Keckeis, F. Lumesberger-Loisl, B. Zens, R. Krusch, M. Tritthart, M. Glas, E. Schludermann, The Danube so colourful: A potpourri of plastic litter outnumbers fish larvae in Europe's second largest river. *Environ. Pollut.* **188**, 177–181 (2014). Medline doi:10.1016/j.envpol.2014.02.006

12. Materials and methods are available as supplementary materials on *Science* Online.

13. "Probabilistic projections of total population: Median and confidence intervals," (United Nations, Department of Economic and Social Affairs, New York, 2012); http://esa.un.org/unpd/ppp/Data-Output/UN_PPP2010_output-data.htm.

14. K. L. Law, S. Morét-Ferguson, N. A. Maximenko, G. Proskurowski, E. E. Peacock, J. Hafner, C. M. Reddy, Plastic accumulation in the North Atlantic subtropical gyre. *Science* **329**, 1185–1188 (2010). Medline doi:10.1126/science.1192321

15. K. L. Law, S. E. Morét-Ferguson, D. S. Goodwin, E. R. Zettler, E. Deforce, T. Kukulka, G. Proskurowski, Distribution of surface plastic debris in the eastern Pacific Ocean from an 11-year data set. *Environ. Sci. Technol.* **48**, 4732–4738 (2014). Medline doi:10.1021/es4053076

16. A. Cózar, F. Echevarría, J. I. González-Gordillo, X. Irigoien, B. Ubeda, S. Hernández-León, A. T. Palma, S. Navarro, J. García-de-Lomas, A. Ruiz, M. L. Fernández-de-Puelles, C. M. Duarte, Plastic debris in the open ocean. *Proc. Natl. Acad. Sci. U.S.A.* **111**, 10239–10244 (2014). Medline doi:10.1073/pnas.1314705111

17. M. Eriksen, L. C. Lebreton, H. S. Carson, M. Thiel, C. J. Moore, J. C. Borerro, F. Galgani, P. G. Ryan, J. Reisser, Plastic pollution in the world's oceans: More than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. *PLOS ONE* **9**, e111913 (2014). Medline doi:10.1371/journal.pone.0111913

18. "2013 resin review" (American Chemistry Council, Washington, DC, 2013).

19. R. E. Marshall, K. Farahbakhsh, Systems approaches to integrated solid waste management in developing countries. *Waste Manag.* **33**, 988–1003 (2013). Medline doi:10.1016/j.wasman.2012.12.023

20. D. Hoornweg, P. Bhada-Tata, C. Kennedy, Environment: Waste production must peak this century. *Nature* **502**, 615–617 (2013). Medline doi:10.1038/502615a

21. M. Braungart, Upcycle to eliminate waste. *Nature* **494**, 174–175 (2013). doi:10.1038/494174b

22. T. Lindhqvist, K. Lidgren, in *Ministry of the Environment, From the Cradle to the Grave - Six Studies of the Environmental Impact of Products* (Ministry of the Environment, Stockholm, Sweden, 1990), pp. 7–44.

23. D. Hoornweg, P. Lam, M. Chaudhry, "Waste management in China: Issues and recommendations" (The World Bank, Washington, DC, 2005).

24. "Municipal solid waste generation, recycling, and disposal in the United States, tables and figures for 2012" (EPA, Washington, DC, 2014); www.epa.gov/waste/nonhaz/municipal/pubs/2012_msw_dat_tbls.pdf.

25. Mid-Atlantic Solid Waste Consultants, "2009 national visible litter survey and litter cost study" (Keep America Beautiful, Washington, DC, 2009).

26. N. Armitage, A. Rooseboom, The removal of urban litter from stormwater conduits and streams: Paper 1 - The quantities involved and catchment litter management options. *Water S.A.* **26**, 181–187 (2000).

27. "Total maximum daily load (TMDL) for debris in the waters of Duck Creek in Mendenhall Valley, Alaska" (EPA, Seattle, 2000); http://dec.alaska.gov/water/tmdl/pdfs/duckcreekdebris.pdf.

28. “Decision rationale: Total maximum daily loads of trash for the Anacostia River watershed, Montgomery and Prince George’s Counties, Maryland and the District of Columbia,” (EPA, Washington, DC, 2010); www.epa.gov/reg3wapd/pdf/AnacostiaTMDLPortfolio.pdf.

29. “Preliminary baseline trash generation rates for San Francisco Bay Area MS4s, technical memorandum” (Bay Area Stormwater Management Agencies Association, Oakland, CA, 2012).

30. Gridded Population of the World (GPW), v3 (2014); http://sedac.ciesin.columbia.edu/data/set/gpw-v3-population-count-future-estimates/data-download.

Country	Economic status[1]	Coastal population[2]	Waste generation rate [kg/person/day][3]	% Plastic in waste stream[4]	% Inadequately managed waste[5]	% Littered waste[6]	Waste generation [kg/day][7]	Plastic waste generation [kg/day][7]	Inadequately managed plastic waste [kg/day][7]	Plastic waste littered [kg/day][7]	Mismanaged plastic waste [kg/person/day][7]	Mismanaged plastic waste in 2010 [tonnes][7]	Mismanaged plastic waste in 2025 [tonnes][7]
Albania	LMI	2,530,533	**0.77**	**9**	45	2	1,948,510	174,392	77,897	3,488	0.032	29,705	63,051
Algeria[8]	UMI	16,556,580	1.2	12	58	2	19,867,896	2,374,214	1,378,693	47,484	0.086	520,555	1,017,444
Angola	LMI	3,790,041	**0.48**	13	71	2	1,819,220	235,589	166,597	4,712	0.045	62,528	136,770
Anguilla	HIC	14,561	2.1	12	2	2	30,578	3,654	68	73	0.010	52	73
Antigua & Barbuda	HIC	66,843	**5.5**	12	6	2	367,637	43,933	2,555	879	0.051	1,253	1,385
Argentina	UMI	16,449,245	**1.22**	**15**	12	2	20,068,079	3,000,178	372,261	60,004	0.026	157,777	320,197
Aruba	HIC	137,910	2.1	12	1	2	289,611	34,609	326	692	0.007	372	476
Australia	HIC	17,235,954	**2.23**	**5**	0	2	38,436,177	1,902,591	0	38,052	0.002	13,889	24,653
Bahamas	HIC	341,145	**3.25**	12	1	2	1,108,721	132,492	1,002	2,650	0.011	1,333	1,718
Bahrain	HIC	743,574	**1.1**	12	10	2	817,931	97,743	10,033	1,955	0.016	4,376	9,915
Bangladesh	LI	70,874,124	**0.43**	**8**	87	2	30,475,873	2,422,832	2,108,603	48,457	0.030	787,327	2,210,230
Barbados	HIC	276,784	**4.75**	12	4	2	1,314,724	157,110	6,400	3,142	0.034	3,483	3,719
Belgium	HIC	4,747,957	**1.33**	**6**	0	2	6,314,783	375,730	0	7,515	0.002	2,743	5,739
Belize	UMI	202,429	**2.87**	**6**	29	2	580,971	34,568	9,972	691	0.053	3,892	6,042
Benin	LI	3,235,418	**0.54**	**8**	83	2	1,747,126	138,896	115,073	2,778	0.036	43,015	119,467
Bermuda	HIC	66,966	2.1	12	0	2	140,629	16,805	0	336	0.005	123	173
Bosnia and Herzegovina	UMI	585,582	1.2	12	40	2	702,698	83,972	33,813	1,679	0.061	12,955	20,201
Brazil	UMI	74,696,771	**1.03**	**16**	9	2	76,937,674	12,271,559	1,046,087	245,431	0.017	471,404	954,198
British Virgin Islands	HIC	29,674	2.1	12	0	2	62,315	7,447	3	149	0.005	55	78
Brunei	HIC	359,871	**0.87**	**3**	1	2	313,088	9,236	71	185	0.001	93	341
Bulgaria	UMI	1,002,695	**1.28**	12	31	2	1,283,450	153,372	48,273	3,067	0.051	18,739	25,770
Burma/Myanmar	LI	18,988,522	**0.44**	**17**	87	2	8,354,950	1,416,164	1,227,208	28,323	0.066	458,269	1,149,267
Cambodia	LI	1,391,254	0.6	**11**	87	2	834,752	91,405	79,502	1,828	0.058	29,686	62,834
Cameroon	LMI	1,986,723	**0.77**	**6**	81	2	1,529,777	91,022	74,107	1,820	0.038	27,713	71,863
Canada	HIC	11,846,863	**2.33**	**4**	0	2	27,603,191	1,090,326	0	21,807	0.002	7,959	14,668
Cape Verde	LMI	522,245	**0.5**	13	74	2	261,123	33,815	24,933	676	0.049	9,347	18,475
Cayman Islands	HIC	51,864	2.1	12	0	2	108,914	13,015	0	260	0.005	95	134
Channel Islands	HIC	153,352	2.1	12	0	2	322,039	38,484	0	770	0.005	281	355
Chile	UMI	5,621,550	**1.08**	**11**	7	2	6,071,274	664,805	46,304	13,296	0.011	21,754	42,562
China	UMI	262,892,387	**1.1**	**11**	74	2	289,181,626	31,665,388	23,530,300	633,308	0.092	8,819,717	17,814,777
Christmas Island	HIC	1,402	2.1	12	0	2	2,944	352	0	7	0.005	3	4
Cocos Islands	HIC	596	2.1	12	0	2	1,252	150	0	3	0.005	1	2
Colombia[8]	UMI	7,498,563	1.2	12	21	2	8,998,276	1,075,294	231,024	21,506	0.034	92,173	179,973
Comoros	LI	938,595	**2.23**	9	83	2	2,093,067	187,329	155,246	3,747	0.169	58,032	100,870
Congo Rep of	LMI	847,807	**0.53**	13	77	2	449,338	58,189	44,769	1,164	0.054	16,766	39,853
Congo, Dem rep. of	LI	1,076,056	**0.5**	9	85	2	538,028	48,154	40,703	963	0.039	15,208	43,480
Cook Islands	UMI	20,934	1.2	12	36	2	25,121	3,002	1,079	60	0.054	416	784
Costa Rica	UMI	2,479,298	**1.36**	**19**	16	2	3,371,845	638,965	102,536	12,779	0.047	42,090	75,895
Cote d'Ivoire[8]	LMI	6,230,583	0.79	13	82	2	4,922,161	637,420	520,288	12,748	0.086	194,558	537,163
Croatia[8]	HIC	1,602,782	2.1	12	9	2	3,365,842	402,218	37,053	8,044	0.028	16,461	19,593
Cuba	UMI	11,333,471	**0.81**	**11**	23	2	9,180,112	1,005,222	228,196	20,104	0.022	90,630	180,454
Curacao	HIC	143,784	2.1	12	0	2	301,946	36,083	12	722	0.005	268	378
Cyprus	HIC	840,556	**2.07**	**12**	0	2	1,739,951	207,924	831	4,158	0.006	1,821	2,611
Denmark	HIC	5,376,386	**2.34**	**2**	0	2	12,580,743	245,324	0	4,906	0.001	1,791	4,233
Dhekelia	HIC	15,700	2.1	12	0	2	32,970	3,940	2	79	0.005	30	42
Djibouti	LMI	621,744	0.79	13	73	2	491,178	63,608	46,191	1,272	0.076	17,324	45,612
Dominica	UMI	70,138	**1.24**	12	19	2	86,971	10,393	1,938	208	0.031	783	1,428
Dominican Republic[8]	UMI	8,232,586	1.2	12	25	2	9,879,103	1,180,553	300,295	23,611	0.039	118,226	228,671
East Timor	LMI	668,749	0.79	13	81	2	528,312	68,416	55,316	1,368	0.085	20,690	64,205
Ecuador	LMI	6,400,048	**1.13**	13	30	2	7,232,054	936,551	280,948	18,731	0.047	109,383	211,021
Egypt	LMI	21,750,943	**1.37**	**13**	67	2	29,798,792	3,858,944	2,572,170	77,179	0.122	967,012	1,937,428
El Salvador	LMI	6,410,726	**1.13**	13	33	2	7,244,120	938,114	306,531	18,762	0.051	118,732	226,860
Equatorial Guinea	UMI	351,600	1.2	12	30	2	421,920	50,419	15,367	1,008	0.047	5,977	14,124
Eritrea	LI	1,266,222	**0.5**	9	77	2	633,111	56,663	43,411	1,133	0.035	16,259	43,484
Estonia	HIC	878,021	**1.47**	12	9	2	1,290,691	154,238	13,296	3,085	0.019	5,979	8,367
Falkland Islands	HIC	2,602	2.1	12	0	2	5,464	653	0	13	0.005	5	7
Faroe Islands	HIC	50,554	2.1	12	0	2	106,163	12,687	0	254	0.005	93	131
Fiji	UMI	896,145	**2.1**	**9**	78	2	1,881,905	168,430	131,582	3,369	0.151	49,257	70,995
Finland	HIC	2,927,674	**2.13**	**11**	0	2	6,235,946	682,836	0	13,657	0.005	4,985	6,422
France	HIC	17,287,280	**1.92**	**10**	0	2	33,191,578	3,302,562	0	66,051	0.004	24,109	34,671
French Guiana	UMI	167,631	1.2	12	25	2	201,157	24,038	5,986	481	0.039	2,360	5,577
French Polynesia	HIC	270,618	2.1	12	3	2	568,298	67,912	2,165	1,358	0.013	1,286	1,824
Gabon	UMI	862,328	**0.45**	12	34	2	388,048	46,372	15,750	927	0.019	6,087	15,446
Georgia	LMI	1,124,249	**1.69**	**4**	51	2	1,899,981	75,049	38,149	1,501	0.035	14,472	24,532
Germany	HIC	8,837,035	**2.11**	**23**	0	2	18,646,144	4,279,290	0	85,586	0.010	31,239	33,317
Ghana[8]	LMI	7,727,702	0.79	**5**	81	2	6,104,885	302,192	244,835	6,044	0.032	91,571	325,116
Gibraltar	HIC	33,483	2.1	12	0	2	70,314	8,403	9	168	0.005	65	92
Greece	HIC	9,794,702	**2**	**10**	0	2	19,589,404	1,949,146	0	38,983	0.004	14,229	18,621
Greenland	HIC	57,068	2.1	12	0	2	119,843	14,321	0	286	0.005	105	148
Grenada	UMI	96,121	**2.71**	12	18	2	260,488	31,128	5,536	623	0.064	2,248	2,452
Guadeloupe	UMI	466,166	1.2	12	25	2	559,399	66,848	16,646	1,337	0.039	6,564	11,502
Guam	HIC	190,809	2.1	12	0	2	400,699	47,884	118	958	0.006	393	574
Guatemala	LMI	2,392,442	**2**	**14**	36	2	4,784,884	667,491	237,508	13,350	0.105	91,563	157,698
Guernsey	HIC	56,289	2.1	12	0	2	118,207	14,126	0	283	0.005	103	146
Guinea	LI	1,996,496	0.6	**5**	84	2	1,197,898	59,296	49,538	1,186	0.025	18,514	59,889
Guinea-Bissau	LI	1,208,106	0.6	9	83	2	724,864	64,875	54,156	1,298	0.046	20,240	51,947
Guyana	LMI	513,235	**5.33**	**11**	36	2	2,735,543	299,542	109,084	5,991	0.224	42,002	36,103
Haiti	LI	9,155,693	**1**	9	47	2	9,155,693	819,435	387,416	16,389	0.044	147,389	324,178
Honduras	LMI	3,324,144	**1.45**	13	40	2	4,820,009	624,191	249,539	12,484	0.079	95,638	189,169
Hong Kong	HIC	7,573,074	**1.99**	**20**	1	2	15,070,417	3,006,548	17,983	60,131	0.010	28,512	37,606
Iceland	HIC	292,708	**1.56**	**18**	0	2	456,624	81,964	0	1,639	0.006	598	883
India	LMI	187,493,433	**0.34**	**3**	85	2	63,747,767	1,880,559	1,605,729	37,611	0.009	599,819	2,881,294
Indonesia	LMI	187,223,476	**0.52**	**11**	81	2	97,356,208	10,660,505	8,600,093	213,210	0.047	3,216,856	7,415,202
Iran[8]	UMI	9,099,695	1.2	12	50	2	10,919,634	1,304,896	651,717	26,098	0.074	247,403	460,067
Iraq	LMI	639,228	0.79	13	63	2	504,990	65,396	41,330	1,308	0.067	15,563	47,694
Ireland	HIC	3,749,576	**3.58**	**12**	0	2	13,423,482	1,604,106	0	32,082	0.009	11,710	14,045
Isle of Man	HIC	81,222	2.1	12	0	2	170,566	20,383	0	408	0.005	149	210
Israel	HIC	6,677,810	**2.12**	**14**	1	2	14,156,957	1,974,896	12,577	39,498	0.008	19,007	28,211
Italy	HIC	33,822,532	**2.23**	**6**	0	2	75,424,246	4,487,743	0	89,755	0.003	32,761	45,058
Jamaica	UMI	2,820,558	**0.18**	**19**	27	2	507,700	96,209	25,525	1,924	0.010	10,019	59,575
Japan	HIC	115,228,891	**1.71**	**10**	0	2	197,041,404	19,605,620	0	392,112	0.003	143,121	177,241
Jersey	HIC	90,076	2.1	12	0	2	189,160	22,605	0	452	0.005	165	233
Jordan[8]	UMI	55,392	1.2	12	58	2	66,470	7,943	4,600	159	0.086	1,737	2,975
Kenya	LI	2,729,945	**0.3**	9	83	2	818,984	73,299	60,610	1,466	0.023	22,658	87,109
Kiribati	LMI	94,487	0.79	13	84	2	74,645	9,666	8,148	193	0.088	3,045	6,977
Korea, North	LI	17,327,483	0.6	9	88	2	10,396,490	930,486	815,165	18,610	0.048	304,328	610,607
Korea, South (Republic of	HIC	41,654,619	**1.24**	**9**	0	2	51,651,728	4,622,830	0	92,457	0.001	33,747	52,764
Kuwait	HIC	2,293,604	**5.72**	12	0	2	13,119,415	1,567,770	122	31,355	0.014	11,489	13,479
Latvia	UMI	1,432,078	**1.03**	12	14	2	1,475,040	176,267	24,651	3,525	0.020	10,284	16,654
Lebanon	UMI	3,890,871	**1.18**	**8**	34	2	4,591,228	365,003	123,700	7,300	0.034	47,815	101,852
Liberia	LI	2,148,271	0.6	**14**	84	2	1,288,963	179,810	151,822	3,596	0.072	56,728	142,787
Libya	UMI	4,050,128	1.2	12	23	2	4,860,154	580,788	132,985	11,616	0.036	52,779	102,577
Lithuania	UMI	443,894	**1.1**	12	14	2	488,283	58,350	8,364	1,167	0.021	3,479	5,478
Macao	HIC	257,911	**1.47**	**25**	0	2	379,129	94,593	22	1,892	0.007	698	1,146
Madagascar	LI	7,062,413	**0.8**	**2**	84	2	5,649,930	110,174	92,377	2,203	0.013	34,522	177,625
Malaysia	UMI	22,890,252	**1.52**	**13**	55	2	34,793,183	4,505,717	2,476,510	90,114	0.112	936,818	1,765,977
Maldives	LMI	392,567	**2.48**	13	66	2	973,566	126,077	83,695	2,522	0.220	31,469	40,143
Malta	HIC	404,707	**1.78**	12	6	2	720,378	86,085	5,456	1,722	0.018	2,620	3,757
Marshall Islands	UMI	58,086	1.2	**16**	77	2	69,703	11,118	8,510	222	0.150	3,187	5,717
Martinique	HIC	402,257	2.1	12	0	2	844,740	100,946	33	2,019	0.005	749	955
Mauritania	LI	1,005,481	**0.5**	9	82	2	502,741	44,995	37,056	900	0.038	13,854	39,754
Mauritius	UMI	1,255,952	**2.3**	**10**	51	2	2,888,690	287,425	147,817	5,748	0.122	56,051	73,172
Mexico	UMI	22,647,771	**1.24**	**7**	12	2	28,083,236	1,951,785	238,616	39,036	0.012	101,343	233,393
Micronesia	LMI	154,895	0.79	13	81	2	122,367	15,847	12,818	317	0.085	4,794	10,699
Monaco	HIC	34,050	2.1	12	0	2	71,505	8,545	0	171	0.005	62	88
Montenegro	UMI	260,336	1.2	12	30	2	312,403	37,332	11,353	747	0.046	4,416	7,244
Montserrat	UMI	5,173	1.2	12	12	2	6,208	742	89	15	0.020	38	71
Morocco	LMI	17,303,431	**1.46**	**5**	66	2	25,263,009	1,250,519	824,650	25,010	0.049	310,126	706,583
Mozambique	LI	9,566,559	**0.14**	**11**	84	2	1,339,318	146,655	123,081	2,933	0.013	45,995	287,067
Namibia[8]	UMI	155,084	1.2	12	66	2	186,101	22,239	14,570	445	0.097	5,480	11,340

Nauru	UMI	15,289	1.2	12	67	2	18,347	2,192	1,473	44	0.099	554	1,043
Netherlands	HIC	8,971,770	**2.12**	**20**	0	2	19,020,152	3,794,520	0	75,890	0.008	27,700	32,387
Netherlands Antilles	HIC	227,165	2.1	12	0	2	477,047	57,007	0	1,140	0.005	416	368
New Caledonia	HIC	257,904	2.1	12	0	2	541,598	64,721	0	1,294	0.005	472	702
New Zealand	HIC	3,862,054	**3.68**	**9**	0	2	14,212,359	1,272,006	0	25,440	0.007	9,286	11,517
Nicaragua	LMI	3,482,653	**1.1**	13	45	2	3,830,918	496,104	221,353	9,922	0.066	84,415	169,439
Nigeria[8]	LMI	27,477,112	0.79	13	81	2	21,706,918	2,811,046	2,276,636	56,221	0.085	851,493	2,481,008
Niue	HIC	1,799	2.1	12	0	2	3,778	451	1	9	0.006	4	5
Norfolk Island	LMI	2,156	0.79	13	82	2	1,703	221	182	4	0.086	68	156
Northern Mariana Islands	HIC	106,256	2.1	12	0	2	223,138	26,665	66	533	0.006	219	309
Norway	HIC	4,131,679	**2.8**	**10**	0	2	11,568,701	1,151,086	0	23,022	0.006	8,403	9,798
Oman	HIC	2,597,556	**0.7**	12	4	2	1,818,289	217,286	8,416	4,346	0.005	4,658	11,774
Pakistan[8]	LMI	14,581,952	0.79	13	86	2	11,519,742	1,491,807	1,286,583	29,836	0.090	480,493	1,221,460
Palau	UMI	23,446	1.2	12	56	2	28,135	3,362	1,895	67	0.084	716	1,350
Palestine (Gaza Strip is or	LMI	3,045,258	0.79	**8**	6	2	2,405,754	191,257	11,515	3,825	0.005	5,599	18,676
Panama	UMI	3,249,531	**1.21**	**12**	18	2	3,931,933	469,866	84,815	9,397	0.029	34,388	70,759
Papua New Guinea	LMI	2,747,514	0.79	13	86	2	2,170,536	281,084	240,502	5,622	0.090	89,835	242,328
Peru[8]	UMI	13,765,608	1.2	12	25	2	16,518,730	1,973,988	493,267	39,480	0.039	194,453	377,111
Philippines	LMI	83,446,862	**0.5**	**15**	81	2	41,723,431	6,237,653	5,035,956	124,753	0.062	1,883,659	5,088,394
Poland	UMI	3,272,933	**0.88**	**11**	12	2	2,880,181	315,380	36,715	6,308	0.013	15,703	26,855
Portugal	HIC	8,507,951	**2.21**	**12**	0	2	18,802,572	2,246,907	0	44,938	0.005	16,402	19,382
Puerto Rico	HIC	4,249,848	2.1	12	2	2	8,924,681	1,066,499	23,923	21,330	0.011	16,517	20,481
Qatar	HIC	653,007	**1.33**	12	0	2	868,499	103,786	0	2,076	0.003	758	1,560
Reunion	UMI	809,426	1.2	12	0	2	971,311	116,072	0	2,321	0.003	847	1,615
Romania	UMI	875,170	**1.04**	**4**	26	2	910,177	35,952	9,172	719	0.011	3,610	8,261
Russia	UMI	10,812,537	**0.93**	12	16	2	10,055,659	1,201,651	197,226	24,033	0.020	80,759	128,946
Saint Helena	UMI	6,839	1.2	12	25	2	8,207	981	244	20	0.039	96	182
Saint Kitts and Nevis	UMI	36,102	**5.45**	12	6	2	196,756	23,512	1,489	470	0.054	715	742
Saint Lucia	UMI	163,227	**4.35**	12	20	2	710,037	84,849	16,683	1,697	0.113	6,709	8,610
Saint Maarten, DWI	HIC	37,429	2.1	12	0	2	78,601	9,393	3	188	0.005	70	98
Saint Pierre	HIC	5,888	2.1	12	0	2	12,365	1,478	0	30	0.005	11	15
Saint Vincent and the Gre	UMI	120,149	**1.7**	**13**	21	2	204,253	26,451	5,621	529	0.051	2,245	2,996
Samoa	LMI	168,025	0.79	13	80	2	132,740	17,190	13,688	344	0.084	5,122	10,989
Sao Tome and Principe[8]	LMI	163,740	0.79	13	81	2	129,355	16,751	13,587	335	0.085	5,081	13,309
Saudi Arabia	HIC	3,593,471	**1.3**	12	8	2	4,671,512	558,246	45,142	11,165	0.016	20,552	43,855
Senegal[8]	LMI	8,125,063	0.79	13	82	2	6,418,800	831,235	681,376	16,625	0.086	254,770	738,264
Seychelles	UMI	91,361	**2.98**	12	37	2	272,256	32,535	12,005	651	0.139	4,619	5,478
Sierra Leone	LI	2,887,017	**0.45**	9	84	2	1,299,158	116,275	97,423	2,325	0.035	36,408	121,312
Singapore	HIC	4,492,494	**1.49**	**13**	0	2	6,693,816	866,849	393	17,337	0.004	6,472	10,878
Slovenia	HIC	336,594	**1.21**	12	1	2	407,279	48,670	550	973	0.005	556	985
Solomon Islands[8]	LMI	618,678	0.79	13	86	2	488,756	63,294	54,608	1,266	0.090	20,394	176,589
Somalia	LI	5,971,169	0.6	9	85	2	3,582,701	320,652	271,753	6,413	0.047	101,531	289,601
South Africa	UMI	12,899,201	**2**	12	54	2	25,798,402	3,082,909	1,664,382	61,658	0.134	630,005	836,279
Spain	HIC	22,771,488	**2.13**	**13**	0	2	48,503,269	6,281,173	0	125,623	0.006	45,853	58,932
Sri Lanka	LMI	14,568,174	**5.1**	**7**	82	2	74,297,687	5,163,689	4,256,120	103,274	0.299	1,591,179	1,918,670
Sudan	LMI	752,529	**0.79**	13	80	2	594,498	76,987	61,277	1,540	0.083	22,928	52,061
Suriname	UMI	402,263	**1.36**	12	15	2	547,078	65,376	9,493	1,308	0.027	3,942	6,394
Svalbard	LMI	2,226	0.79	13	0	2	1,759	228	0	5	0.002	2	4
Sweden	HIC	6,202,234	**1.61**	**3**	0	2	9,985,597	294,575	0	5,892	0.001	2,150	5,245
Syria	LMI	3,621,997	**1.37**	**13**	65	2	4,962,136	642,597	419,763	12,852	0.119	157,904	304,960
Taiwan	HIC	22,211,567	2.1	12	0	2	46,644,291	5,573,993	13,776	111,480	0.006	45,718	64,631
Tanzania	LI	6,688,695	**0.26**	9	84	2	1,739,061	155,646	129,999	3,113	0.020	48,586	214,196
Thailand[8]	UMI	26,043,442	1.2	12	73	2	31,252,130	3,734,630	2,741,031	74,693	0.108	1,027,739	2,179,508
The Gambia	LI	1,324,214	**0.53**	9	84	2	701,833	62,814	52,485	1,256	0.041	19,616	53,498
Togo	LI	1,991,642	**0.52**	**11**	84	2	1,035,654	113,404	94,721	2,268	0.049	35,401	96,994
Tokelau	LMI	1,379	0.79	13	82	2	1,089	141	116	3	0.086	43	100
Tonga	LMI	102,872	**3.71**	**6**	78	2	381,655	22,708	17,694	454	0.176	6,624	10,272
Trinidad and Tobago	HIC	1,358,433	**14.4**	**25**	3	2	19,561,435	4,880,578	160,103	97,612	0.190	94,066	73,512
Tunisia[8]	UMI	7,274,973	1.2	12	60	2	8,729,968	1,043,231	621,077	20,865	0.088	234,309	440,701
Turkey	UMI	34,042,862	**1.77**	12	16	2	60,255,866	7,200,576	1,187,323	144,012	0.039	485,937	790,235
Turks and Caicos Islands	HIC	22,570	2.1	12	0	2	47,397	5,664	22	113	0.006	49	70
Tuvalu	UMI	11,563	1.2	12	73	2	13,876	1,658	1,218	33	0.108	457	861
Ukraine	LMI	6,812,799	0.79	13	49	2	5,382,111	696,983	338,841	13,940	0.052	128,765	233,388
United Arab Emirates	HIC	2,018,302	**1.66**	12	0	2	3,350,381	400,371	256	8,007	0.004	3,016	5,910
United Kingdom	HIC	43,258,889	**1.79**	12	0	2	77,433,411	9,253,293	0	185,066	0.004	67,549	94,165
United States	HIC	112,925,034	**2.58**	**13**	0	2	291,346,588	37,729,383	0	754,588	0.007	275,424	336,819
Uruguay[8]	HIC	2,433,597	2.1	12	8	2	5,110,554	610,711	46,861	12,214	0.024	21,562	8,037
USVI	HIC	134,219	2.1	12	0	2	281,860	33,682	11	674	0.005	250	296
Vanuatu	LMI	251,851	**3.28**	**9**	81	2	826,071	73,933	60,104	1,479	0.245	22,478	38,006
Venezuela[8]	HIC	16,094,897	2.1	12	5	2	33,799,284	4,039,014	199,585	80,780	0.017	102,333	154,243
Vietnam[8]	LMI	55,858,245	0.79	13	86	2	44,128,014	5,714,578	4,909,870	114,292	0.090	1,833,819	4,172,828
Yemen	LMI	6,048,920	0.79	13	73	2	4,778,647	618,835	451,134	12,377	0.077	169,181	513,907
											Total	31,865,274	69,143,290

Notes:

1 - Based upon 2010 Gross National Income

2 - Based upon a 50 km coastal buffer created in GIS with global population densities[24]

3 - Bold data were taken directly from World Bank estimates[6,21]

4 - Bold data were taken directly from World Bank estimates[6,21]

5 - Using a model developed for this study (see Supplemental Information)

6 - From U.S. national litter study[22]

7 - Calculated values

8 - Economic status changed from 2005 to 2010[6]; waste generation rate and %plastic were assigned using average values for the 2010 economic category

March 31, 2020

Executive Director
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Department of the Environment
Gatineau, Quebec K1A 0H3

by email at eccc.substances.eccc@canada.ca

Re: American Chemistry Council Comments on Draft Science Assessment of Plastic Pollution

Dear Executive Director:

The American Chemistry Council (ACC) and its Plastics Division jointly submits these comments on the Government of Canada's Draft Science Assessment of Plastic Pollution (Draft), published in the Gazette Part 1 on February 1, 2020 under the Canadian Environmental Protection Act, 1999. ACC represents the business of chemistry in the United States, including domestic manufacturers and importers of chemicals. Many of our member companies conduct commercial operations in both the United States and in Canada. ACC's Plastics Division represents the major US producers of plastic resins.

As a literature review of the state of the science regarding plastics and microplastics, the Draft is, in our view, reasonably complete and up-to-date. That said, it is merely a literature review coupled with extremely broad research recommendations non-specific to any plastic product, packaging, or resin. It is simply not legally or technically sufficient to support classification of plastics as CEPA toxic. We have a number of serious concerns about this review and the proposed path forward under Canadian Environmental Protection Act (CEPA):

- First, we believe that aggressive, global and regional public-private commitments and partnerships are in place to drive solutions to plastic waste and marine debris, and should be given an opportunity to work. Global waste, litter, and marine debris challenges require solutions through these partnerships.
- Second, CEPA is not an appropriate regulatory framework to apply to leakage of plastic into the environment – what is fundamentally a solid waste issue, not a chemical management, issue.[1] For that matter, CEPA assesses the individual chemical, while here, a true risk assessment must also take into account the behavior leading to the waste issue.

[1] CEPA is even more ill-suited to evaluating polymers which would be considered low hazard in a chemical management regime. For example, many polymers are so low in toxicity that they are widely considered non-toxic and would be eligible for the polymer exemption under the US Toxic Substance Control Act. Under the revised TSCA in the US, these polymers might be better considered low priority for risk evaluation under that statute.

- Third, the generalized approach to reviewing all macroplastics and microplastics lacks an adequate scientific foundation to support conclusions specific to particular plastic products, packaging, or resins to support further action under CEPA. In short, such an approach is inadequate to support adding "plastics" or "single use plastics" generally as a category under CEPA;
- Fourth, we are deeply concerned that Canada appears to be poised to skip a critical step under CEPA, namely, the development of a scientifically robust risk assessment that presents knowledge of exposures and hazards and integrates these to quantify potential risks to ecological species and human health;
- Fifth, we believe a truncated and incomplete CEPA review that bypasses risk assessment is necessarily inconsistent with Canada's commitments to risk principles under the recently signed US-Mexico-Canada trade agreement; and
- Sixth, and perhaps most importantly, we are deeply concerned that moving forward with risk management action now could jeopardize public health given the key role that many plastic products play in health care, particularly in light of the expanding global coronavirus crisis. In fact, sanitary single-use plastic medical products and food packaging are on the front lines protecting public health during the current crisis – and every day.

We urge Environment and Climate Change Canada and Health Canada to consider an alternative, better suited legal mechanism to address the issue of plastic waste. We likewise urge the agencies to consider the public health consequences of making a CEPA toxic determination that the public would associate with plastics, plastic packaging, or resins - a government determination that surely will be misunderstood and misinterpreted by the public at the worst possible time. In the event that CEPA continues to be used as a platform for regulatory decision making, we urge Environment and Climate Change Canada and Health Canada to consider the basis for specific risk assessments supported by complete scientific assessments for each specific plastic product, packaging, or resin as warranted.

Our specific comments follow.

1. **Partnership-based solutions that harness the power of the private sector should be given a chance to work.**

ACC and our members are deeply committed to ending plastic and other waste in the environment and creating a more circular economy for plastics. We agree that plastic waste in the environment is unacceptable and that the benefits of plastic are diminished when it ends up in the marine environment or improperly on land. We believe these challenges, while significant, are ultimately solvable. The stakes are high: plastics are critical to modern society, from light-weighting vehicles to reduce their emissions, to sealing and insulating our offices and homes, to delivering essential health care, preserving food and preventing food waste, and contributing to an overall higher quality of life.

Likewise, polymers used in contact with foods for food packaging applications already regulated to meet the US Federal Food and Drug Administration's criteria for safety –taking migration into foods into account.

ACC and our member companies have been cornerstones of the global effort to address marine debris and plastic waste. In January 2019, global companies in the plastics value chain, from manufacture to disposal, including many ACC members, announced the creation of the Alliance to End Plastic Waste. This non-profit organization is committing $1.5 billion over five years to help end plastic waste in the environment and will focus on providing solutions to the largest sources of plastic in our ocean. Initially that work will be largely focused on so-called "high leakage" countries—where waste collection and management has not kept pace with growing populations and growing economies. A study in *Science* magazine estimates that almost 60 percent of plastic waste going into our oceans comes from just five countries, primarily in Southeast Asia. Although the United States accounts for less than one percent of this plastic waste, ACC and its members have committed to reusing, recycling or recovering all plastic packaging by 2040 and making all plastic packaging reusable, recyclable or recoverable by 2030.

ACC also helped launch Circulate Capital, a $106 million fund that provides zero-interest financing for waste management infrastructure projects in South and Southeast Asia. The fund seeks to implement many of the findings from the Ocean Conservancy's Trash Free Seas Alliance reports *Stemming the Tide* and *The Next Wave*. Stemming the Tide found that improvements in waste management are critically needed to stop plastic waste in China, Indonesia and the Philippines.

ACC has also led the development of The Declaration of the Global Plastics Associations for Solutions on Marine Litter, announced at the 5th International Marine Debris Conference in 2011. Otherwise known as the Global Declaration, this is a global commitment to combat ocean pollution. Since its inception, seventy-five plastics associations in 40 countries have signed the Declaration and more than 355 projects to address marine debris are planned, underway, or have been completed around the globe. We are working to advance innovative new technologies, increased traditional and advanced recycling infrastructure, develop new uses and end markets for recovered plastics, and a number of other innovative solutions to reduce the amount of plastic that ends up in the environment.

2. CEPA is the wrong tool to address a solid waste problem.

The issue of plastic waste and marine debris is not insignificant, and we agree that it is a matter of public concern and deserves concerted action and meaningful progress. There are many dialogues underway considering whether products are using the right material for the job – the best material to deliver safety as well as environmental benefits across the life cycle. There are different policy approaches available to achieving these ends. But CEPA is designed to evaluate substances with respect to their potential human health and environmental risks. Waste issues are better addressed by policy solutions tailored to them. We encourage Environment and Climate Change Canada and Health Canada, and other sectors of the Canadian government, to work with industry to find viable solutions, including source reduction, innovative product design and delivery systems, increased recycling, advanced (chemical) recycling technologies, and extended producer responsibility programs, to name a few.

When it comes to health and environmental issues, on occasion, there may be multiple legislative options to consider as a platform to evaluate risk and implement risk management solutions.

Indeed, the same chemicals, materials, and products (here, "product") are themselves subject to multiple statutes, with overlapping jurisdiction among agencies. The threshold question must always be whether the statute covers the affected product and gives a regulatory body authority to act. But beyond that, there should be a legal evaluation of whether one statute precludes application of the other, and a policy determination regarding which statute is better suited to regulating the product. This is the "Best Placed Act" principle. A corollary of this principle also applies to sections of a statute, which might be described as the "Best Placed Provision."

CEPA Part 7, International Water Pollution, should be reviewed within this context. Section 175 defines "water pollution" broadly, to include substances that are not otherwise unsafe or toxic – in other words, inert, non-toxic materials – that nevertheless interfere with the normal enjoyment of life or property. This is apt description of unwanted litter and marine debris in water bodies (regardless of substance or material). Section 176 allows the Minister to Act if there is reason to believe that a substance released from a source in Canada into water creates water pollution in a country other than Canada. Given that the US and Canada share an international border with multiple major rivers and the Great Lakes system, this element appears to be met. The Minister has authority under Section 176(3) to "recommend regulations to the Governor in Council for the purpose of preventing, controlling, or correcting the water pollution." While we believe CEPA should not be used at all to address the concerns at issue, that said, it appears that his section of CEPA is better placed to address litter and marine debris concerns than Appendix 1.

3. **The Draft improperly "groups" all plastics, and thus the Draft does not reach individualized findings that support further action on any particular plastic, plastic packaging, or resin.**

Section 64 of CEPA defines a substance as "toxic" if it is entering or may enter the environment in a quantity or concentration or under conditions that:

> have or may have an immediate or long-term harmful effect on the environment or its biological diversity;
>
> constitute or may constitute a danger to the environment on which life depends; or
>
> constitute or may constitute a danger in Canada to human life or health.

The term "plastic" does not denote "a substance" under CEPA. It is generally understood to mean a subset of chemically-distinct polymeric substances. However, there are many thousands of unique polymers used in commerce today, each of which having its own chemical identity, chemical resistance, and other characteristics. In addition, polymers are compounded to make plastic, such that each particular "plastic" used in a particular application is composed of a number of chemically distinct substances.

The Draft presents a literature review that broadly considers available information about macroplastics and microplastics, but does not individually assess each "plastic," either with respect to the specific polymer relevant to that plastic or the relevant and specific additives; each plastic as used in packaging; or each plastic as used in a particular product.

The Draft also does not adequately present specific findings that take into account use, exposure, and environmental fate specific to each plastic, plastic packaging, and resin. It does not support substance-specific findings related to the entry of the substance into the environment in a quantity or concentration or under conditions that justify further action. For example, a particular additive might be used with some frequency in insulation or electronics, but never used in food packaging. To support further action under CEPA with respect to concerns about that particular additive, a scientific assessment would need to be able to identify which products contain the additive of concern, and to describe exposures to the product and quantify the particular health or environmental concern arising from that particular product. The Draft does not do this.

4. Any risk-based review process should include a risk assessment step.

We believe CEPA is not well-suited to evaluate and recommend solutions to an underlying solid waste problem, to the extent a chemical management regime is used for plastics and microplastics, a comprehensive, scientifically robust risk assessment must precede any considerations of potential risk management actions. The global chemical industry supported a multi-stakeholder workshop from which a risk-based framework[2] for microplastics was developed and published in a peer reviewed scientific journal; this framework is available for use and has been cited in the Draft.

The Draft itself is not a risk assessment.[3] For that matter, it is not a problem formulation or scope of a risk assessment. It does even characterize the most important areas for research, data development, or analysis. It is a literature review. The research recommendations contained in the Draft are quite broad. There are no specific research recommendations tied to conclusions specific to particular plastic products, packaging, or resins. At bottom, regardless of the quality and completeness of the literature review itself, it does not adequately support the broad recommendations made for additional research, and the Draft should be revised to make discrete recommendations based on an expert analysis of data or research needs. As presented, the Draft falls well short of presenting specific, discrete recommendations. If a particular product, packaging, or resin is evaluated, using best available science and weight of the evidence, taking into consideration the quality of studies, and as a result, is deemed to present significant enough concern to warrant a risk assessment, then a robust scientific risk assessment could proceed. It appears, however, that the proposed course of action is to skip over the risk assessment. This is unwise and wholly inconsistent with the provisions of CEPA.

We appreciate that CEPA's preamble indicates that the precautionary considerations should be applied "where there are threats of serious or irreversible damage." But it is critical to use risk assessment approaches so that any finding of "threats of serious or irreversible" damage is supported by objective and transparent scientific analysis of exposures (both current and modeled future exposures) and hazards. In this manner, risk management actions, if warranted can be selected to address, and be commensurate with, the specific potential risks identified. The

[2] https://setac.onlinelibrary.wiley.com/doi/full/10.1002/etc.4529
[3] The Draft itself says "is not intended to quantify the risks of plastic pollution on the environment or human health."

Draft does not provide the foundation for such a conclusion, nor does it purport to make that finding with respect to particular sources of plastics and microplastics as contributions to specific damage. Rather, it calls for additional research. The call for additional research is consistent with specific recommendations for particular targeted research that have been advanced in various scientific reviews, including the World Health Organization's 2019 review of microplastics in drinking water[4] and the European Union's SAPEA January 2019 expert review.[5]

Furthermore, risk-based decision making must also take into consideration alternatives analysis. For example, in the United States, the risk management step under a revised Toxic Substances Control Act requires EPA, "in deciding to "whether to prohibit or restrict in a manner that substantially prevents a specific condition of use of a chemical substance or mixture, and in setting an appropriate transition period for such action, [EPA must] consider, to the extent practicable, whether technically and economically feasible alternatives that benefit health or the environment, compared to the use so proposed to be prohibited or restricted, will be reasonably available as a substitute when the proposed prohibition or other restriction takes effect." (emphasis added).

Alternatives analysis makes clear that focusing on plastic product bans without consideration of the availability and environmental impacts of alternatives is counterproductive. Studies by TruCost and Franklin & Associates show that alternatives to plastics have greater environmental impacts such as greater energy use, increased greenhouse gas emissions and more waste. In a 2016 report, the environmental accounting firm TruCost found the natural capital cost of plastic in 16 sectors to be $139 billion but the environmental costs for alternative materials was estimated at $533 billion annually. This 3.8 fold increase in natural capital costs of alternatives included greenhouse gas emissions, marine litter, and other impacts. In a study of plastic packaging compared to alternatives, Franklin Associates found that greenhouse gas emissions would be doubled by banning plastic packaging.

The potential for policies to increase environmental impacts is especially large for packaged goods, such as food, which often requires a significant amount of energy and water to produce. According to the United Nation's Food and Agriculture Organization (FAO), one third of all food produced never reaches the consumer's table. FAO further states that this food waste results in a greenhouse gas impact of 4.4 GtCO2, which would rank third in terms of total greenhouse gas emissions behind only China and the United States. Reducing food waste through improved handling, logistics, and packaging of food is essential to reducing food waste and the associated greenhouse gas emissions. The essential role that plastic packaging plays in reducing food waste must be considered.

5. The process underway is inconsistent with Canada's obligations under the Canada-U.S.-Mexico trade agreement.

The Canadian Parliament ratified the Canada-U.S.-Mexico Agreement (CUSMA) on Friday, March 13, also receiving royal assent that same day. This high standard, comprehensive trade

[4] https://apps.who.int/iris/bitstream/handle/10665/326499/9789241516198-eng.pdf?ua=1
[5] https://www.sapea.info/wp-content/uploads/report.pdf

agreement contains several regulatory cooperation provisions that require Canada to work closely with the United States and Mexico to foster greater regulatory compatibility on chemical substances in North America. These provisions include the following:

> Each Party shall endeavor to use a risk-based approach to the assessment of specific chemical substances and chemical mixtures, where appropriate. Each Party also intends to encourage, as appropriate, a risk-based approach to regulating chemical substances and chemical mixtures both in international fora and in its relations with non-Parties (emphasis added).
>
> The Parties shall endeavor, if appropriate, to align their respective risk assessment methodologies and risk management measures for chemical substances and chemical mixtures provided that alignment does not prevent a Party from determining and achieving its levels of protection. In its alignment efforts, each Party shall strive to continue to improve its levels of protection.
>
> Each Party, when developing, modifying, or adopting a measure concerning chemical substances or chemical mixtures, shall endeavor to consider how a measure adopted by another Party could inform its decision-making.

The Draft does not itself provide a risk-based review consistent with the CUSMA. Without an evaluation of specific plastic, plastic packaging, and resins, as used and as each specific substance does or may enter the environment, the requirement for risk-based assessment cannot be satisfied. In short, the Draft should be revised substantially to make such specific findings, and any subsequent risk assessment should be based on an appropriate revision of the Draft.

Non-compliance with the CUSMA agreement may also implicate Canada's obligations under the Technical Barriers to Trade (TBT) Chapter of CUSMA and the World Trade Organization (WTO) TBT Agreement. Article 2.2 of the TBT Agreement requires WTO Members to "ensure that technical regulations are not prepared, adopted or applied with a view to or with the effect of creating unnecessary obstacles to international trade." Article 2.2 of the TBT Agreement also requires that "technical regulations shall not be more trade-restrictive than necessary to fulfill a legitimate objective, taking account of the risks non-fulfilment would create." In our view, the Draft, if implemented would create an unnecessary obstacle to bilateral trade in used plastics between Canada and the United States, and would be more trade-restrictive than necessary to fulfill a legitimate objective.

Furthermore, as no international standard exists and the technical regulation may have a significant effect on trade of the United States, Canada must honor its obligations under Article 2.9 of the TBT Agreement with respect to transparency and notification. When Health Canada notifies the Draft to the WTO Committee, we request that it provide at least 60 days for interest parties to make comments in writing, consistent with the recommendations of the TBT Committee (see G/TBT/1/Rev.12, paragraph 4.3.1.6).

We urge Health Canada to re-evaluate its use of the Draft under CEPA through the lens of its TBT Agreement obligations.

6. Finalization of the Draft should be deferred until the current global coronavirus crisis is resolved.

Plastics deliver critical health and safety benefits across a wide range of products and packaging. Sanitary, single-use plastics are right now delivering critical health and safety benefits across a wide range of products and packaging. On March 19, 2020, the US Department of Homeland Security issued Guidance on the Essential Critical Infrastructure Workforce: Ensuring Community and National Resilience in COVID-19 Response, which recognizes as essential "single use plastics and packaging that prevents the contamination or supports the continued manufacture of food, water, medicine, and other essential products…" [6] (emphasis added). In light of the current global crisis, plastics will be needed to protect the safety and integrity of food and necessary for use in a wide variety of medical devices and products. Plastics are used in single use surgical and medical gowns; N95 respirators and face masks; protective sheeting; single use disinfecting wipes; surgical gloves and other gloves; food service packaging; packaging for medicines and pharmaceuticals; bottled water; and a wide variety of other critical goods and services.

Globally, regulators and other government bodies are relaxing restrictions and requirements during the crisis. The state of Maine, for example, just extended compliance deadlines for single use plastic bags to ensure that they remain available during the crisis as part of its coronavirus response plan.[7] In Massachusetts, to help protect the health of workers, the Governor banned the use of reusable shopping bags and lifted local bans of plastic bags in grocery stores and pharmacies.[8] New York has announced an enforcement delay of its plastic bag ban.[9] Other states are now following suit, such as New Hampshire. In the meantime, the US Federal Food and Drug Administration has reduced inspections of imported and domestic foods as part of its coronavirus response.[10]

At the same time, global health authorities and businesses are issuing recommendations and requirements for use of face masks for individuals showing symptoms, as well as practices that necessitate use of plastic to protect foods, medical devices and other items, food preparation and delivery, and other critical services.

Making a CEPA-toxic determination at this time could confuse consumers, businesses, and others, and lead to choices that impede the global coronavirus response, impacting public health and potentially the spread of the virus. At a minimum, we urge Canada to delay further action until this crisis is abated – but we further urge Canada to explore better tailored approaches to addressing marine debris and plastic waste, removing this issue from the CEPA process.

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[6] https://www.cisa.gov/publication/guidance-essential-critical-infrastructure-workforce
[7] https://www.plasticsnews.com/news/maine-halts-plastic-bag-ban-part-plan-mitigate-coronavirus-spread
[8] www.wwlp.com/news/health/coronavirus-local-impact/plastic-bag-ban-lifted-during-coronavirus-outbreak
[9] https://nypost.com/2020/03/19/enforcement-of-new-yorks-plastic-bag-ban-postponed-due-to-coronavirus
[10] https://www.fda.gov/news-events/press-announcements/coronavirus-covid-19-update-fda-issues-temporary-policy-fsma-onsite-audit-requirements

ACC appreciates the opportunity to offer comments on the Draft.

Best regards,

Chris Jahn
President and CEO
American Chemistry Council

cc: Hon. Jonathan Wilkinson, Minister of the Environment and Climate Change
Hon. Patty Hajdu, Minister of Health
Mary Ng, Minister of Small Business, Export Promotion and International Trade
Chrystia Freeland, Deputy Prime Minister and Minister of Intergovernmental Affairs
Aldona Wos, US Ambassador to Canada

A Guide to Understanding the *Canadian Environmental Protection Act, 1999*

December 10, 2004

Table of Contents

1 Introduction

This Guide explains the key features of the *Canadian Environmental Protection Act, 1999* (CEPA 1999). CEPA 1999 is an important part of Canada's federal environmental legislation aimed at preventing pollution and protecting the environment and human health. The goal of CEPA 1999 is to contribute to sustainable development—development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs.

CEPA 1999 came into force on March 31, 2000 following an extensive Parliamentary review of the former CEPA. CEPA 1999 contains significant improvements for the protection of the environment over the former Act. It:

- makes pollution prevention the cornerstone of national efforts to reduce toxic substances in the environment;
- sets out processes to assess the risks to the environment and human health posed by substances in commerce;
- imposes timeframes for managing toxic substances;
- provides a wide range of tools to manage toxic substances, other pollution and wastes;
- ensures the most harmful substances are phased out or not released into the environment in any measurable quantity;
- includes new provisions to regulate vehicle, engine and equipment emissions;
- strengthens enforcement of the Act and its regulations;
- encourages greater citizen input into decision-making; and
- allows for more effective cooperation and partnership with other governments and Aboriginal peoples.

This Guide describes CEPA 1999's:

- role in environmental management in Canada;
- objectives and guiding principles;
- environmental protection management process; and
- key programs aimed at protecting the environment and human health.

For further information on CEPA 1999, please refer to the list of contacts and sources in Appendix A and the various websites listed in Appendix B.

2 Environmental Management in Canada

In Canada, each level of government has powers to protect the environment. This shared nature of environmental jurisdiction makes close cooperation among the federal, provincial, territorial and Aboriginal governments important to Canada's environmental well-being.

Within the federal government, CEPA 1999 is the primary element of the legislative framework for protecting the Canadian environment and human health. A key aspect of CEPA 1999 is the prevention and management of risks posed by toxic and other harmful substances. CEPA 1999 also manages environmental and human health impacts of products of biotechnology, marine pollution, disposal at sea, vehicle, engine and equipment emissions, fuels, hazardous wastes, environmental emergencies and other sources of pollution. The Minister of the Environment is accountable to Parliament for the administration of all of CEPA 1999. Both the Minister of the Environment and the Minister of Health jointly administer the task of assessing and managing the risks associated with toxic substances.

Efforts taken under CEPA 1999 are complemented by actions taken under other federal Acts administered by the Minister of the Environment. The *Fisheries Act*, which is administered by the Minister of the Environment on behalf of the Minister of Fisheries and Oceans, includes provisions to prevent pollution of waters inhabited by fish. Through the *Canada Water Act*, water resources and their environmental quality are managed. The Minister of the Environment also manages some aspects of wildlife through the *Species at Risk Act*, the *Canada Wildlife Act*, the *Migratory Birds Convention Act, 1994*, and the *Wild Animal and Plant Protection and Regulation of International and Interprovincial Trade Act*. Efforts under the *Canadian Environmental Assessment Act* ensure that the environmental effects of various projects are carefully reviewed before action is taken in order to avoid significant adverse environmental effects.

There are also a number of specialized Acts administered by other federal departments that are useful in protecting the environment. Several Acts are used to control, among other things, the introduction of new substances and products of biotechnology into the Canadian market so that the risk to the environment and human health is reduced. These Acts include the *Pest Control Products Act*, the *Feeds Act*, the *Seeds Act*, and the *Health of Animals Act*. In addition to the previously mentioned *Fisheries Act* and the *Canada Water Act*, the federal government also has a number of other Acts designed to protect our waters. The *Arctic Waters Pollution Prevention Act* was introduced to prevent pollution of waters in the Canadian arctic. The *Oceans Act* includes provisions for the protection of marine areas. Several Acts contain provisions that ensure environmentally responsible actions. Examples include the *Canada Shipping Act* and the *Transportation of Dangerous Goods Act*.

Canada is intricately linked to other countries around the globe economically, environmentally and socially. While global and regional environmental problems impact on Canada's vast geography (e.g., ozone depletion, persistent organic pollutants, climate change), Canada also has a responsibility to reduce its contributions to these problems. Canada has a long history of international cooperation across a broad range of environmental issues. Arrangements range from informal sharing of information to the adoption of formal cooperative agreements to achieve common goals. CEPA 1999 provides the means and opportunity to cooperate with international governments to achieve Canada's environmental policy and regulatory goals.

3 CEPA 1999 Guiding Principles

CEPA 1999 sets out several guiding principles in the preamble and embodies them in the administrative duties of the government. Key among them include:

Sustainable Development — The Government of Canada's environmental protection strategies are driven by a vision of environmentally sustainable economic development. This vision depends on a clean, healthy environment and a strong, healthy economy that meets the needs of the present generation without compromising the ability of future generations to meet their own needs.

Pollution Prevention — CEPA 1999 shifts the focus away from managing pollution after it has been created to preventing pollution. Pollution prevention is "the use of processes, practices, materials, products, substances or energy that avoid or minimize the creation of pollutants and waste and reduce the overall risk to the environment or human health."

Virtual Elimination — CEPA 1999 requires the virtual elimination of releases of substances that are persistent (take a long time to break down), bioaccumulative (collect in living organisms and end up in the food chain), toxic (according to CEPA 1999 Section 64) and primarily the result of human activities. Virtual elimination is the reduction of releases to the environment of a substance to a level below which its release cannot be accurately measured.

Ecosystem Approach — Based on natural geographic units rather than political boundaries, the ecosystem approach recognizes the interrelationships between land, air, water, wildlife and human activities. It also considers environmental, social and economic elements that affect the environment as a whole.

Precautionary Principle — The government's actions to protect the environment and health are guided by the precautionary principle, which states that "where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation."

Intergovernmental Cooperation — CEPA 1999 reflects that all governments have the authority to protect the environment and directs the federal government to endeavour to act in cooperation with governments in Canada to ensure that federal actions are complementary to and avoid duplication with other governments.

National Standards — CEPA 1999 reinforces the role of national leadership to achieve ecosystem health and sustainable development by providing for the creation of science-based, national environmental standards.

Polluter Pays Principle — CEPA 1999 embodies the principle that users and producers of pollutants and wastes should bear the responsibility for their actions. Companies or people that pollute should pay the costs they impose on society.

Science-based Decision-Making — CEPA 1999 emphasizes the integral role of science and traditional aboriginal knowledge (where available) in decision-making and that social, economic and technical issues are to be considered in the risk management process.

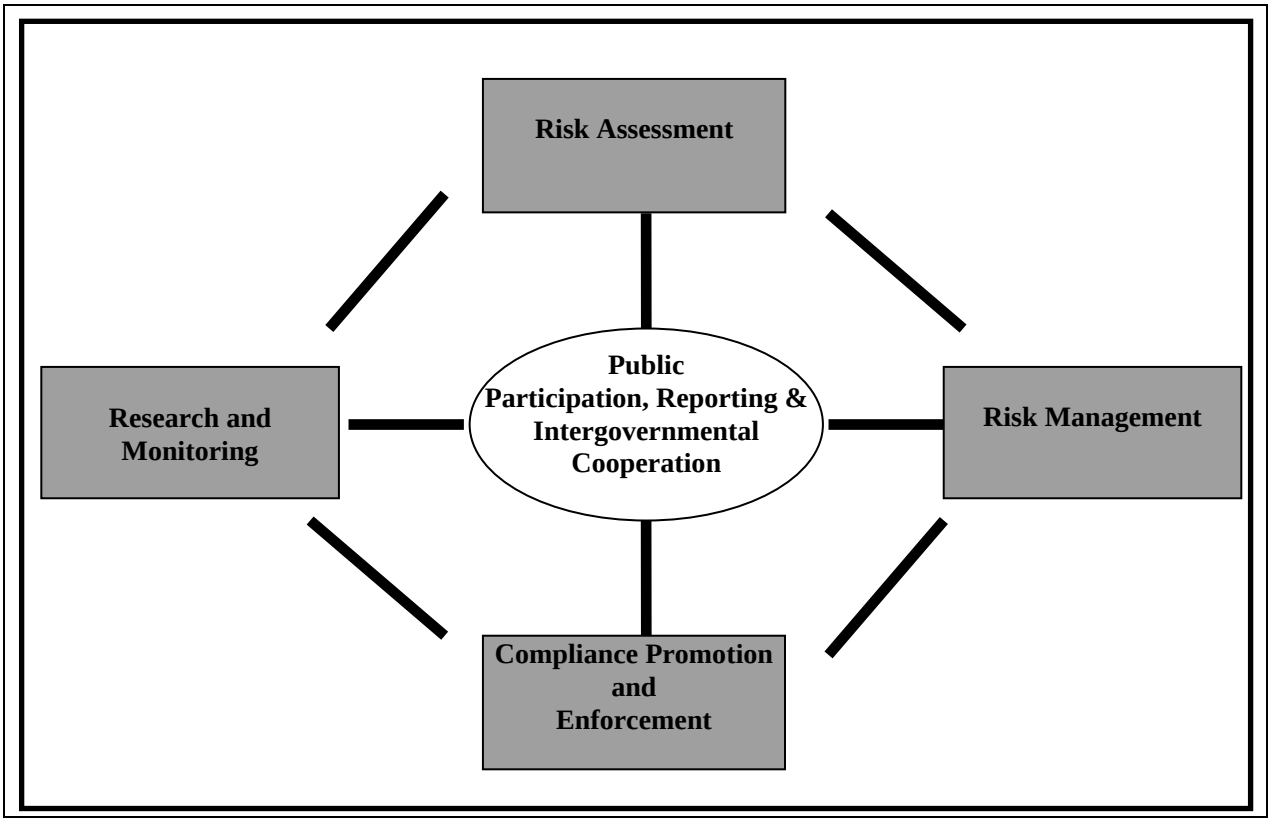
4 Environmental Protection Management Process

The environmental management process used in the implementation of CEPA centres around four key activities (see Figure 1)

- research and monitoring
- risk assessment
- risk management; and
- compliance promotion and enforcement.

Each stage of the process includes information exchange in the form of cooperation with other governments, public participation and reporting on progress.

Figure 1: Environmental protection management process



Research and Monitoring — Scientific research and development are used to evaluate the impact of substances on the environment and human health, determine the extent of exposure to contaminants, guide the development of preventive and control measures by identifying pollution prevention and technology solutions, and provide specialized sampling and analytical techniques used in compliance promotion and enforcement. Monitoring changes in the environment and in human health trends is essential for assessing the impact of toxic substances and the effectiveness of measures meant to minimize environmental damage and real and potential threats to human life. Information gathering on the use and release of substances informs understanding and decision-making by governments, industry and the public.

Risk Assessment — Substance risk assessments are based on sound science, which supports a better understanding of their impacts and exposure to the environment and human health. The assessments incorporate the precautionary principle and a weight of evidence approach. Risk assessment also helps to identify the sources of pollution that pose the greatest risk to the environment and human health. While risk assessment is the prelude to, and informs, the risk management stage for all programs under CEPA 1999, the Act provides explicit direction on the assessment of toxic substances and the assessment of wastes and other matter that are destined for disposal at sea.

Risk Management — Based on the scientific information available, strategies are developed to determine how best to manage toxic and other substances and what kinds of actions are required. Social, economic and technology factors are integral to risk management decision making, including considering which risk management instruments are the most cost-effective. While CEPA 1999 provides for certain instruments developed under the Act such as regulations, pollution prevention plans, guidelines and codes of practice, other tools such as voluntary agreements, other Acts of Parliament or provincial/territorial actions may also be suitable to manage particular risks posed by a substance.

Compliance Promotion and Enforcement — Compliance promotion and enforcement of CEPA 1999 and its regulations are necessary to achieve the highest level of environmental quality for all Canadians. Providing public opportunities for input to the creation of regulations and compliance promotion programs should result in a high rate of compliance. In cases of non-compliance, CEPA 1999 enforcement officers will investigate. If an alleged violation is confirmed, action will be taken using one or more of the enforcement tools available under CEPA 1999. Information gathered during the compliance promotion and enforcement stage helps to evaluate the effectiveness of controls and monitoring.

5 Existing Substances

CEPA 1999 includes specific requirements for the assessment and management of substances currently existing in commerce or being released to the environment in Canada. The Minister of the Environment and the Minister of Health jointly administer this part of the Act.

5.1 What are Existing Substances?

There are currently about 23 000 substances, which can be manufactured in, imported into, or used in Canada on a commercial scale, that have not been assessed for the risks they pose to the environment or human health. These substances comprise the Domestic Substances List. Substances not on this list are considered to be new to Canada. A substance as defined under CEPA 1999 includes any distinguishable kind of organic or inorganic matter, whether animate or inanimate that is capable of being released as a single substance, an effluent, emission, waste or a mixture into the Canadian environment.

What is the Domestic Substances List?

The Domestic Substances List includes substances that were, between January 1, 1984, and December 31, 1986, in commercial use in Canada, or were used for commercial manufacturing purposes, or were manufactured in or imported into Canada in a quantity of 100 kg or more in any one calendar year. The list is regularly amended to include additional substances that have been assessed under the Act and allowed into Canada. The Domestic Substances List currently contains approximately 23 000 substances from the original list along with an additional 1954 substances that have been added to the list following assessments of new substances.

CEPA 1999 introduced more processes for assessing these substances to determine if they are toxic according to CEPA 1999. The three key assessment processes are:

- categorization and screening assessment of the Domestic Substances List;
- assessment of the Priority Substances List; and.
- review of other jurisdictions' decisions.

Other assessments may be triggered by information provided by other programs, industry and scientific research.

5.2 How are the Risks Assessed?

5.2.1 What are Risk Assessments?

Risk assessments done under CEPA 1999 consider impacts on human and non-human organisms and the physical environment. These assessments consider not only the hazard posed by a substance, but the exposure or likelihood that a person, organism or the environment will come in contact with that substance. The exposure or potential for exposure of a substance depends on the amount of substance

released into the environment and its fate. The conclusion of the assessment is based on the application of the precautionary principle and a weight of evidence approach.

5.2.2 What are Categorization and Screening Assessments?

Under CEPA 1999, all 23 000 substances on the Domestic Substances List that have not been subject to notification and assessment as new substances must be "categorized" by September 13, 2006, along with all living substances added to the list. Categorization is essentially an initial priority setting mechanism, which involves the systematic identification of substances on the Domestic Substances List that meet the following criteria:

- are inherently toxic (cause toxic effects) to humans or non-human organisms and display either the characteristics of persistence (take a long time to break down) or bioaccumulation (collect in living organisms and end up in the food chain); or

- may present to individuals in Canada the greatest potential for exposure.

What is Toxic under CEPA 1999?

Determining a substance to be toxic under CEPA 1999 is a function of its release or possible release into the environment, the resulting concentrations in environmental media and its inherent toxicity. Section 64 of CEPA 1999 defines a substance as toxic "if it is entering or may enter the environment in a quantity or concentration or under conditions that:

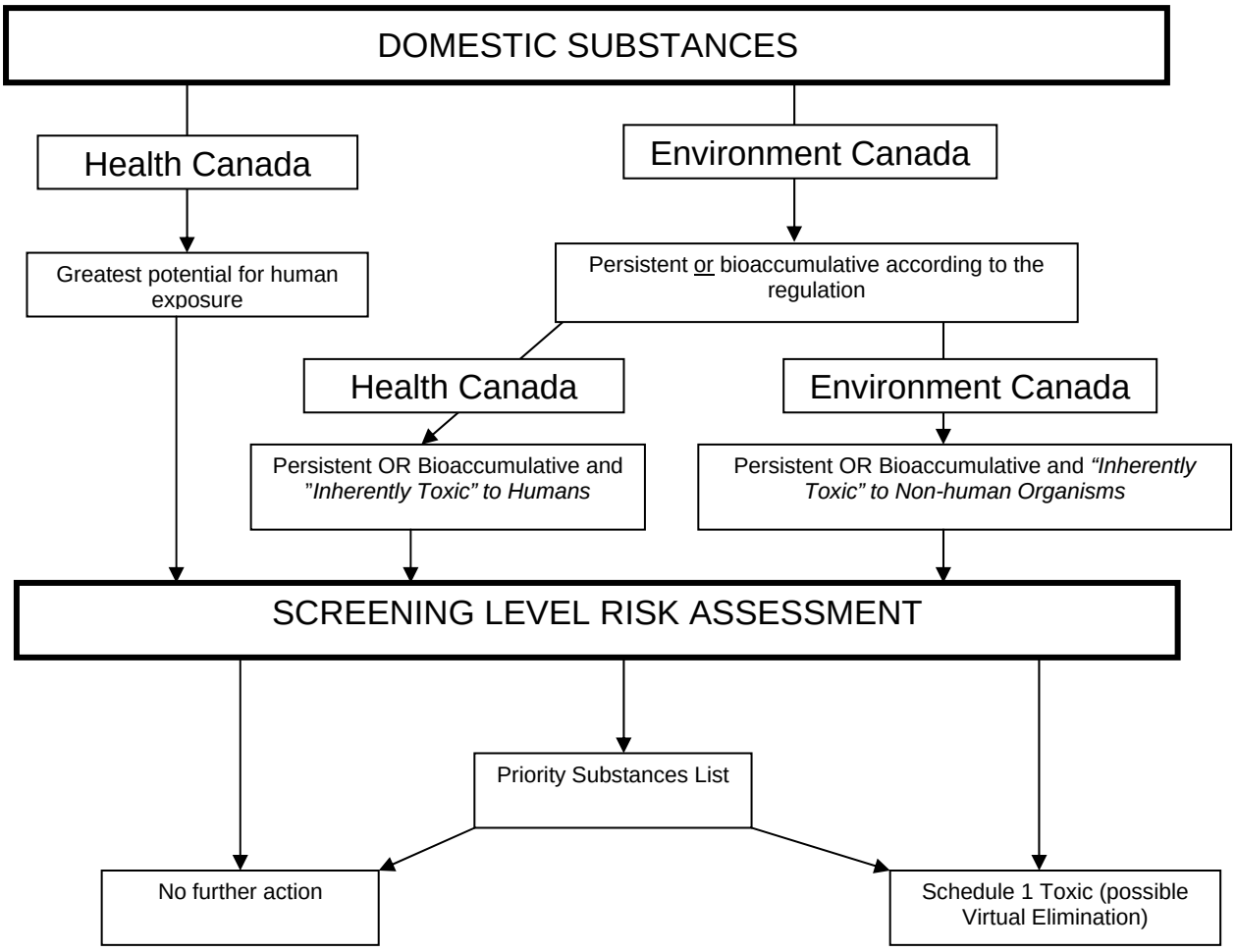
- *have or may have an immediate or long-term harmful effect on the environment or its biological diversity;*

- *constitute or may constitute a danger to the environment on which life depends; or*

- *constitute or may constitute a danger in Canada to human life or health."*

Substances that meet the specified criteria will undergo a screening level risk assessment. A screening assessment involves an analysis of a substance to determine whether the substance is toxic or capable of becoming toxic as defined in CEPA 1999.

Figure 2: Categorization and Screening Process



5.2.3 What is the Priority Substances List?

The CEPA 1999 Priority Substances List continues to be the method used to focus on those chemicals and other substances that require investigation on a priority and in-depth basis to determine if they are toxic under the Act. Substances can be added to the Priority Substances List when a more comprehensive assessment is required following a screening assessment or review of another jurisdiction's decision. Also, any person may ask the Minister to add a substance to that list. CEPA 1999 requires that the substance be assessed within five years from the date the substance is added to the list.

Priority Substances Lists

The first Priority Substances List was established in 1989. Out of the 44 chemicals on the first list, 25 were declared to be toxic under the previous CEPA. The second Priority Substances List of 25 more substances was published in 1995. Out of the 23 assessments published, 18 substances were deemed to be toxic The Ministers of the Environment and Health suspended the assessment period for the other two substances (aluminum compounds and ethylene glycol) in order for Health Canada to collect new or additional information required to assess whether the substances are toxic or capable of becoming toxic.

5.2.4 What is the Review of Decisions of Other Jurisdictions?

CEPA 1999 calls for cooperating and developing procedures for exchanging information on substances with other governments in Canada and member states of the Organization for Economic Co-operation and Development. When the Minister of the Environment receives information that another government has prohibited or substantially restricted a substance for environmental or health reasons, the Ministers of the Environment and Health are obliged to review the decision. The review determines whether the substance is toxic or capable of becoming toxic in the Canadian environment. In this way, Canada will benefit from a streamlined decision-making process through the sharing of scientific data, the capacity of other governments and efforts by others to develop risk management measures.

5.2.5 What are the Outcomes of a Risk Assessment or Review of a Decision by Another Jurisdiction?

Under CEPA 1999, once the Ministers have conducted a risk assessment of an existing substance under the Priority Substances List, a screening level risk assessment or a review of a decision by another jurisdiction, they must propose one of three measures:

- They may add the substance to the Priority Substances List. Typically, they will do this if they decide that there is a need for a more comprehensive risk assessment.

- They may recommend that the Governor in Council (the federal Cabinet) add the substance to the List of Toxic Substances (Schedule 1) and, if applicable, to the Virtual Elimination List. They will typically add the substance to Schedule 1 if they determine that the substance meets the criteria for "toxic" under the Act and that regulatory or pollution prevention or environmental emergency planning risk management measures should be taken under CEPA 1999.

What is the List of Toxic Substances?

Substances that meet the definition of toxic under CEPA 1999 can be placed on Schedule 1 of the Act, the List of Toxic Substances. This does not control the substance but allows the Government to proceed with regulations, pollution prevention plans or environmental emergency plans.

- They may propose no further action under CEPA 1999. They will typically do this if they determine that the substance is not "toxic." They also may propose no further action under CEPA 1999 if they determine that the substance is toxic but that actions being taken or about to be taken under other federal acts or by provincial, territorial or Aboriginal governments are sufficient to manage the risks in a timely manner.

5.2.6 What are the Other Triggers for Risk Assessment?

Other assessments may be triggered by information provided by other programs, industry and scientific research. Substances can be added to the List of Toxic Substances based on any assessment process that satisfies the Ministers that a substance is toxic, without having gone through one of the three types of CEPA 1999 assessments already discussed. Any other type of assessment can be used that satisfies the Governor in Council, on the recommendation of the Ministers of Environment and Health, that a substance is toxic. The other types of assessments that have been used in the past to add a substance to the list were based on collaborative efforts nationally or internationally.

CEPA 1999 allows the government to require persons to submit information on substances where a significant new activity for a substance has been identified. A significant new activity is an alternative use of the substance or other activity that results or may result in:

- a significantly greater quantity or concentration of the substance in the environment; or

- a significantly different manner or circumstances of exposure of the environment to the substance.

Significant new activities can apply to existing substances on the Domestic Substances List or new substances. The government assesses the new information on the substance to determine if it is toxic in relation to the significant new activity.

CEPA 1999 requires that persons who obtain new information on a substance that indicates it might be toxic must submit this information to the government.

5.2.7 What are the Opportunities for Public Participation?

Summaries of the assessment conclusions and the proposed measure (no further action, addition to the Priority Substances List, or addition to the List of Toxic Substances) are published in the *Canada Gazette*, Part I for a 60-day public comment period. Interested parties may bring forward additional scientific evidence to support or refute the Ministers' decision or file a notice of objection requesting that a Board of Review be established (see 18.3 for more information). Depending on the nature of the comments received, the Minister of the Environment then determines if further discussions or a Board of Review are warranted.

After taking into account any information provided during this 60-day period, the Ministers publish their final decision in the *Canada Gazette,* Part I. The Gazette notices are published on the CEPA Environmental Registry, a website found at http://www.ec.gc.ca/CEPARegistry.

5.3 How are the Risks Managed?

5.3.1 What Risk Management Measures are Available?

Examples of risk management measures under CEPA 1999 for existing substances include regulations, pollution prevention plans, environmental emergency plans, guidelines, codes of practice and administrative agreements. These measures may target any aspect of the substance's life cycle, from the research and development stage through manufacture, use, storage, transport and ultimate disposal. Risk management measures for toxic substances are developed through the Toxics Management Process. For regulations, pollution prevention plans or environmental emergency plans the substance must be on the List of Toxic Substances or in the case of environmental emergency plans be, at least, recommended for addition to the List.

CEPA 1999 provides the authority for various risk management measures:

- Regulations impose restrictions on an activity related to a substance, or set limits on the concentrations of a substance that can be used, released to the environment or be present in a product;

- Pollution prevention plans require the preparation and implementation of a plan outlining actions to prevent or minimize the creation or release of pollutants and waste;

- Environmental emergency plans require persons to prepare and implement a plan regarding the prevention of, preparedness for, response to, and recovery from an environmental emergency;

- Environmental quality objectives recommend qualitative or quantitative goals or purposes for pollution prevention or control of toxic substances. They often recommend ambient environmental quality targets or maximum acceptable levels.

- Environmental codes of practice recommend procedures, practices, or quantities of releases relating to facilities and activities during any phase of development of and operation involving a substance, and any subsequent monitoring activities.

- Environmental quality guidelines can be developed to recommend a concentration for toxic substances in surface water, agricultural water, soil, sediment, and human and animal tissue. Guidelines may also be developed to prevent, prepare for, or respond to an environmental emergency or to restore environmental quality.

- Environmental release guidelines include standards expressed as concentrations or quantities, for the release of substances into the environment from facilities or activities.

- Agreements respecting environmental data and research are usually cooperative arrangements with a provincial, territorial, aboriginal or foreign government or any person respecting the creation, operation, and maintenance of a system for monitoring environmental quality.

- Administrative agreements are usually work-sharing arrangements between the federal government and provincial, territorial, or aboriginal governments or aboriginal peoples respecting the administration of CEPA 1999.

In developing risk management measures, the government must give priority to pollution prevention actions. When substances are inherently toxic to humans or non-human organisms, persistent, bioaccumulative, and present in the environment primarily as a result of human activity but are not naturally occurring radionuclides or naturally occurring inorganic substances, then they must be recommended for addition to the List of Toxic Substances. They are also proposed for virtual elimination of releases to the environment and added to the Virtual Elimination List. Virtual elimination is the reduction of releases to the environment of a substance to a level below which its release cannot be accurately measured (the level of quantification). Risk management measures are developed through the Toxics Management Process.

Toxics Management Process

Environment Canada is committed to considering the full range of potential preventive and control measures and recognizing other governments' roles when developing strategies to manage toxic substances under CEPA 1999. The National Advisory Committee of CEPA 1999 plays a key role in advising the federal government on activities under the Act and on cooperative, coordinated approaches to the management of toxic substances.

Risk management measures for toxic substances are developed through the Toxics Management Process. This process allows the federal government to meet the obligations set out in CEPA 1999 and ensures that stakeholder consultations are effective. Central to the toxics management process is the development of a risk management strategy. The risk management strategy, which can vary in format, outlines the proposed approach for managing the risks to the environment and human health for a particular toxic substance.

In developing the risk management strategy, Environment Canada and Health Canada identify the sources that pose the greatest risk to the environment and human health, guided by the science in the risk assessment. A risk management objective is then identified for these sources. This objective is usually based on results achieved from the best available processes, products or techniques used by the sector or, in some cases, environmental quality objectives.

Once an objective has been set, the management measures that could achieve the risk management objective for each source are selected. All available tools, including existing management initiatives, are initially considered. These include instruments under CEPA 1999 as well as other risk management tools that are outside of CEPA 1999, including the regulatory provisions of other governments and voluntary approaches. The suite of tools can comprise a combination of measures representing the most feasible options for managing the substance. For a toxic substance that is subject to the time-clock provisions, at least one of the risk management measures must be a CEPA 1999 instrument. For example, there may be cases in which a new regulation or pollution prevention plan under CEPA 1999 would be the best option for addressing risks posed by one source and would satisfy the time clock requirements of CEPA 1999, while provinces, territories or aboriginal governments may be better situated to address another source, and an existing voluntary agreement may sufficiently address yet another source.

5.3.2 What are CEPA 1999's Time-Clock Provisions?

For a substance found to be toxic through a Priority Substances List assessment, a screening assessment, or the review of a decision by another jurisdiction, and when that substance has been proposed for addition to the List of Toxic Substances, a proposed regulation or instrument establishing "preventive or control actions" for managing the substance must be developed with 24 months. The proposal is

published in the *Canada Gazette*, Part I, for a 60-day comment period. Once proposed, the Ministers have a further 18 months to finalize the regulation or instrument. The Gazette notices are also published on the CEPA Environmental Registry website.

For a risk management instrument to satisfy CEPA 1999's requirements, it must not simply be made under a provision of CEPA 1999 but must also pass the "legal test" of establishing preventive or control actions that reduce or eliminate the risks to the environment or human health. Each instrument is assessed on a case-by-case basis to determine whether this requirement is met.

The time clock provisions do not apply to substances added to the List of Toxic Substances on the basis of assessments that are not the formal CEPA 1999 assessments (i.e. through assessments other than Priority Substances List assessment, a screening assessment, or the review of a decision by another jurisdiction). However, all of the risk management processes, tools and instruments available to the government for toxic substances described above are also available when substances are listed in this manner.

5.3.3 What are the Opportunities for Public Participation?

Within the Toxics Management Process, the government may hold preliminary consultations with the most affected stakeholders during the development of the risk management strategy.

CEPA 1999 also provides formal opportunities for public participation during the risk management stage. Proposed instruments are published in the *Canada Gazette*, Part I for a 60-day comment period and on the CEPA Environmental Registry. Interested parties can provide comments on the proposed regulation or instrument or file a notice of objection requesting that a Board of Review be established. A Board of Review inquires into the nature and extent of the danger posed by the substance that is the subject of the order or the proposed instrument or regulation (see section 18.3 for more information). Depending on the nature of the comments received, the Minister of the Environment then determines if further discussions or a Board of Review are warranted.

After taking into account any information provided during this 60-day period, the Ministers publish the final instrument in the *Canada Gazette*, Part I or II depending on whether the measure consists of a regulation or other instrument, as well as on the CEPA Environmental Registry.

5.4 How are Exports of Substances Managed?

CEPA 1999 provides the authority to establish an Export Control List (Schedule 3 of the Act) containing substances whose export is controlled because their use in Canada is prohibited or severely restricted or because Canada has accepted, through an international agreement, to control their export. Prohibited substances can be exported only if they are to be destroyed or if the export is in compliance with regulations. Regulations can be made addressing:

- prohibitions on export;
- conditions under which an export may be made;
- the type of information to be provided to the Minister with respect to the export; and

- the type of information to accompany an export and to be kept by the exporter.

Details concerning these exports are made public through the CEPA Environmental Registry website. These provisions of CEPA 1999 allow the federal government to ratify the *Rotterdam Convention on Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade*.

6 New Substances

6.1 What are New Substances?

CEPA 1999 ensures that no new substances are introduced into the Canadian marketplace before they have been assessed to determine whether or not they are toxic or capable of becoming toxic to the environment or human health. Substances that are not on the Domestic Substances List are considered to be new to Canada and must be notified. New substances that are accepted as being in commercial use internationally are listed on the Non-Domestic Substances List. Substances on the Non-Domestic Substances List must also be notified, but are subject to lesser information requirements. New substances cannot be manufactured or imported until:

- the Minister has been notified prior to manufacturing or importation of the substance;
- relevant information needed for an assessment of its potential toxicity has been provided to the Minister and the appropriate fee has been paid; and
- the period for assessing the information (as set out in regulations) has expired

What is the Non-Domestic Substances List?

The Non-Domestic Substances List is an inventory of substances that are not on the Domestic Substances List but are accepted as being in commercial use internationally. The list is based on the United States Environmental Protection Agency's Toxic Substances Control Act Chemical Substances Inventory, and contains more than 58 000 entries.

The risks of substances determined to be or suspected of being toxic or capable of becoming toxic may be managed, as necessary, through conditions or prohibitions imposed on their import or manufacture.

If these notification and assessment requirements are met by another federal Act, then the CEPA 1999 requirements do not apply. This means that CEPA 1999 in effect acts as a "safety net"—unless new substances fall under other Acts that are specifically listed in Schedule 2 regarding chemicals and polymers, CEPA 1999 requirements will apply to all new substances. Federal Acts and regulations currently listed on Schedule 2 are the *Pest Control Products Act, Feeds Act* and *Fertilizers Act,* as well as their regulations.

6.2 How are the Risks Assessed?

Anyone interested in manufacturing or importing a new substance will be required to provide specific information for risk assessment purposes. Importers or manufacturers may also be required to provide information on "significant new activities," where a substance's exposure may change significantly based on factors such as new uses or volume of use (see Section 6.4).

Environment Canada and Health Canada evaluate new substances for risks to the environment and human health. A new substance assessment results in one of the following outcomes:

- if the substance is not suspected to be toxic, the notifier may import or manufacture the substance after the assessment period has expired;

- if the substance is suspected of being toxic or becoming toxic, the government may take risk management measures;

- if the substance is not suspected of being toxic but a significant new activity could result in the substance becoming toxic, the substance can be subject to re-notification under certain conditions.

6.3 How are the Risks Managed?

The government can take the following risk management measures for new substances that are toxic or suspected to be toxic:

- permit the manufacture or import of the substance subject to specified conditions;

- prohibit the manufacture or import of the substance for a period not exceeding two years unless replaced by a regulation; or

- prohibit the manufacture or import of the substance until additional information or test results have been submitted and assessed.

The government must undertake these risk management measures and publish them in the *Canada Gazette*, Part I before the expiration of the assessment period. The Gazette notices are also made public on the CEPA Environmental Registry website.

6.4 What is a Significant New Activity?

A significant new activity is an alternative use of a substance or other activity that results or may result in:

- a significantly greater quantity or concentration of the substance in the environment; or

- a significantly different manner or circumstances of exposure to the substance.

If there is a suspicion that a significant new activity in relation to the substance may result in the substance becoming toxic, the substance can be subject to a Significant New Activity Notice. The Notice communicates the criteria under which the government must be re-notified. The government assesses the new information on the substance to determine if it is toxic in relation to the significant new activity. Significant new activities can apply to existing substances on the Domestic Substances List or to new substances.

7 Animate Products of Biotechnology

Animate products of biotechnology (living organisms) are dealt with under a separate part of CEPA 1999, which mirrors the new substances requirements but with some differences to account for the special characteristics of living organisms. Important differences from the way chemicals and polymers are handled as new substances include recognition that:

- quantity thresholds or limitations used for chemicals and polymers are not relevant for notification of new organisms because organisms are capable of reproduction; and

- special regulation-making powers allow for implementing international agreements and respecting the safe and effective use of living organisms in pollution prevention.

Notification and assessment requirements are met by another federal Act, then the CEPA 1999 requirements do not apply. This means that CEPA 1999 in effect acts as a "safety net"—unless new substances fall under other Acts that are specifically listed in Schedule 4 regarding animate products of biotechnology, CEPA 1999 requirements will apply to all new animate products of biotechnology. Federal Acts and regulations currently listed on Schedule 4 are the *Pest Control Products Act,* the *Fertilizers Act*, the *Feeds Act*, the *Seeds Act* and the *Health of Animals Act*, as well as the Regulations under those Acts.

There are currently 35 living organisms listed on the Domestic Substances List.

8 Marine Environment and Disposal at Sea

8.1 Land-Based Sources of Marine Pollution

8.1.1 What are Land-based Sources of Marine Pollution?

The major threats to the health, productivity and biodiversity of the marine environment result from human activities on land in coastal areas and further inland. About 80% of the pollution load in the oceans originates from land-based activities. This includes wastes and run-off from municipal, industrial and agricultural activities, as well as deposits from the atmosphere. These contaminants affect the most productive areas of the marine environment, including estuaries and nearshore coastal waters. The marine environment is also threatened by physical alterations of the coastal zone, including destruction of habitats of vital importance to maintain ecosystem health. The impacts from land-based activities include closures of shellfish growing areas, degraded beaches, destroyed habitat and contaminated sites.

8.1.2 Who Protects Canada's Marine Environment?

The protection of the marine environment in Canada is a responsibility shared by all levels of government. The CEPA 1999 provisions are intended to complement existing regulatory measures and supplement the authority that exists in other federal, provincial, territorial and aboriginal government laws.

8.1.3 How is CEPA 1999 Used to Manage Land-based Sources of Marine Pollution?

CEPA 1999 provides the authority to issue non-regulatory objectives, guidelines and codes of practice to prevent and reduce marine pollution from land-based sources. This is done after consultation with other affected governments.

Keeping in mind the shared responsibility and cost-effectiveness of building on existing programs, Environment Canada, Fisheries and Oceans Canada, and the provinces and territories developed a National Programme of Action for the Protection of the Marine Environment from Land-based Activities. As a national framework and plan, the Programme provides an assessment of the state of Canada’s coastal and marine environment and identifies the management objectives strategies and priority actions that need to be implemented.

8.2 Disposal at Sea

8.2.1 What is Disposal at Sea?

Each year in Canada, two to three million tonnes of material is disposed of at sea. Most of this is material dredged from ocean floors that must be moved to keep shipping channels and harbours clear for navigation and commerce. CEPA 1999 covers the disposal of certain substances at sea from ships, aircraft, platforms or other structures. Discharges from land-based facilities or from normal ship operations are not considered disposal at sea, but are subject to controls under other Acts.

8.2.2 How is CEPA 1999 Used to Manage Disposal at Sea?

CEPA 1999 prohibits the disposal of wastes and other matter at sea within Canadian jurisdiction and by Canadian ships in international waters and waters under foreign jurisdiction, unless the disposal is done under a permit issued by the Minister. Permits typically govern timing, handling, storing, loading, placement at the disposal site and monitoring requirements. Permits are published in the *Canada Gazette*, Part I and on the CEPA Environmental Registry website. The permit system allows Canada to meet international obligations under the *London Convention, 1972* and the *1996 Protocol to the Convention*.

Only those substances listed in Schedule 5 of CEPA 1999 may be considered for disposal at sea. These include dredged material, fisheries waste, ships, inert geological matter, uncontaminated organic matter and bulky substances that are primarily composed of iron, steel, concrete or other similar matter. Incineration at sea is banned except under emergency situations or if it is waste generated on board the ship or structure.

Permits are granted on a case-by-case basis after an application and review process. Applicants for a disposal at sea permit must provide detailed disposal data, proof that the applicant published a notice of intent in a local newspaper, any required samples and analyses and payment of fees. Applicants must also comply with the Assessment of Waste or Other Matter in Schedule 6 of CEPA 1999, which requires consideration of other disposal options, such as recycling and means to prevent or reduce the generation of waste. A permit for disposal at sea will be approved only if it is the environmentally preferable and practical option. Permits are not granted if practical opportunities are available to recycle or reuse the material.

Once a permit is issued, Environment Canada conducts periodic inspections during disposal operations to ensure compliance with the permit's conditions. After disposal operations are completed, monitoring studies are conducted at selected sites to verify that permit conditions were met and that scientific assumptions made during the permit review process were correct and sufficient to protect the environment. Results of the monitoring studies are considered in future permit assessments.

9 Vehicles, Engines and Fuels

9.1 What are Emissions from Transportation?

Transportation is the largest source of air pollution in Canada. The use of internal combustion engines to power vehicles and equipment results in a number of smog-causing pollutants, including nitrogen oxides, volatile organic compounds, particulate matter and carbon monoxide. Fuels that are burned in cars, trucks and in stationary equipment also contain sulphur. These substances are directly related to major adverse impacts on the environment and health of Canadians.

The use of internal combustion engines for off-road vehicles, lawn and garden equipment and other machines similarly causes air pollution. The contribution to air pollution from these sources has become more prominent as road vehicles meet ever-tighter emission standards.

Pollutant emissions can be effectively controlled through improvements to fuel quality and through stringent vehicle and engine emission standards. With authorities to control both fuel and vehicle emissions in CEPA 1999, there are better opportunities to ensure that a system approach is taken.

9.2 How is CEPA 1999 Used to Manage Fuels?

CEPA 1999 includes provisions to control the quality of fuels. It provides for maximums, minimums or a range of characteristics to be set, and also allows for a performance-based approach to fuel standards.

Other provisions in CEPA 1999 permit flexibility in the authority to make regulations covering, for example, different sources of fuels, the place or time of their use and the fuel's effect on the operation of emissions control equipment. There are also provisions for a "national fuels mark," a trademark that could be used to promote a national standard for fuels where certain characteristics may be desirable.

9.3 How is CEPA 1999 Used to Manage Emissions from Vehicles, Engines and Equipment?

CEPA 1999 incorporates responsibility for regulating emissions from on-road vehicles that were previously contained in the *Motor Vehicle Safety Act* and its regulations and administered by Transport Canada. In addition, CEPA 1999 allows for regulating emissions from engines used in off-road applications. Examples include spark-ignition (gasoline) engines used in lawnmowers, chainsaws, light industrial machines, outboard motors and off-road recreational vehicles as well as compression ignition (diesel) engines used in construction, industrial, farm and forestry machines. The authority for regulating emissions from engines used to power large marine vessels, aircraft and trains are covered under separate federal legislation administered by Transport Canada.

The main objective is to reduce the contribution of on-road and off-road vehicles and engines to air pollution in Canada through the development and implementation of regulated emission performance standards for vehicles, engines and equipment manufactured in Canada and imported into Canada. The Act provides for the adoption of emission regulations from other countries, including those in the United

States, which have the most progressive emission standards for vehicles and engines. This approach provides for harmonized products in North America and combined environmental and economic benefits.

CEPA 1999 also provides for a "national emissions mark," which can be used to show that vehicles, engines and equipment meet emissions standards. Companies are not permitted to import into or to transport within Canada or sell any prescribed vehicles, engines or equipment that do not have a national emissions mark or do not meet prescribed requirements.

10 Hazardous Wastes

10.1 What is Hazardous Waste?

Hazardous waste includes a wide range of residues from industrial production including used solvents, acids and bases, leftovers from oil refining and the manufacture of chemicals and metal processing. Several common consumer products, including old car batteries and oil-based paints are also hazardous once they are discarded. The nature and concentration of certain chemicals in many wastes makes them potentially hazardous to the environment and human health. They have characteristics such as flammability, toxicity and corrosivity. They may represent an immediate danger, such as ability to burn skin on contact, or longer-term environmental or human health risks due to accumulation and persistence of toxic substances in the environment.

Every year, approximately six million tonnes of hazardous waste are produced in Canada. Imports of hazardous waste total about 417 000 tonnes, of which approximately 55% is destined for recycling. Exports of hazardous wastes total about 320 000 tonnes, of which approximately 65% destined for recycling. Until ways can be found to avoid creating hazardous waste, it must be managed in a way that minimizes risks to the environment and human health.

10.2 How is CEPA 1999 Used to Manage Hazardous Waste?

Under CEPA 1999, transboundary movements (imports, exports, or transits across provincial or territorial borders) of hazardous wastes or hazardous recyclable materials cannot take place unless the Minister is notified and a permit is issued. The prior informed consent of the countries of transit and destination are required. Shipments are also tracked from point of origin to destination. Notification information is made public in the *Canada Gazette*, Part I and on the CEPA Environmental Registry website. These provisions allow for the implementation of Canada's obligations under three international agreements:

- *the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal;*
- the Organisation for Economic Cooperation and Development's *Decision of the Council concerning the Control of Transboundary Movements of Wastes Destined for Recovery Operations C(2001)107/Final;* and
- the *Canada-United States Agreement on the Transboundary Movement of Hazardous Waste*.

CEPA 1999 provides additional authority to:

- define hazardous waste and hazardous recyclable material, which will enable progress towards a federal-provincial-territorial harmonized approach to the management of hazardous waste and hazardous recyclable material;
- regulate exports and imports of prescribed non-hazardous wastes destined for final disposal (e.g., municipal solid wastes);

- develop environmentally sound management criteria to consider prior to refusing to issue an export, import and transit permit, to form an opinion as to whether the hazardous waste or hazardous recyclable material will be managed in a manner that will protect the environment and human health;

- require exporters of hazardous wastes to develop and implement reduction plans for exports of waste destined for final disposal; and

- control interprovincial movements of hazardous wastes and hazardous recyclable materials.

The Minister may also issue a Permit of Equivalent Level of Environmental Safety for export, import and transit of hazardous wastes, hazardous recyclable material or prescribed non-hazardous waste being sent for final disposal, or for interprovincial movement of hazardous waste and hazardous recyclable material. By these permits, the Minister can vary or set aside provisions of regulations governing these activities if satisfied that the level of environmental safety under the permit will be equivalent to what would have been achieved under the regulations.

11 Other Sources of Pollution and Wastes

11.1 International Air and Water Pollution

The international air and water pollution provisions allow the Minister to address Canadian sources that pollute or may pollute the air or water in another country or where that pollution violates an international agreement binding on Canada. This section addresses any type of release of substances that contributes to international air or water pollution, not just those that may have been determined to be toxic. Before using the powers in this division, the Minister must first consult with the provincial, territorial or aboriginal government responsible for the area in which the pollution source is located. This consultation will determine if that government is willing or able to address the problem. If that government is not willing or able to take action, the Minister must take action to reduce or prevent the pollution including:

- requiring pollution prevention planning;
- recommending the making of regulations; or
- issuing an interim order for emergency situations.

11.2 Nutrients

Nutrients, as defined in CEPA 1999, are substances that promote the growth of aquatic vegetation. CEPA 1999 provides authority to regulate nutrients that degrade or have a negative impact on an aquatic ecosystem, such as nutrients contained in cleaning products and water conditioners. CEPA 1999 prohibits the manufacture for use, sale or import of a cleaning product or water conditioner that contains a prescribed nutrient in a concentration or quantity that exceeds the regulated limit. For example, the level of phosphates in laundry detergent is currently regulated under CEPA 1999. CEPA 1999, however, cannot be used to regulate sources of nutrients already regulated under other federal Acts that provide sufficient protection of the environment.

12 Environmental Emergencies

12.1 What is an Environmental Emergency?

An environmental emergency, as defined in CEPA 1999, is an uncontrolled, unplanned or accidental release of a substance (listed in regulations made under Part 8) into the environment or the reasonable likelihood of such a release that may affect the environment or human health. There are an estimated 20 000 environmental emergencies annually in Canada. The majority of the releases are minor and have minimal adverse impact on the environment. About 9 000 emergencies get reported to Environment Canada in any given year and about 1 000 of these require some form of involvement or action by Environment Canada. These incidents are primarily the result of accidents, improper maintenance or human error.

12.2 How is CEPA 1999 Used to Manage Environmental Emergencies?

When it comes to environmental emergencies, no one organization can do it all. Effective emergency response requires teamwork between governments, industry, communities and local organizations. CEPA 1999's environmental emergency provisions provide a "safety net" for the comprehensive management of environmental emergencies. Where no other federal or provincial regulations exist that adequately respond to environmental emergencies, CEPA 1999 can be used to fill these gaps to protect the environment and human health.

CEPA 1999 authorizes the government to make regulations and take non-regulatory measures to prevent, prepare for, respond to and recover from environmental emergencies. The preparation of environmental emergency plans can be required for substances that have been assessed to be toxic under CEPA 1999 and are on the List of Toxic Substances or are recommended for addition to that List. The Government also has the authority to make regulations to require that environmental emergency plans be developed for any substances prescribed in the regulations. These need not be limited to those assessed as toxic under CEPA 1999—they can be substances that are or may be hazardous to the environment or human health in an environmental emergency. *Environmental Emergency Regulations*, listing over 170 substances and requiring the preparation and implementation of environmental emergency plans for those substances, were made under CEPA 1999 in 2003.

These provisions of the Act also establish a regime that makes the person who owns or controls the substance liable for restoring the damaged environment and the costs and expenses incurred in responding to an environmental emergency.

The Minister has the authority to conduct research, conduct and publicize demonstration projects and issue guidelines and codes of practice respecting environmental emergencies. Research could include studies and public demonstrations on the causes of environmental emergencies and remedial measures for dealing with them. The Minister may also establish a national system for the notification and reporting of environmental emergencies.

Under the enforcement provisions, the court can require anyone who has been convicted of a violation under CEPA 1999 to prepare and implement an environmental emergency plan.

13 Government Operations and Federal and Aboriginal Lands

13.1 What are Government Operations, Federal Facilities and Aboriginal Lands?

CEPA 1999 applies to activities on Government of Canada lands. This includes federal departments, boards and agencies, federal works and undertakings, Crown corporations, federal land, persons on that land and other persons in so far as their activities involve that land. This part of the Act also applies to Aboriginal lands.

13.2 Why is there a Special Part of CEPA 1999 for Government Operations and Federal and Aboriginal Lands?

CEPA 1999 applies to all Canadian persons, whether individuals or companies, including federal operations. This means that regulations for toxic substances, fuels, disposal at sea and other matters apply equally to federal operations. Compliance is monitored and CEPA 1999 violations by federal operations are dealt with in the same manner as any other violations, including court action such as injunction and prosecution.

However, under Canada's Constitution, provincial environmental laws do not generally apply to federal lands. This means that federal operations and land, including aboriginal land, are, for the most part, not subject to provincial regulations or permit systems covering emissions, effluents, environmental emergencies, waste handling and other environmental matters. The non-application of these environmental protection laws creates the so-called “environmental protection regulatory gap” with respect to federal departments, boards, agencies, Crown corporations, federal works and undertakings on federal and aboriginal lands.

Under CEPA 1999, Environment Canada can establish regulatory and non-regulatory instruments to manage many, but not all, of the environmental protection risks on federal and aboriginal lands that would otherwise be addressed by provincial and territorial legislation.

13.3 How is CEPA 1999 Used to Manage Government Operations and Federal and Aboriginal Lands?

CEPA 1999 provides the government with broad powers to issue a range of nationally applied regulatory and non-regulatory tools specifically for activities carried out on federal and aboriginal lands. These regulatory and non-regulatory tools include the use of regulations, pollution prevention planning and the creation of codes of practice and guidelines for operations where non-regulatory measures would effectively protect the environment and human health. When created, these tools apply throughout Canada. This means that federal entities, federal land or aboriginal land situated in one province have the same standards as federal entities, federal land or aboriginal land situated in another province. In certain circumstances, CEPA 1999 may also be used to develop tools that would apply only to federal entities,

federal land or only to aboriginal lands, but the standards would have to be the same across the country, even though the corresponding provincial requirements may vary across the country.

14 Enforcement

14.1 What are the Principles of Enforcement?

CEPA 1999 provides the authority to carry out inspections and investigations to ensure that regulations made under the Act and the Act itself are followed. Enforcement of CEPA 1999 follows *the Compliance and Enforcement Policy* established for the Act, which includes the following guiding principles:

- compliance with CEPA 1999 and its regulations is mandatory;
- CEPA 1999 enforcement officers will:
    - apply the Act in a manner that is fair, predictable and consistent;
    - use rules, sanctions and processes securely founded in law;
    - administer the Act with an emphasis on prevention of damage to the environment;
    - examine every suspected violation of which they have knowledge, and take action consistent with the *Compliance and Enforcement Policy*; and
    - encourage Canadians to report CEPA 1999 violations to them.

14.2 What Powers do Enforcement Officers Have?

CEPA 1999 provides the authority to designate persons or classes of persons as enforcement officers. Enforcement officers have a wide range of powers to enforce the Act, including all the powers of a peace officer. Powers include the right to:

- enter premises;
- examine any substance, product, fuel, cleaning product or water conditioner;
- open and examine the contents of any receptacle or package;
- examine any books, records, electronic data or other documents;
- take samples;
- seize evidence;
- conduct tests or take measurements;
- stop and detain conveyances such as a vehicle, ship or aircraft for the purpose of conducting an inspection; and

- use enforcement tools (see Section 14.4).

14.3 What Powers do CEPA Analysts Have?

CEPA 1999 also allows for the designation of individuals as CEPA analysts, who will support the enforcement function. CEPA analysts can be chemists, biologists, engineers, forensic accountants or laboratory personnel. They are entitled to accompany enforcement officers on inspections and they have the power to enter premises, open receptacles, take samples, conduct tests and measurements, and require that documents and data be provided to them. These powers can only be exercised when accompanied by an enforcement officer. Analysts do not have the power to use enforcement tools.

14.4 What are CEPA 1999's Enforcement Tools?

CEPA 1999 enforcement officers have the following enforcement tools at their disposal:

- warnings, when there is minimal or no threat to the environment or human life or health, to indicate the existence of an alleged violation, so that the alleged violator can take notice and return to compliance;
- directions to deal with or prevent illegal releases of regulated substances;
- tickets for offences under the Act where there is minimal or no threat to the environment or human life or health, such as the failure to submit a written report;
- Ministerial orders requiring remedial measures;
- detention orders for ships;
- environmental protection compliance orders to prevent or stop a violation;
- injunctions to stop or prevent a violation;
- prosecution under the authority of a Crown prosecutor; and
- environmental protection alternative measures, as an alternative to prosecution, to come to agreement on measures that the accused must take in order to restore compliance.

14.5 What are CEPA 1999's Penalties for Violations?

The maximum penalties include fines of up to $1 million a day for each day an offence continues, imprisonment for up to three years or both. The Act includes mandatory sentencing criteria for consideration by the courts, such as the cost to remedy the damage done to the environment. Violators may also have to pay for clean-up costs or forfeit any profits earned as a result of an offence. Corporate officials can be prosecuted if they authorize, accept or participate in any violation of CEPA 1999 or its regulations.

15 Research and Monitoring

Science is the foundation of decision-making under CEPA 1999. The Act requires that the Minister of the Environment conduct research on the effects of pollution on environmental quality, the nature and dispersion of pollution on ecosystems, pollution prevention and the control and abatement of pollution. CEPA 1999 also requires both the Minister of the Environment and the Minister of Health to conduct research and studies specifically on hormone disrupting substances and measures to prevent or control the risks associated with these substances. In addition, the Minister of Health must conduct research on the role of substances in illnesses or health problems.

Scientific research also supports the assessment of substances and whether and how to control such substances. Environment Canada and Health Canada participate in a multitude of cooperative projects with universities and research agencies in Canada and around the world to conduct research related to environmental sciences.

Examples of CEPA 1999-related research include:

- field work and sampling programs to collect environmental information;
- laboratory analysis and the development of sampling and analytical techniques to measure environmental parameters, including protocols referenced in regulations and other pollution control instruments;
- research, modeling and monitoring activities to better understand and predict environmental impacts;
- research on the development of techniques for the categorization and assessment of priority substances;
- research and risk assessments to better understand new environmental issues, their impacts on the environment and to fill scientific data gaps;
- research and studies related to pollution prevention and the abatement of pollution; and
- technology development, demonstration, evaluation and research into new potential technologies to address environmental problems.

Additionally, CEPA 1999 requires the government to maintain a system for monitoring environmental quality in Canada, maintain environmental quality data and monitor ocean disposal sites.

16 Information Gathering and Reporting

The authority to gather information allows for environmental monitoring, research, state of the environment reporting, creating inventories and for the development of objectives, guidelines, codes of practice and regulations. Information gathering authorities under Part 3 of the Act are limited to what is in the possession of any person or is reasonably accessible to that person. As required under the Act, the Minister has issued guidelines respecting the use of these information-gathering powers.

Additional information gathering authorities are included under Part 5 of the Act in order to allow the Minister to assess whether or not a substance is toxic or capable of becoming toxic or for assessing whether to control or how to control a substance. In addition, the Minister can require toxicological and other tests if the Ministers of Environment and of Health have reason to suspect that a substance is toxic or capable of becoming toxic.

Both departments are required to distribute information to the public. Publishing information promotes public participation and gives Canadians access to environmental information that relates to their communities. CEPA 1999 requires the Minister of the Environment to distribute information on pollution prevention and periodic reports on the state of the environment. The Minister provides annual reports to Parliament on the administration and enforcement of the Act. The Minister of Health distributes available information about the effects of substances on human health.

The Minister of the Environment must maintain the CEPA Environmental Registry. The Registry is a comprehensive on-line source of CEPA-related documents including policies, guidelines, regulations, orders, agreements, notices and permits.

CEPA 1999 also requires that the Minister maintain and publish a National Pollutant Release Inventory. This inventory (searchable by postal code or substance) provides Canadians with facility-specific information regarding on-site releases and off-site transfers of over 300 substances listed on the inventory. Companies that manufacture, process or otherwise use a listed substance at or above the reporting threshold must report their releases or transfers to Environment Canada annually.

17 Public Participation

17.1 What are the Opportunities for Public Input in Decision-Making?

The role of the public in government decision-making processes is critical, as public trust and broad acceptance of risk management measures are acknowledged to be key for effective risk management implementation.

CEPA 1999 provides a structured, predictable approach to risk management decision-making that provides for the input and full consideration of public values and concerns at all stages of the decision-making process. The CEPA 1999 decision-making framework:

- enables the government to be informed on an ongoing basis of the public's concerns;
- allows the public to influence the identification of environmental problems to be assessed;
- engages a wide spectrum of stakeholders, including environmental groups, industries, aboriginal people, other governments and communities;
- provides an opportunity for public values to influence environmental objectives and solutions; and
- allows the public to articulate the levels of risk that are tolerable or acceptable, which influences the choice of appropriate risk management instruments.

Industry and individuals are continually invited to participate in a wide variety of public consultations through notices published in Canada's official parliamentary journal, the *Canada Gazette*. All consultations are also posted on the CEPA Environmental Registry website. The primary objective of the Environmental Registry is to communicate various types of initiatives under CEPA 1999 to better allow for public participation in the consultation process and to increase public understanding of the Act. The "Public Participation" section of the CEPA Environmental Registry website highlights all consultation opportunities and provides the background information needed for informed environmental decision-making. The Environmental Registry enables the public to monitor the progress of proposed regulations and other CEPA 1999 instruments.

17.2 What Rights do Citizens Have?

Part 2 of CEPA 1999 includes whistleblower protection that safeguards an individual's identity when reporting violations under this Act. This protection is extended to all employees in Canada. CEPA 1999 prohibits the disclosure of the identity of individuals who voluntarily report CEPA 1999 violations. In addition, it is an offence to dismiss, harass or discipline any employee who:

- voluntarily reports a CEPA 1999 violation,
- refuses to carry out conduct that the employee, in good faith, believes may result in a violation of the Act; or

- wishes to carry out conduct required by the Act or its regulations.

Under CEPA 1999, an individual who is at least 18 years of age and a resident of Canada can request that the Minister conduct an investigation of an alleged offence. Should the Minister fail to conduct an investigation or respond unreasonably and if there has been significant harm to the environment, then the individual has the right to proceed with an "Environmental Protection Action." This is a civil suit and seeks remediation of damage to the environment. The individual is not entitled to any personal damage award under the CEPA 1999 provisions, but can seek reimbursement of their costs in bringing the action.

18 Administrative Requirements

18.1 What is the National Advisory Committee?

CEPA 1999 requires the Minister to establish a National Advisory Committee composed of one representative for each of the federal Ministers of the Environment and Health, representatives from each province and territory and six representatives of aboriginal governments drawn from across Canada. An aboriginal government means a governing body that, through an agreement with the Government of Canada, is authorized to enact laws respecting the protection of the environment or the registration of vehicles or engines.

The Committee advises the Ministers on actions taken under the Act, which enables national, cooperative action and avoids duplication in regulatory activity among governments. The Committee also serves as the single window into provincial and territorial governments and representatives of aboriginal governments on offers to consult.

The duties of the NAC include advising the federal Ministers of the Environment and Health on:

- proposed regulations for toxic substances;
- proposed regulations on environmental emergencies;
- a cooperative, coordinated approach to the management of toxic substances; and
- any other matter of mutual interest.

18.2 What are Administrative and Equivalency Agreements?

CEPA 1999 includes provisions that allow the federal government to enter into administrative agreements with provincial and territorial governments, aboriginal governments as well as an aboriginal people (e.g., Band Councils). The Act allows the federal government to sign equivalency agreements with provincial, territorial and aboriginal governments.

Administrative agreements are work-sharing arrangements that can cover any matter related to the administration of the Act. Such matters can include inspections, investigations, information gathering, monitoring and reporting of collected data. These agreements do not release the federal government from any of its responsibilities under the law, nor do they delegate legislative power from one government to another.

The Act allows the use of equivalency agreements where, by Cabinet decision, a regulation under CEPA 1999 is declared to no longer apply in a province, a territory or an area under the jurisdiction of an aboriginal government that has equivalent requirements. The equivalent regulation does not have to have the same wording as the CEPA 1999 regulation, but have the same effect. The provincial, territorial or aboriginal government must also have a mechanism that allows individuals to request an investigation of alleged violations. Equivalency agreements are possible for CEPA 1999 regulations dealing with toxic substances, international air or international water pollution, environmental emergencies and, for

aboriginal governments only, regulations relating to aboriginal land or environmental protection generally and made under Part 9.

CEPA 1999 requires that all proposed equivalency and administrative agreements undergo a 60-day public comment period. Agreements terminate five years after coming into force to ensure regular review and renewal as necessary. Agreements may be terminated at any time with three months notice.

18.3 What is a Board of Review?

CEPA 1999 sets out procedures for establishing and conducting Boards of Review in response to notices of objection filed by members of the public. These provisions are an important component of the Act's enhanced provisions for public participation.

Any person may file a notice of objection to a decision, an order or a proposed regulation and request that Board of Review be established. The Ministers can establish a Board of Review to inquire into the nature and extent of the danger posed by the substance that is the subject of the order or the proposed regulation. In addition, the Minister may establish a Board of Review for other instruments (e.g., administrative or equivalency agreements) when request for such a Board is filed during the 60-day public comment following publication of the instrument in the *Canada Gazette*.

18.4 When is the Act Reviewed?

Every five years, a committee of one or both Houses of Parliament must review the Act, as required under CEPA 1999. The committee conducts a comprehensive review of the provisions and operations of the Act and makes recommendations regarding any changes to the Act or its administration. The review can therefore monitor the effectiveness of the legislation in protecting the environment and human health and preventing pollution.

Appendix A: Contacts

Further information on CEPA 1999 and related activities can be found online at:

- CEPA Environmental Registry Website (http://www.ec.gc.ca/CEPARegistry)
- Environment Canada's Green Lane ™ (http://www.ec.gc.ca); and
- Health Canada's Website (http://www.hc-sc.gc.ca).

Departmental publications are available from the departmental library or the nearest regional library. Many departmental publications are also available online at http://www.ec.gc.ca/publications or through Environment Canada's Inquiry Centre:

Inquiry Centre:
70 Crémazie St.
Gatineau, Quebec
K1A 0H3
Telephone: 819-997-2800 or 1-800-668-6767
Fax: 819-994-1412
TTY : 819-994-0736 (Teletype for the hearing impaired)
E-mail: enviroinfo@ec.gc.ca

The following communications contacts are also available to provide additional information:

Environment Canada
Mark Colpitts
Ottawa, Ontario
Canada, K1A 0K9
Telephone: (819) 953-6603
Fax: (819) 953-8125
E-mail: Mark.Colpitts@ec.gc.ca

Health Canada
A.L. 0900C2
Ottawa, Ontario
Canada, K1A 0K9
Telephone: (613) 957-2991
Fax: (613) 941-5366
TTY: 1-800-267-1245
E-mail: info@hc-sc.gc.ca

Appendix B: CEPA 1999 Program Websites

Disposal at Sea	http://www.ec.gc.ca/seadisposal
Enforcement	http://www.ec.gc.ca/ele-ale
Environmental Emergencies	http://www.ec.gc.ca/ee-ue
Existing Substances	http://www.ec.gc.ca/substances/ese
Hazardous Wastes	http://www.ec.gc.ca/tmb
Management of Toxic Substances	http://www.ec.gc.ca/toxics
National Office of Pollution Prevention	http://www.ec.gc.ca/nopp
New Substances	http://www.ec.gc.ca/substances/nsb/eng/index_e.htm
Science and Technology	http://www.ec.gc.ca/scitech

Appendix C: CEPA 1999 Provisions

Topic	Part, Section of CEPA 1999
Administration	Part 1
Administrative and Equivalency Agreements	Part 1, Sections 9 and 10
Animate Products of Biotechnology	Part 6
Boards of Review	Part 11, Sections 333 to 341
Citizens' Rights	Part 2
Disposal at Sea	Part 7, Division 3
Enforcement	Part 10
Environmental Emergencies	Part 8
Existing Substances	Part 5, Sections 64 to 79 and 90 to 99
Export of Substances	Part 5, Sections 100 to 103
Fuels	Part 7, Division 4
Government Operations and Federal and Aboriginal Lands	Part 9
Hazardous Wastes	Part 7, Division 8
Information Gathering and Reporting	Parts 3, 5 and 11
International Air and Water Pollution	Part 7, Division 6 and 7
Marine Environment (Land-based Sources of Pollution)	Part 7, Division 2
National Advisory Committee	Part 1, Section 6
New Substances	Part 5, Sections 80 to 89
Nutrients	Part 7, Division 1
Public Participation	Part 2
Research and Monitoring	Part 3
Review of the Act	Part 11, Section 343
Vehicles, Engines and Equipment Emissions	Part 7, Division 5

Risk assessments under section 90(1) of Canadian Environmental Protection Act, 1999

Substances may also be added to the List of Toxic Substances in Schedule 1 of the *Canadian Environmental Protection Act, 1999* (CEPA) through section 90(1) if, on the recommendation of the Ministers of Environment and Health, the Governor in Council is satisfied that a substance is toxic. A substance is "CEPA-toxic equivalent" if it satisfies the definition of "CEPA-toxic" as a result of a systematic, risk-based assessment. Such assessments can include determinations made under other federal statutes, or can incorporate appropriate elements of assessments done by or for provinces or territories, international organizations or other appropriate scientific authorities such as:

a. Stockholm Convention
b. Montreal Protocol

Risk assessment

Substance	Justification
Bromochloromethane	Montreal Protocol
Dichlorodiphenyltrichloroethane (DDT)	Stockholm Convention

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Government of Canada Gouvernement du Canada

A FRAMEWORK FOR THE APPLICATION OF PRECAUTION IN SCIENCE-BASED DECISION MAKING ABOUT RISK

Canada

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Table of Contents

1.0 Introduction

This Framework outlines guiding principles for the application of precaution to science-based decision making in areas of federal regulatory activity for the protection of health and safety and the environment and the conservation of natural resources.

What is the application of precaution?

The application of "precaution", "the precautionary principle" or "the precautionary approach"[1] recognizes that the absence of full scientific certainty shall not be used as a reason for postponing decisions where there is a risk of serious or irreversible harm.

The application of precaution is distinctive within science-based risk management and is characterized by three basic tenets: the need for a decision, a risk of serious or irreversible harm and a lack of full scientific certainty.

Canada has a long-standing history of applying precaution in areas of federal regulatory activities. The Government's obligations in this regard are governed by applicable provisions of federal law, binding federal-provincial agreements and international agreements to which Canada is a party.

Are guidance and assurance needed?

Given the distinctive circumstances associated with the application of precaution, notably the lack of full scientific certainty about a risk of serious or irreversible harm, guidance and assurance are required as to the conditions governing decision making. Guidance and assurance are particularly needed in circumstances when the scientific uncertainty is high.

What is the purpose of the framework?

This Framework serves to strengthen and describe existing Canadian practice. The purpose of the framework is to:

- improve the predictability, credibility and consistency of the federal government's application of precaution to ensure adequate, reasonable and cost-effective decisions;[1]

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1 This document uses these expressions interchangeably. It focuses on the guiding principles of precautionary decision making rather than discussing distinctions that may be drawn between different expressions of precaution.

- support sound federal government decision making while minimizing crises and controversies and capitalizing on opportunities;
- increase public and stakeholder confidence, in Canada and abroad, that federal precautionary decision making is rigorous, sound and credible; and
- increase Canada's ability to positively influence international standards and the application of precaution.

Ultimately, the Framework provides a lens to assess whether precautionary decision making is in keeping with Canadians' social, environmental and economic values and priorities. It complements the Government's *Integrated Risk Management Framework* and *A Framework for Science and Technology Advice: Principles and Guidelines for the Effective Use of Science and Technology Advice in Government Decision making.*

2.0 Context

Canada has a long-standing history of applying precaution in science-based regulatory programs. Technology, globalization and the knowledge-based economy are driving tremendous changes in both the private and public sector. Risk, inherent in the activities of individuals and business, contributes to even greater uncertainty. When combined with high-profile, risk-based events, these changes highlight the need for more effective strategies to manage risk and seize the opportunities that change presents.

Governments can rarely act on the basis of full scientific certainty and cannot guarantee zero risk. Indeed, they are traditionally called upon and continue to address new or emerging risks and potential opportunities, and to manage issues where there is significant scientific uncertainty. However, the need for decision making in the face of scientific uncertainty has grown both in scope and public visibility and this has led to a growing awareness of and emphasis on the application of precaution to decision making.

While the application of precaution primarily affects the development of options and the decision phases within science-based risk management, it is clearly linked to scientific analysis (it cannot be applied without an appropriate assessment of scientific factors and consequent risks). Ultimately, it is guided by judgment, based on values and priorities but its application is complicated by the inherent dynamics of science — even though scientific information may be inconclusive, decisions will still have to be made as society expects risks to be addressed and managed and living standards enhanced.

Canada's application of precaution is flexible and responsive to particular circumstances. Moreover, rules-based approaches are employed to achieve the results required by specific legislation or international obligations (e.g., fisheries management).

3.0 Science and uncertainty in decision making

As the scientific process is often characterized by uncertainty and debate, the decision-making process for managing risks associated with scientific information requires sound judgment. The application of precaution to decision making is distinctive within traditional risk management on the basis of a higher degree of scientific uncertainty and the parameters that can establish what constitutes an adequate scientific basis and sound and rigorous judgment. As it applies here, judgment focuses on addressing:

- what is a sufficiently sound or credible scientific basis?
- what follow-up activities may be warranted?
- who should produce a credible scientific basis? and
- the inherent dynamics of science on decision making.

What is a sufficiently sound or credible scientific basis?

In traditional situations of decision making to manage risks, "sound scientific evidence" is generally interpreted as either definitive and compelling evidence that supports a scientific theory or significant empirical information that clearly establishes the seriousness of a risk.

Within the context of precaution, determining what constitutes a sufficiently sound or credible scientific basis is often challenging and can be controversial. The emphasis should be on providing a sound and credible case that a risk of serious or irreversible harm exists. "Sufficiently sound" or credible scientific basis should be interpreted as a body of scientific information — whether empirical or theoretical — that can establish reasonable evidence of a theory's validity, including its uncertainties and that indicates the potential for such a risk.

What follow-up activities may be warranted?

Given the significant scientific uncertainty implicit in the application of precaution, follow-up activities such as research and scientific monitoring are usually a key part of the application of precaution. In some cases, international agreements (e.g., World Trade Organization Agreement on the Application of Sanitary and Phytosanitary Measures) require scientific monitoring and follow-up when precaution is applied. Such efforts can help reduce the scientific uncertainty associated with certain risks and allow informed follow-up decisions to be made. In other circumstances, scientific uncertainty may take a long time to resolve or, for practical purposes, never be resolved to any significant degree.

In order to capture the full diversity of scientific thought and opinion, the basis for decision making should be drawn from a variety of scientific sources and experts from many disciplines. Decision makers should give particular weight, however, to peer-reviewed science and reasonableness in their judgments. Moreover, the science function can be further supplemented by formal, structured and, where warranted, independent advisory processes that include widely recognized and credible individuals.

Who should produce a credible scientific basis?

Establishing who should be responsible for producing a credible scientific basis raises a different question: Who should be designated as having the responsibility to produce the scientific data and provide the basis for decision making? Decision makers should assess such criteria as who holds the legal responsibility or authority (e.g., the proponent who is designated as the legal agent in Canada), who would be in the best position to provide the scientific data and who has the capacity to produce timely and credible information.

While the party who is taking an action associated with potential serious harm is generally designated as the responsible party, this may best be decided on a case-by-case basis. Innovative strategies may also be introduced, such as collaborative arrangements among different levels of government and industry. As the scientific knowledge evolves, this responsibility may shift among governments, industry or another proponent (e.g., health practitioners documenting adverse effects from a product already on the market).

The inherent dynamics of science on decision making

The inherent dynamics of uncertainty in science present unique challenges. Climate change provides a good example. There is international consensus that human activities are increasing the amounts of greenhouse gases in the atmosphere and that these increases are contributing to changes in the earth's climate. However, there is scientific uncertainty regarding the sensitivity of climate to these increases, particularly the timing and regional character of climate change. There is also a degree of uncertainty in the economic costs of potential measures to reduce greenhouse gases, although the modelling suggests that these impacts are manageable, as well as the economic costs, to adapt to the expected changes in climate.

While scientific information is still inconclusive, decisions will have to be made to meet society's expectations about enhancing living standards and addressing the potential for risks. An understanding of the full potential of the products and processes arising from rapidly evolving science and technology is critical to shaping Canada's laws and regulations, as well as international agreements and guidelines. The implications are only now starting to emerge and will ultimately influence decisions.

4.0 Guiding Principles for the application of precaution to science-based decision making

As noted earlier, the application of precaution to science-based decision making to manage risk is driven by specific circumstances and factors and is characterized by three basic tenets: the need for a decision, a risk of serious or irreversible harm and a lack of full scientific certainty.

Guiding principles outlined in this Framework reflect current practices and, in their entirety, are intended to support overall consistency in application, allow for flexibility to respond to specific circumstances and factors and help to counter misuse or abuse. While they focus on those aspects of the process that are distinctive within risk management overall, they could not direct decision makers to act in a way inconsistent with their legal authority. Moreover, this Framework is not meant to create any new legal obligations to apply precaution.

General principles of application outline distinguishing features of precautionary decision making whereas principles for precautionary measures describe specific characteristics that apply once a decision has been taken that measures are warranted.

Five General Principles of Application

4.1 The application of precaution is a legitimate and distinctive decision-making approach within risk management.

- While precaution primarily affects the development of options and the decision phases, it is clearly linked to scientific analysis (it cannot be applied without an appropriate assessment of scientific factors and consequent risks). Ultimately, it is guided by judgment, based on values and priorities.

- The Government's obligations to apply precaution are governed by applicable provisions of federal law, binding federal-provincial agreements and international agreements to which Canada is a party.

- The Government does not yet consider the precautionary principle/approach to be a rule of customary international law.

4.2 It is legitimate that decisions be guided by society's chosen level of protection against risk.

- To the extent possible, the level of protection should be established in advance through domestic policy instruments such as legislation and international agreements.

- While societal values and public willingness to accept risk are key in determining the level of protection, in all cases sound scientific evidence is a fundamental prerequisite to applying the precautionary approach.

- It should be recognized that some risks are new or emerging and evolution of scientific knowledge may influence society's tolerances and its chosen level of protection. In such circumstances, public involvement mechanisms that seek the input of those most affected by decisions should help advance understanding of the level of protection against risk.

4.3 Sound scientific information and its evaluation must be the basis for applying precaution; the scientific information base and responsibility for producing it may shift as knowledge evolves.

- It is particularly relevant that sound scientific information and its evaluation be the basis for (i) the decision to act or not to act (i.e., to implement precautionary measures or not) and (ii) the measures taken once a decision is made.

- In determining what constitutes a sufficiently sound or credible scientific basis, the emphasis should be on providing a sound and credible case that a risk of serious or irreversible harm exists. "Sufficiently sound" or credible scientific basis should be interpreted as a body of scientific information — whether empirical or theoretical — that can establish reasonable evidence of a theory's validity, including its uncertainties and that indicates the potential for such a risk.

- Scientific data relevant to the risk must be evaluated through a sound, credible, transparent and inclusive mechanism leading to a conclusion that expresses the possibility of occurrence of harm and the magnitude of that harm (including the extent of possible damage, persistency, reversibility and delayed effect).

- Available scientific information must be evaluated with emphasis on securing high quality scientific evidence (not quantity). Reports should summarize the existing state of knowledge, provide scientific views on the reliability of the assessment and address remaining uncertainties and areas for further scientific research or monitoring.

- Peer review represents a concrete test for the practical application of precaution to decision making. A peer-review process can assess the soundness of the scientific evidence and its inherent credibility within the scientific community.

- Scientific advice should be drawn from a variety of sources and experts and should reflect the full diversity of scientific interpretations consistent with the evidence available. This does not preclude contributions of traditional knowledge from sources such as Aboriginal peoples or fishing communities; these have a valid role in providing both evidence and its interpretations. Scientific advisors should give weight to peer-reviewed science and aim at sound and reasonable evidence on which to base their judgments.

- In circumstances where there is a potential for imminent harm, it may be appropriate to make decisions and implement precautionary measures in the near term, with an understanding that close monitoring would occur to assess the effectiveness of the measures in addressing risk and overall impacts.

- Follow-up activities, including research and monitoring, are key to reducing scientific uncertainty and allow improved decisions to be made in the future.

- Overall, the responsibility for providing the sound scientific basis should rest with the party who is taking an action associated with a risk of serious harm (e.g., the party engaged in marketing a product, employing a process or extracting natural resources). However, when faced with a concrete scenario, there should be an assessment of who would be in the best position to provide the information base. This could depend upon which party holds the responsibility or authority, and could also be informed by such criteria as who has the capacity to produce timely and credible information.

- The responsibility for providing the sound scientific basis may best be decided on a case-by-case basis and may be collaborative. Moreover, it should be recognized that what constitutes an appropriate scientific base and responsibility for producing it may shift as the knowledge grows and roles of the public and private sectors evolve.

4.4 Mechanisms should exist for re-evaluating the basis for decisions and for providing a transparent process for further consideration.

- It is desirable that those affected by a decision have input into the re-evaluation process.

- The impact (benefits and drawbacks) of re-evaluation and consultative mechanisms in any particular situation should be assessed (i.e., in some cases, they may not be practical or productive). Given some existing re-evaluation and consultative mechanisms (e.g., fishery conservation), it should be recognized that additional mechanisms may not be appropriate.

- A re-evaluation may be triggered by the emergence of new scientific information, new technology or a change in society's tolerance for risk. Effective review of decisions requires monitoring the effectiveness of decisions on an ongoing basis with provision for regular feedback and reporting of performance measurements results.

- The decision-making hierarchy and the duties and responsibilities of participants in the process should be clearly laid out so that accountabilities can be understood, respected and communicated. This would also facilitate requests for additional re-evaluation and consultation.

- The nature, type and frequency of re-evaluation and consultation mechanisms may be related to the specific circumstances of a situation, for example whether precaution is applied within an ongoing mechanism for conservation of resources or in circumstances where there is a potential for imminent harm.

4.5 A high degree of transparency, clear accountability and meaningful public involvement are appropriate.

- An understanding of the "public's tolerance for risks" or "society's chosen level of protection" underpins the need for high transparency, clear accountability and meaningful public involvement.

- Transparency in documenting the rationale for making decisions strengthens accountability.

- Two-way sharing of information and the inclusion of a range of perspectives in the decision-making process can become the cornerstone of openness and transparency for the decision-making process and enhance credibility of and trust in the decisions that the Government makes. The Government's Communications

Policy provides principles for well co-ordinated, effectively managed and responsive communications.

- Public involvement can provide a platform to resolve conflict or engage in joint problem solving by a specific set of rules. It can bring about the recognition of ambiguities and uncertainties, and promote acceptance of different perspectives. Moreover, it can provide impetus for peer review and an opportunity to receive interpretations on uncertainty and risk from the public.

- Public involvement should be structured into the scientific review and advisory process, as well as the decision-making process. At the same time, it should be recognized that the opportunity for public involvement often depends on the specific context and timeliness of the required decision. In situations of significant uncertainty (regarding the magnitude and/or likelihood of harm or the most effective means of addressing the harm, combined with complex science), public involvement is needed to provide an opportunity to receive interpretations on uncertainty and risk.

Five Principles for Precautionary Measures

4.6 Precautionary measures should be subject to reconsideration, on the basis of the evolution of science, technology and society's chosen level of protection.

- Precautionary measures should generally be implemented on a provisional basis; that is, they should be subject to review in light of new scientific information or other relevant considerations, such as society's chosen level of protection against risk.

- Given the limitations of evolving scientific knowledge, decision makers should recognize that scientific uncertainty may not be resolved quickly and, in some cases is intrinsic to the situation (e.g., change is intrinsic to natural resources) — they should review new scientific knowledge if and as it evolves. In certain instances, setting time considerations would be counter-productive.

- Domestic or international obligations may require that some precautionary measures be deemed explicitly provisional and subject to re-evaluation; they may include obligations requiring mechanisms for ongoing monitoring and reporting.

- Regardless of whether there is a formal obligation, follow-up scientific activity (e.g., further research and monitoring) should be promoted, as it can help reduce uncertainty and allow improved decisions as the science evolves.

4.7 Precautionary measures should be proportional to the potential severity of the risk being addressed and to society's chosen level of protection.

- There is an implicit need to identify, where possible, both the level of society's tolerance for risks and potential risk-mitigating measures. This information should be the basis for deciding whether measures are proportional to the severity of the risk being addressed and whether the measures achieve the level of protection, recognizing that this level of protection may evolve.

- While judgments should be based on scientific evidence to the fullest extent, decision makers should also consider other factors such as societal values and willingness to accept risk and economic and international considerations. This would allow for a clearer assessment of the proportionality of the measure and ultimately help maintain credibility in the application of precaution.

- Generally, the assessment of whether measures are considered proportional to the severity of risk should be in relation to the magnitude and nature of the potential harm in a particular circumstance, not in comparison with measures taken in other contexts.

4.8 Precautionary measures should be non-discriminatory and consistent with measures taken in similar circumstances.

- Consistent approaches should be used for determining an appropriate level of protection against risk. Ultimately, the level of protection should be set in the public interest by weighing potential (or perceived) costs and benefits of assuming the risk in a manner that is consistent overall with societal values.

- Similar situations should not be treated substantially differently and decision makers should consider using processes used in comparable situations to ensure consistency. Except where the choice of precautionary measures is predetermined in agreements or legislation, it should be flexible and determined on a case-by-case basis.

- Domestic applications of precaution should be consistent with Canada's obligations arising from international agreements to which it is a party and where applicable, meet the requirements established by the Regulatory Policy.

4.9 Precautionary measures should be cost-effective, with the goal of generating (i) an overall net benefit for society at least cost, and (ii) efficiency in the choice of measures.

- The real and potential impacts of making a precautionary decision (whether to act or not to act), including social, economic and other relevant factors, should be assessed.

- Decision making should identify potential costs and benefits as explicitly and as soon as possible, and distinguish what risk the public is prepared to accept on the basis of sound and reasonable, albeit incomplete, scientific evidence.

- Consideration of risk–risk tradeoffs or comparative assessments of different risks would generally be appropriate, although this may not be possible in circumstances where urgent action is needed. This can ensure that society receives net benefits from decision making and that the application of precaution is inherently responsive to the potential from innovation or technological change and the overall benefits that such change can entail.

- Assessing the efficiency of precautionary measures generally involves comparing various policy instruments to determine which options could most efficiently address the risk at least overall cost. The outcome of this process should result in any measures taken imposing the least cost or other negative impact while reducing risks to an acceptable level.

- As science evolves, it is inherently appropriate that the cost-effectiveness of decisions and associated measures be assessed and taken into account at the start, in the interim and, possibly, over the longer term. For some issues, a net benefit may not be realized for a long period of time, for example, decisions associated with biodiversity. However, the emphasis should always be on ensuring that ongoing costs are assessed and minimized, so that new scientific data that alters cost-effectiveness considerations can be incorporated (including performance monitoring results), while maintaining the reduction of risks and, where appropriate, maximizing the benefits (e.g., from innovation).

- Decision makers should consider broader costs and benefits from decisions to help ensure that society receives net benefits overall (e.g., benefits associated with enhanced health status of children as a segment of the population or benefits from innovation or technological change).

4.10 Where more than one option reasonably meets the above characteristics, then the least trade-restrictive measure should be applied.

- When making a choice among different types of measures that would provide a similar level of response to the potential for harm, there should be an endeavour to select measures that would be "least trade-restrictive".

- Least trade-restrictive considerations should apply to both international and internal trade. This is especially relevant in terms of international trade where disciplines and mechanisms exist for other States to challenge the nature and impact of precautionary measures.

5.0 Conclusion

A Framework for the Application of Precaution in Science-based Decision Making About Risk sets out guiding principles to achieve coherent and cohesive application of precaution to decision making about risks of serious or irreversible harm where there is lack of full scientific certainty, with regard to federal domestic policies, laws and agreements and international agreements and guidelines in areas where science is implicated.

Departmental and agency officials are expected to consider its guiding principles in decision making and to work together in developing, in consultation with their stakeholders, guidance for the application of precaution in their particular area of responsibility.

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Your health and safety… our priority. *Votre santé et votre sécurité… notre priorité.*

Weight of Evidence:
General Principles and Current Applications at Health Canada

PREPARED FOR: Task Force on Scientific Risk Assessment

PREPARED BY: Weight of Evidence Working Group

Canada

Health Canada is the federal department responsible for helping the people of Canada maintain and improve their health. Health Canada is committed to improving the lives of all of Canada's people and to making this country's population among the healthiest in the world as measured by longevity, lifestyle and effective use of the public health care system.

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Document Revision History

VERSION	DATE	SECTION/PARAGRAPH CHANGED	CHANGE(S) MADE
1	November 2011		Initial issuance of final document.
2	May 2018	Document Revision History	Section added to track revisions.
		Annex 2	Program areas, interpretations and applications updated.
		8.0 References	Section 8.0 added; Reference list updated throughout document.

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5 Centre for Evaluation of Radiopharmaceuticals and Biotherapeutics, Biologics and Genetic Therapies Directorate, HPFB, Ottawa, ON

6 Bureau of Microbial Hazards, Food Directorate, HPFB, Ottawa, ON

7 Office of Risk Management and Science, Marketed Health Products Directorate, HPFB, Ottawa, ON

8 Existing Substances Risk Assessment Bureau, Safe Environments Directorate, HECSB, Ottawa, ON

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1. Introduction

Weight of Evidence (WoE) is frequently cited as the basis on which risk assessment conclusions are made. However, multiple interpretations and a lack of consensus about its meaning could potentially compromise communication between diverse stakeholders in the decision-making process. In response to this issue, an analysis of the WoE approach was initiated by Health Canada's Science Policy Directorate in 2010, as a project under the Task Force on Scientific Risk Assessment. By examining current interpretations and identifying potential best practices, this analysis aims to enhance the consistency and coherence of risk assessments across the Department.

2. Purpose and scope

The current document aims to inform senior management about WoE in Health Canada risk assessments by providing an overview of the approach in terms of its:

- role in scientific risk assessments;
- main guiding principles; and
- application by various risk assessment programs at Health Canada.

In addition, this explanatory document serves as a value-added Departmental resource of high level contextual information and guiding principles to supplement program specific guidelines, procedures and/or tools.

While this document acknowledges that WoE could also be applied in the risk management decision making context, where scientific evidence is weighed against other policy considerations, it will not expand on this information as it is considered **not** within the scope of this document.[9]

[9] The terms evidence, information and data are used interchangeably in this document, and refer to general scientific usage, not specific legal definitions of what constitutes evidence, or "admissible" evidence, in a court of law.

3. Role in risk assessments

In general, scientific risk assessments encompass the following steps: identifying and characterizing the hazard, assessing the exposure, and characterizing the risk; risk assessments also play an integrated role in an evidence- informed decision making process which also involves managing and communicating the risk.

WoE in the risk assessment context is defined in *Health Canada Decision-Making Framework for Identifying, Assessing, and Managing Health Risks* (Health Canada, 2000) as:

> *"A qualitative measure that takes into account the nature and quality of scientific studies intended to examine the risk of an agent. Uncertainties that result from the incompleteness and unavailability of scientific data frequently require scientists to make inferences, assumptions, and judgements in order to characterize a risk. Making judgements about risk based on scientific information is called "evaluating the weight of evidence".*

The above description can be interpreted to implicitly include two separate concepts frequently associated with WoE terminology:

1. **Totality of Evidence:** what types and sources of information are to be gathered and considered for subsequent assessment; and
2. **Weighing Evidence:** how such individual sources of evidence are assessed and integrated into an overall conclusion or recommendation.

Totality of evidence can be influenced by varying interpretations of *"all"* available or relevant evidence to date. This concept provides the opportunity to make use of information/studies that may be regarded insufficient individually, but which contribute to a total "weight of evidence" case in support of conclusions during risk assessment when they are considered alongside other studies/sources of evidence. Moreover, an evaluation of evidence and of any subsequent decision can be reassessed, at a later date, based on the availability of data that may not have been readily available at the time of the original assessment.

The latter, methodological concept of weighing evidence is applicable to most risk assessments. While specific methodologies and tools used for assessing and integrating evidence (e.g., quantitative or qualitative) may vary and are context dependent, the general principles for the assessment and integration process remain the same.

European Science and Technology Observatory

ON SCIENCE AND PRECAUTION IN THE MANAGEMENT OF TECHNOLOGICAL RISK

Volume I

A Synthesis Report of case studies

About the IPTS

The Institute for Prospective Technological Studies (IPTS) is one of the eight institutes making up the Joint Research Centre (JRC) of the European Commission. It was established in Seville, Spain, in September 1994.

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Although particular emphasis is placed on key Science and Technology fields, especially those that have a driving role and even the potential to reshape our society, important efforts are devoted to improving the understanding of the complex interactions between technology, economy and society. Indeed, the impact of technology on society and, conversely the way technological development is driven by societal changes, are highly relevant themes within the European decision-making context.

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The European Science and Technology Observatory (ESTO) is a network based on a core group of 17 European leading organisations with expertise on Science and Technology Assessment. ESTO provides real-time information on the socio-economic significance of scientific and technological advances. The ESTO Network is directed and managed by the IPTS.

Along with the 14 initial members, another group of institutes later became associated to the ESTO network covering all the fifteen EU Member States as well as Israel. Membership is being continously reviewed and expanded with a view to match the evolving needs of the IPTS and to incorporate new competent organisations from both inside and outside of the EU.
The ESTO Network was formally constituted in February 1997 and its principal tasks are:

- To contribute to The IPTS Report with articles on relevant topics
- To issue, on a periodic basis, a Techno-Economic Analysis report, which reviews socio-economic developments either arising from technological change or driving it
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- To provide Quick Responses to specific S&T assessment queries.

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On Science and Precaution

In the Management of Technological Risk

An ESTO Project Report

Prepared for the
European Commission - JRC
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Seville

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Prologue

This study co-ordinated by Dr. Stirling shows how greatly our thinking has changed on issues of the management of risks. The use of sophisticated scientific methods in the assessment and then management of risks began with the problems of major industrial hazards, notably those of nuclear power. At first it was believed that quantitative techniques, either of statistics or of modeling, would suffice for the guidance of risk policy and risk management. But as experience accumulated, it became clear that while science is an essential core of the assessment process, it could not be the whole. The supplementary materials have a variety of names, including 'participation' and 'precaution'; and their practical content is still being developed.

Now the hazards we face are more diffuse, and in their own way more threatening. There are concerned publics, capable of acting in a co-ordinated way and directly affecting government policies for the environment and whole industries. We may say that in such issues, facts are uncertain, values in dispute, stakes high and decisions urgent. The traditional peer communities, formerly restricted to qualified experts, are now extended to include citizen participants at many levels.

The management of these new processes present many difficulties. It is to the credit of Dr. Stirling and his colleagues that the problems are analysed to such depth, and that such important and useful practical lessons are drawn. This report can become a valuable contribution to the resolution of an urgent problem.

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November 1999

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different assumptions in analysis and the associated pictures of the relative importance of different options. Since many of the incommensurable dimensions of variability discussed here (and shown in Tables 4 and 6) typically remain implicit in risk assessment, serious questions must be raised over whether the associated results – no matter how precisely expressed – are of any practical policy use at all.

Table 6: A Set of Methodological Variables Influencing Variability and Ambiguity in Risk Assessment (after Stirling)

VARIABLE	ILLUSTRATIVE QUESTION AND PRACTICAL EXAMPLE
Quantifiability	Are the effects associated with different options all equally quantifiable? How has appraisal avoided a disproportionate emphasis on the more quantifiable aspects - and thus an overemphasis of the impacts of the associated options? (*Eg: acid emissions vs aesthetic landscape impacts*).
Coherence	How coherent is the classificatory scheme adopted in any particular study with respect to the full range of environmental effects? Are there gaps or overlaps between the different classes of effect which are recognised for the purposes of analysis? (*Eg: emissions, burdens, or effects*).
Trajectories	How long a historic data series is appropriate as a basis for the appraisal of current options? How robust are assumptions concerning the likely future behaviour of those at risk? Are different options on different 'learning curves' in terms of the potential for future improvements in performance? (*Eg: radioactive waste, photovoltaics*).
System Boundaries	How systematically does analysis address the resource chains and facility life cycles associated with the different options? How far back into the wider economy should analysis regress in assessing energy and material inputs? (*Eg: material and energy inputs to renewable capital equipment, overseas uranium mining for nuclear*).
Articulation	How are the results of analysis to be articulated with wider considerations and the subsequent decision making process. At what point does the domain of analysis end and that of politics begin? (*Eg: claims to 'real', 'true' or 'full' status?*).

The practical conclusions both for 'scientific' and 'precautionary' approaches seem clear. The theoretical impossibility of the 'analytical fix' is borne out in practice. The appraisal of technological risk is evidently as much about systematic qualitative exploration of the consequences of divergent social values as it is about precise numerical characterisations of the physical impacts of the technologies themselves. It is better to be *roughly accurate* in this task of mapping the social and methodological context-dependencies. than it is to be *precisely wrong* in spurious aspirations to a one-dimensional quantitative expression of technological risk.

5: KNOWING YOUR IGNORANCE IS THE BEST PART OF KNOWLEDGE

the limits to probability

Before turning to more positive themes, there is one further fundamental difficulty in the 'narrow risk' approach to the management of technology which has been explored in detail in this project and which should be considered in this Report. It has already been shown how risk is conventionally regarded to comprise the two basic elements of probabilities and magnitudes. The problems of multidimensionality and incommensurability discussed so far reside with the 'magnitude' component of risk. It is with the concept of 'probability', on the other hand, that we invoke the problems of uncertainty and ignorance. These themes are discussed in some detail in the Field Studies by Salo and Stirling.

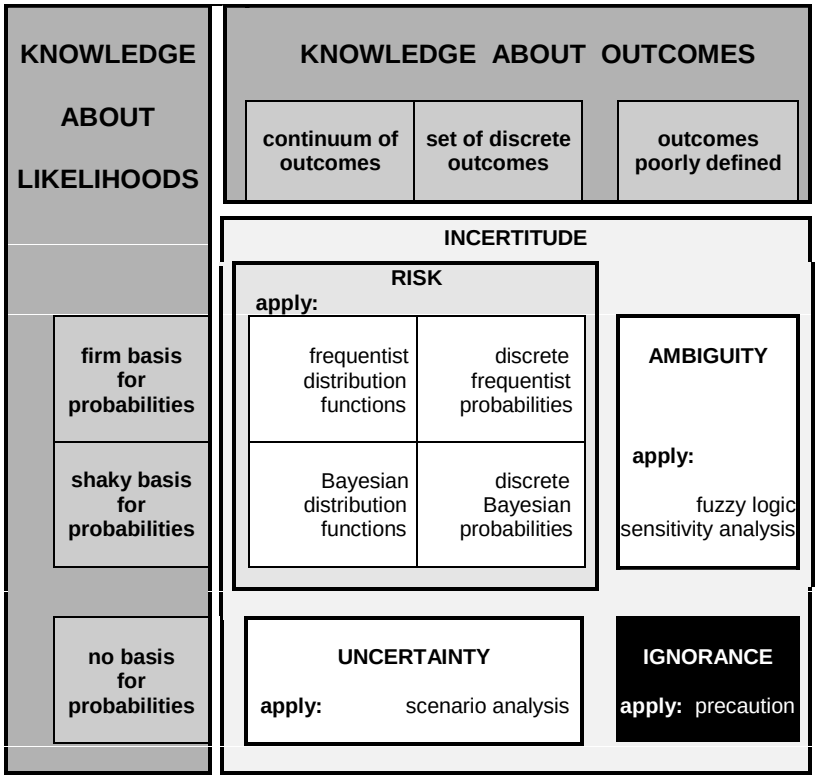
In economics and decision analysis, the well-established formal definition of *risk* is that it is a condition under which it is possible both to define a comprehensive set of all possible outcomes *and* to resolve a discrete set of probabilities (or a density function) across this array of outcomes. This is the domain under which the various probabilistic techniques of risk assessment are applicable, permitting (in theory) the full characterisation and ordering of the different options under appraisal. There are a host of details relating to this picture (such as those hinging on the

distinction between 'frequentist' and 'Bayesian' understandings of probability), which are discussed further in the Stirling Field Study but these are irrelevant for present purposes.

The strict sense of the term *uncertainty*, by contrast, applies to a condition under which there is confidence in the completeness of the defined set of outcomes, but where there is acknowledged to exist no valid theoretical or empirical basis for the assigning of probabilities to these outcomes. Here, the analytical armoury is less well-developed, with the various sorts of scenario analysis being the best that can usually be managed.. Whilst the different options under appraisal may still be broadly characterised, they cannot be ranked even in relative terms without some knowledge of the relative likelihoods of the different outcomes.

Finally, there is the condition of *ignorance*. This applies in circumstances where there not only exists no basis for the assigning of probabilities (as under uncertainty), but where the definition of a complete set of outcomes is also problematic. In short, it is an acknowledgement of the possibility of surprises. Here, it is not only impossible to rank the options but even their full characterisation is difficult. Under a state of ignorance (in this strict sense), it is always possible that there are effects (outcomes) which have been entirely excluded from consideration. These formal definitions for the concepts of risk, uncertainty and ignorance are illustrated schematically in Figure 4.

Figure 4: The Concepts of 'Incertitude', 'Risk', 'Uncertainty' and 'Ignorance' (after Stirling)



In order to avoid confusion between the strict definition of the term uncertainty as used here, and the looser colloquial usage, the Field Study by Stirling introduces the term 'incertitude' to apply in a broad overarching sense to the conditions of risk, uncertainty and ignorance (as well as a fourth category of '*ambiguity*' which is not so relevant here, but whose definition is also illustrated in Figure 4). Drawing on a wide literature, the Field Study by Salo, resolves three broad dimensions cross-cutting all these forms of incertitude which are confronted in the management of technological risk.. These concern:

i) the scientific knowledge on which risk assessment/analysis depends,

ii) the stakeholders' value judgements about the impacts engendered by new technology, and

iii) the scope and efficacy of the policy measures which may be adopted to control and monitor technology.

Under each of these dimensions, a series of different sources of uncertainty and ignorance are documented by Salo and summarised (with some additions) in Table 7 below.

Table 7: Some Sources of Uncertainty and Ignorance in the Appraisal of Technological Risk (after Salo)

Physical Causation	Causes	• What particular characteristics of the technology are potentially harmful?
	Consequences	• What harmful effects is can the introduction of technology have?
	Causation	• What causal relationships govern the emergence of harmful consequences?
	Conditions	• Under what specific external circumstances may harmful consequences arise?
	Detection	• What means are available for detecting and monitoring the harms?
	Time of manifestation	• When might the harm come about?
Value Concerns	Stakeholders	• Who are the stakeholders that may be affected by the harm?
	Communication	• Do the stakeholders have sufficient, impartial and intelligible information about the technology? Can they interact with each other?
	Preferences	• Do the stakeholders have stable preferences that they can explicate?
	Representation	• What deficiencies are associated with the mechanisms of representation through which the stakeholders' views are brought into the regulatory discourse?
Policy Response	Practice	• Can we be confident that the operation of technology conforms with assumed practice?
	Measures	• What policy measures could be instituted to counter any harm?
	Effectiveness	• How effective are these measures?
	Cost	• What budgetary, social and general opportunity costs are incurred by regulation?

By reference to the regulation of energy technologies, toxic chemicals and genetically modified organisms, the Field Study by Stirling reviews a variety of practical examples of situations in the regulatory appraisal of technology which are dominated by the conditions of uncertainty and ignorance rather than risk. Indeed, the imponderables associated with global climate models, the sheer number of chemicals and the unpredictability of their behaviour in the environment and the unprecedented nature of genetic modification technology are all such as to render ignorance and uncertainty (in their formal senses) the dominant conditions in the management of each of these types of risk. Other examples from the field of technological risk are too numerous to mention.

The curious thing is, that these and other sources of intractable uncertainty and ignorance are routinely treated in the regulatory appraisal of technology by using the probabilistic techniques of risk assessment. Given the manifest inapplicability of probabilistic techniques under uncertainty and ignorance, this is a quite remarkable phenomenon. On the basis of a review of a wide literature on decision-making under incertitude, the Field Study by Stirling observes that the seductive appeal of the elegance and facility of probabilistic calculus can easily overwhelm measured judgements as to its efficacy. For all this, the continued treatment of ignorance and uncertainty as if they were mere risk provides an example of the kind of "pretence at knowledge" lamented by the economist Hayek in his Nobel acceptance speech. Either way, it is the highly circumscribed practical applicability of probabilistic techniques which forms part of the basis for the use of the term 'narrow risk' introduced in Section 2 of this Study.

The literature on the sociology of science has repeatedly documented a similar phenomenon to this 'pretence at knowledge' in considerable detail in a number of fields other than risk assessment and goes some way towards a general explanation. Here (drawing on the work of MacKenzie), the Field Study by Rip introduces the concept of the 'trough of uncertainty'. This refers to the observed tendency for the acknowledgement of ignorance and uncertainty to diminish in the intermediate domain between the forefront of research activity and its broader public dissemination. And it is precisely in this region of intermediate proximity to the knowledge production process itself, that a body of knowledge tends to be most intensively employed as a basis for action. This is certainly the case in the use of sciences such as climatology, toxicology, genetics or ecology in the regulatory appraisal of technology. In other words, it is precisely where the stakes are highest that the uncertainties (and ignorance) tend to most strongly understated.

As is noted in three of the Field Studies, similar observations underlie a series of other important themes in the study of scientific uncertainty, including Funtowicz and Ravetz's recent influential notion of 'post-normal science' and the seminal preceding concept of 'trans-science' introduced thirty years ago by Weinberg. All these bodies of work

share a common insight. It is in the recognition, characterisation and articulation of incertitude that the interplay of science, interests and values becomes most intense. The dilemma is, of course, that those who through their expertise are most in a position to document and explicate the sources of ignorance associated with a particular body of knowledge are often those who are subject to the most pronounced interests in the use of this knowledge as a basis for action. Nowhere is this more true than in the appraisal of technological risk. Though they may be explicable in sociological terms, the systematic understating of incertitude and the 'pretence at knowledge' associated with Rip's 'trough of uncertainty' are of quite profound importance in considering the practical relationships between science and precaution in the management of technological risk.

For the purposes of the present study, however, the implications are clear. Stirling points out in his Field Study that judgements concerning the extent to which "we don't know what we don't know", no matter how well informed, are ultimately unavoidably subjective and value laden. This would continue to be true, even in cases where the multiple aspects and dimensions were held to be compressible into a single metric. In a fashion similar to incommensurability, then, it seems that here again with the problem of ignorance, we encounter a rationale for a more inclusive approach to the appraisal of technological risk. It is by harnessing the imagination and intuition about different possibilities engendered by the inclusion of disparate perspectives, for instance, that the condition of ignorance can systematically be converted into uncertainty. This cannot be achieved by a process which systematically understates the property of ignorance. It is here that the wisdom of the Chinese philosopher Lao Tzu becomes highly pertinent. When he wrote that "knowing one's ignorance is the greater part of knowledge" he crafted an injunction which, as well as being both 'scientific' and 'precautionary' in sentiment, carries a profoundly important message for the management of technological risk.

6: SCIENCE SHOULD BE ON TAP, NOT ON TOP

subjectivity and interests in the framing of science

The account presented in this Report thus far may seem somewhat pessimistic – even despairing – in relation to the role of science in the management of technological risk. The Conservative British Prime Minister Winston Churchill, by contrast, was famous for his bullish attitudes. He was hardly the paradigm example of what have later in some quarters come to be seen as post-modern anxieties over risk. And yet it is to a remark of Churchill's that a central theme of this study can be linked: "science should be on tap not on top". For – in different ways – this is a key conclusion of each of the Field Studies on which this report is based.

On the face of it, this sentiment seems to run directly counter to the widespread advocacy of 'sound science' as a basis for the regulation of risk. However, it is a central finding of this project that, whether viewed from the point of view of policy analysis, science and technology studies, decision analysis or risk assessment, notions of a unitary definitive concept of 'sound science' are highly problematic. In situations where different bodies of scientific evidence, alternative theoretical paradigms or different disciplinary perspectives appear to be in tension, it is often far from clear what criteria are to be employed in determining the practical substance of 'sound science'. As is discussed in particular in the Field Study by Rip, institutional procedures for verification, validation and learning have long been the object of close scrutiny in the history, philosophy and social studies of science. What emerges throughout the field as a whole – encompassing areas far removed from the sciences of risk assessment – is that the quality of 'soundness' in science is highly ambiguous, context dependent and value laden. Indeed, where appeals to 'sound science' remain unsubstantiated by explicit criteria for what precisely is meant by 'soundness', they often amount to little more than rhetorical strategies. In short, there exist so many alternative ways of conceiving of 'sound science' and of the appropriate ways by which to achieve it in any given context, that it becomes extremely difficult to render the concept operational in any practical fashion.

Given the polarisation and strength of feeling so often encountered in discussions over the role of science in the regulation of risk, it is important to be clear from the outset about what is *not* being argued here. The argument is *not* that "nothing is more true than anything else". Put simply, the implication is rather that "a number of things may be equally true" in the appraisal of risk. It is, of course, uncontentious that policy making, regulation and the day-to-day management of technological risk must be informed by all the available empirical evidence and should be consistent with prevailing scientific understandings. However, this does not mean that science on its own should be assumed to *determine* particular regulatory or policy responses. In fact, in the terms discussed in the Field Study by Rip, the key

IPCS

INTERNATIONAL PROGRAMME ON CHEMICAL SAFETY

WHO

IPCS Harmonization Project

Uncertainty and Data Quality in Exposure Assessment

Part 1:
Guidance Document on Characterizing and Communicating Uncertainty in Exposure Assessment

Part 2:
Hallmarks of Data Quality in Chemical Exposure Assessment

IOMC

INTER-ORGANIZATION PROGRAMME FOR THE SOUND MANAGEMENT OF CHEMICALS
A cooperative agreement among FAO, ILO, UNEP, UNIDO, UNITAR, WHO and OECD

World Health Organization

Harmonization Project Document No. 6

PART 1: GUIDANCE DOCUMENT ON CHARACTERIZING AND COMMUNICATING UNCERTAINTY IN EXPOSURE ASSESSMENT

PART 2: HALLMARKS OF DATA QUALITY IN CHEMICAL EXPOSURE ASSESSMENT

This project was conducted within the IPCS project on the Harmonization of Approaches to the Assessment of Risk from Exposure to Chemicals.

Published under the joint sponsorship of the World Health Organization, the International Labour Organization and the United Nations Environment Programme, and produced within the framework of the Inter-Organization Programme for the Sound Management of Chemicals.

The **International Programme on Chemical Safety (IPCS)**, established in 1980, is a joint venture of the United Nations Environment Programme (UNEP), the International Labour Organization (ILO) and the World Health Organization (WHO). The overall objectives of the IPCS are to establish the scientific basis for assessment of the risk to human health and the environment from exposure to chemicals, through international peer review processes, as a prerequisite for the promotion of chemical safety, and to provide technical assistance in strengthening national capacities for the sound management of chemicals.

The **Inter-Organization Programme for the Sound Management of Chemicals (IOMC)** was established in 1995 by UNEP, ILO, the Food and Agriculture Organization of the United Nations, WHO, the United Nations Industrial Development Organization, the United Nations Institute for Training and Research and the Organisation for Economic Co-operation and Development (Participating Organizations), following recommendations made by the 1992 UN Conference on Environment and Development to strengthen cooperation and increase coordination in the field of chemical safety. The purpose of the IOMC is to promote coordination of the policies and activities pursued by the Participating Organizations, jointly or separately, to achieve the sound management of chemicals in relation to human health and the environment.

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PART 2: HALLMARKS OF DATA QUALITY IN CHEMICAL EXPOSURE ASSESSMENT

EXECUTIVE SUMMARY

This guidance has been developed as a basis for transparently characterizing uncertainty in chemical exposure assessment to enable its full consideration in regulatory and policy decision-making processes. Uncertainties in exposure assessment are grouped under three categories—namely, parameter, model and scenario—with the guidance addressing both qualitative and quantitative descriptions. Guidance offered here is consistent with other projects addressing exposure in the WHO/IPCS Harmonization Project, including a monograph on *IPCS Risk Assessment Terminology*, which includes a glossary of key exposure assessment terminology, and a monograph on *Principles of Characterizing and Applying Human Exposure Models*.

The framework described in this monograph is considered applicable across a full range of chemical categories, such as industrial chemicals, pesticides, food additives and others. It is intended primarily for use by exposure assessors who are not intimately familiar with uncertainty analysis. The monograph aims to provide an insight into the complexities associated with characterizing uncertainties in exposure assessment and suggested strategies for incorporating them during human health risk assessments for environmental contaminants. This is presented in the context of comparability with uncertainties associated with hazard quantification in risk assessment.

This document recommends a tiered approach to the evaluation of uncertainties in exposure assessment using both qualitative and quantitative (both deterministic and probabilistic) methods, with the complexity of the analysis increasing as progress is made through the tiers. The report defines and identifies different sources of uncertainty in exposure assessment, outlines considerations for selecting the appropriate approach to uncertainty analysis as dictated by the specific objective and identifies the information needs of decision-makers and stakeholders. The document also provides guidance on ways to consider or characterize exposure uncertainties during risk assessment and risk management decision-making and on communicating the results. Illustrative examples based on environmental exposure and risk analysis case-studies are provided.

The monograph also recommends the adoption of 10 guiding principles for uncertainty analysis. These guiding principles are considered to be the general desirable goals or properties of good exposure assessment. They are mentioned in the text where most appropriate and are supported by more detailed recommendations for good practice. The 10 guiding principles are as follows:

1) Uncertainty analysis should be an integral part of exposure assessment.

2) The level of detail of the uncertainty analysis should be based on a tiered approach and consistent with the overall scope and purpose of the exposure and risk assessment.

3) Sources of uncertainty and variability should be systematically identified and evaluated in the exposure assessment.

4) The presence or absence of moderate to strong dependencies between model inputs is to be discussed and appropriately accounted for in the analysis.

5) Data, expert judgement or both should be used to inform the specification of uncertainties for scenarios, models and model parameters.

6) Sensitivity analysis should be an integral component of the uncertainty analysis in order to identify key sources of variability, uncertainty or both and to aid in iterative refinement of the exposure model.

7) Uncertainty analyses for exposure assessment should be documented fully and systematically in a transparent manner, including both qualitative and quantitative aspects pertaining to data, methods, scenarios, inputs, models, outputs, sensitivity analysis and interpretation of results.

8) The uncertainty analysis should be subject to an evaluation process that may include peer review, model comparison, quality assurance or comparison with relevant data or independent observations.

9) Where appropriate to an assessment objective, exposure assessments should be iteratively refined over time to incorporate new data, information and methods to better characterize uncertainty and variability.

10) Communication of the results of exposure assessment uncertainties to the different stakeholders should reflect the different needs of the audiences in a transparent and understandable manner.

1. INTRODUCTION

Individuals are exposed to a wide variety of chemicals in various indoor and outdoor microenvironments during the course of a typical day through inhalation, ingestion or dermal contact. *Exposure* is defined as contact between an *agent* and a *target*, where contact takes place on an *exposure surface* over an *exposure period* (Zartarian et al., 1997; IPCS, 2004). In the case of the present monograph, the agents of concern are chemical—although the World Health Organization (WHO)/International Programme on Chemical Safety (IPCS) Working Group considered the guidance to be also broadly applicable to other (physical and biological) agents. The targets are children, adults or sensitive subgroups in populations; the exposure surfaces are the external human boundaries (e.g. skin) or internal organs (e.g. gastrointestinal tract, lung surface); the exposure duration may be short (i.e. from minutes to hours to a day) or long (i.e. from days to months to a lifetime); and the health effects may be acute, intermittent or chronic. The process of estimating or measuring the magnitude, frequency and duration of exposure to an agent, along with the number and characteristics of the population exposed, is called an *exposure assessment*. In some health studies, the term "exposure assessment" may also include assessing the dose within the body after the agent enters the body via ingestion, inhalation or dermal absorption. This absorbed dose of the agent or its metabolite is also known as the *uptake*.

Historically, risk assessments have included four principal components: *hazard identification*,[1] or the identification of the type and nature of adverse effects that an agent has the inherent capacity to cause; *hazard characterization*, or the qualitative and, wherever possible, quantitative description of the inherent property of the agent of concern; *exposure assessment*, or the assessment of the magnitude of likely human exposures of an individual or a population to that agent; and *risk characterization*, or the qualitative and, wherever possible, quantitative determination of the probability of occurrence of adverse effects of the agent under defined exposure conditions. The entire risk assessment process is itself only one component of *risk analysis*, the other two being *risk management* and *risk communication*. Risk reduction is often achieved through exposure mitigation. Therefore, knowledge of the exposure is the basic prerequisite for risk characterization and for characterizing subsequent risk management strategies. The importance of exposure assessment is to provide information about the nature of the source and route of exposure and the individuals who are exposed. Risks cannot be reliably estimated if exposures and their uncertainties are not properly characterized and sufficiently quantified.

There are a number of aspects that must be taken into account in accurate estimation of exposure. Quantification of the magnitude and timing of personal exposures to agents of concern requires the identification of sources and media of concern, key exposure microenvironments, and routes and pathways of exposure that contribute most to an individual's exposure. Unfortunately, the information base on which to estimate emissions, concentrations, exposures and doses associated with each of these steps is sometimes completely lacking, frequently incomplete, not representative or otherwise uncertain. Given

[1] See the IPCS document on risk assessment terminology (IPCS, 2004). Important definitions are repeated in the text. Definitions of selected terms not included in IPCS (2004) are given in the Glossary of terms.

that complete information is never available, exposure assessors must make simplifying assumptions (e.g. use defaults) or rely on data that are not necessarily representative of the populations or conditions of interest (e.g. by extrapolating results that have been generated for other purposes). For example, concentrations of dioxins may be available for only one species of fish, so it may be necessary to extrapolate from these data to other species, if an estimate of overall exposure to dioxins from fish consumption is required.

Uncertainties in risk assessment include considerations related to missing, incomplete and/or incorrect knowledge, as well as those associated with ignorance and/or lack of awareness. Uncertainties should be characterized as transparently as possible to ensure their adequate consideration in decision-making concerning the need for and nature of appropriate risk management and communication.

Part 2 of this Harmonization Project Document is on data quality for chemical exposure assessment, because of its importance to the acceptance and credibility of the evaluation of uncertainty in an exposure analysis. Data quality for the purposes of this report deals with the completeness and clarity with which uncertainty is explained. This means that data with high uncertainty may be of high quality if the data and its uncertainty are clearly explained and carefully documented. A high-quality evaluation of uncertainty in an exposure analysis would provide the readers with the ability to re-evaluate all the choices and trade-offs made in the evaluation and to explore alternative choices and trade-offs. This is a difficult goal to achieve in most cases.

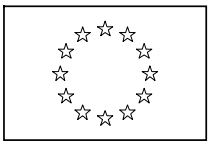
1.1 Why uncertainty analysis?

Uncertainty in risk assessment in the general sense is defined by IPCS (2004) as “imperfect knowledge concerning the present or future state of an organism, system, or (sub)population under consideration”. In the context of exposure assessment, the exposures may be past, present or predicted future exposures, and the uncertainties in respect of each may differ. An adequate characterization of the uncertainties in exposure assessment is essential to the transparency of risk assessment and characterization of relevant data gaps to improve defensibility; it is also a critical basis for informed decision-making regarding the need for action to reduce risk and the nature of appropriate measures. Uncertainties should be considered explicitly in each step of the analysis and communicated throughout the process.

For exposure assessors, uncertainty analysis increases transparency and, thereby, the credibility of the process. Consequently, reliance on worst-case assumptions can be reduced and decision support improved. Uncertainty analysis also identifies important data gaps, which can be filled to improve the accuracy of estimation.

The consideration and expression of uncertainty are given particular attention in the Working Principles for Risk Analysis recently adopted by the Codex Alimentarius Commission (Codex, 2005: p. 104):

> 23. Constraints, uncertainties and assumptions having an impact on the risk assessment should be explicitly considered at each step in the risk assessment and documented in a transparent manner. Expression of uncertainty or variability in risk estimates may be qualitative or quantitative, but should be quantified to the extent that is scientifically achievable.



COMMISSION OF THE EUROPEAN COMMUNITIES

Brussels, 2.2.2000
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COMMUNICATION FROM THE COMMISSION

on the precautionary principle

SUMMARY

1. The issue of when and how to use the precautionary principle, both within the European Union and internationally, is giving rise to much debate, and to mixed, and sometimes contradictory views. Thus, decision-makers are constantly faced with the dilemma of balancing the freedom and rights of individuals, industry and organisations with the need to reduce the risk of adverse effects to the environment, human, animal or plant health. Therefore, finding the correct balance so that the proportionate, non-discriminatory, transparent and coherent actions can be taken, requires a structured decision-making process with detailed scientific and other objective information.

2. The Communication's fourfold aim is to:

   - outline the Commission's approach to using the precautionary principle,
   - establish Commission guidelines for applying it,
   - build a common understanding of how to assess, appraise, manage and communicate risks that science is not yet able to evaluate fully, and
   - avoid unwarranted recourse to the precautionary principle, as a disguised form of protectionism.

   It also seeks to provide an input to the ongoing debate on this issue, both within the Community and internationally.

3. The precautionary principle is not defined in the Treaty, which prescribes it only once - to protect the environment. But *in practice*, its scope is much wider, and specifically where preliminary objective scientific evaluation, indicates that there are reasonable grounds for concern that the potentially dangerous effects on the *environment, human, animal or plant health* may be inconsistent with the high level of protection chosen for the Community.

   The Commission considers that the Community, like other WTO members, has the right to establish the level of protection - particularly of the environment, human, animal and plant health, - that it deems appropriate. Applying the precautionary principle is a key tenet of its policy, and the choices it makes to this end will continue to affect the views it defends internationally, on how this principle should be applied.

4. The precautionary principle should be considered within a structured approach to the analysis of risk which comprises three elements: risk assessment, risk management, risk communication. The precautionary principle is particularly relevant to the management of risk.

   The precautionary principle, which is essentially used by decision-makers in the management of risk, should not be confused with the element of caution that scientists apply in their assessment of scientific data.

Recourse to the precautionary principle presupposes that potentially dangerous effects deriving from a phenomenon, product or process have been identified, and that scientific evaluation does not allow the risk to be determined with sufficient certainty.

The implementation of an approach based on the precautionary principle should start with a scientific evaluation, as complete as possible, and where possible, identifying at each stage the degree of scientific uncertainty**.**

5. Decision-makers need to be aware of the degree of uncertainty attached to the results of the evaluation of the available scientific information. Judging what is an "acceptable" level of risk for society is an eminently *political* responsibility. Decision-makers faced with an unacceptable risk, scientific uncertainty and public concerns have a duty to find answers. Therefore, all these factors have to be taken into consideration.

In some cases, the right answer may be not to act or at least not to introduce a binding legal measure. A wide range of initiatives is available in the case of action, going from a legally binding measure to a research project or a recommendation.

The decision-making procedure should be transparent and should involve as early as possible and to the extent reasonably possible all interested parties.

6. Where action is deemed necessary, measures based on the precautionary principle should be, *inter alia*:

- *proportional* to the chosen level of protection,
- *non-discriminatory* in their application,
- *consistent* with similar measures already taken,
- *based on an examination of the potential benefits and costs* of action or lack of action (including, where appropriate and feasible, an economic cost/benefit analysis),
- *subject to review,* in the light of new scientific data, and
- *capable of assigning responsibility for producing the scientific evidence* necessary for a more comprehensive risk assessment.

Proportionality means tailoring measures to the chosen level of protection. Risk can rarely be reduced to zero, but incomplete risk assessments may greatly reduce the range of options open to risk managers. A total ban may not be a proportional response to a potential risk in all cases. However, in certain cases, it is the sole possible response to a given risk.

Non-discrimination means that comparable situations should not be treated differently, and that different situations should not be treated in the same way, unless there are objective grounds for doing so.

Consistency means that measures should be of comparable scope and nature to those already taken in equivalent areas in which all scientific data are available.

Examining costs and benefits entails comparing the overall cost to the Community of action and lack of action, in both the short and long term. This is not simply an economic cost-benefit analysis: its scope is much broader, and includes non-economic considerations, such as the efficacy of possible options and their acceptability to the public. In the conduct of such an examination, account should be taken of the general principle and the case law of the Court that the protection of health takes precedence over economic considerations.

Subject to review in the light of new scientific data, means measures based on the precautionary principle should be maintained so long as scientific information is incomplete or inconclusive, and the risk is still considered too high to be imposed on society, in view of chosen level of protection. Measures should be periodically reviewed in the light of scientific progress, and amended as necessary.

Assigning responsibility for producing scientific evidence is already a common consequence of these measures. Countries that impose a prior approval (marketing authorisation) requirement on products that they deem dangerous *a priori* reverse the burden of proving injury, by treating them as dangerous unless and until businesses do the scientific work necessary to demonstrate that they are safe.

Where there is no prior authorisation procedure, it may be up to the user or to public authorities to demonstrate the nature of a danger and the level of risk of a product or process. In such cases, a specific precautionary measure might be taken to place the burden of proof upon the producer, manufacturer or importer, but this cannot be made a general rule.

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The Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement) clearly sanctions the use of the precautionary principle, although the term itself is not explicitly used. Although the general rule is that all sanitary and phytosanitary measures must be based on scientific principles and that they should not be maintained without adequate scientific evidence, a derogation from these principles is provided for in Article 5 (7) which stipulates that: "*in cases where relevant scientific evidence is insufficient, a Member may provisionally adopt sanitary or phytosanitary measures on the basis of available pertinent information, including that from the relevant international organizations as well as from sanitary or phytosanitary measures applied by other Members. In such circumstances, Members shall seek to obtain the additional information necessary for a more objective assessment of risk and review the sanitary or phytosanitary measure accordingly within a reasonable period of time.*"

Hence, according to the SPS Agreement, measures adopted in application of a precautionary principle when the scientific data are inadequate, are provisional and imply that efforts be undertaken to elicit or generate the necessary scientific data. It is important to stress that the provisional nature is not bound up with a time limit but with the development of scientific knowledge.

The use of the term "more objective assessment of risk" in Article 5.7 infers that a precautionary measure may be based on a less objective appraisal but must nevertheless includes an evaluation of risk.

The concept of risk assessment in the SPS leaves leeway for interpretation of what could be used as a basis for a precautionary approach. The risk assessment on which a measure is based may include non-quantifiable data of a factual or qualitative nature and is not uniquely confined to purely quantitative scientific data. This interpretation has been confirmed by the WTO's Appellate body in the case of growth hormones, which rejected the panel's initial interpretation that the risk assessment had to be quantitative and had to establish a minimum degree of risk.

The principles enshrined in Article 5.7 of the SPS must be respected in the field of sanitary and phytosanitary measures; however, because of the specific nature of other areas, such as the environment, it may be that somewhat different principles will have to be applied.

International guidelines are being considered in relation to the application of the Precautionary Principle in Codex Alimentarius. Such guidance in this, and other sectors, could pave the way to a harmonised approach by the WTO Members, to drawing up health or environment protection measures, while avoiding the misuse of the precautionary principle which could otherwise lead to unjustifiable barriers to trade.

In the light of these observations, the Commission considers that, following the example set by other Members of the WTO, the Community is entitled to prescribe the level of protection, notably as regards the environment and human, animal and plant health, which it considers appropriate. In this context, the Community must respect Articles 6, 95, 152 and 174 of the Treaty. To this end,

reliance on the precautionary principle constitutes an essential plank of its policy. It is clear that the choices made will affect its positions at international and notably multilateral level, as regards recourse to the precautionary principle.

> Bearing in mind the very origins of the precautionary principle and its growing role in international law, and notably in the agreements of the World Trade Organisation, this principle must be duly addressed at international level in the various areas in which it is likely to be of relevance.
>
> ***Following the example set by the other members of the WTO, the Commission considers that the Community is entitled to prescribe the level of protection, notably as regards environmental protection and human, animal and plant health, that it considers appropriate. Recourse to the precautionary principle is a central plank of Community policy. The choices made to this end will continue to influence its positions at international level, and notably at multinational level, as regards the precautionary principle.***

5. THE CONSTITUENT PARTS OF THE PRECAUTIONARY PRINCIPLE

An analysis of the precautionary principle reveals two quite distinct aspects: (i) **the political decision to act or not to act as such,** which is linked to the **factors triggering** recourse to the precautionary principle; (ii) in the affirmative, **how to act, i.e.** the **measures** resulting from application of the precautionary principle.

There is a controversy as to the role of scientific uncertainty in risk analysis, and notably as to whether it belongs under risk assessment or risk management. This controversy springs from a confusion between a prudential approach and application of the precautionary principle. These two aspects are complementary but should not be confounded.

The prudential approach is part of risk assessment policy which is determined before any risk assessment takes place and which is based on the elements described in 5.1.3; it is therefore an integral part of the scientific opinion delivered by the risk evaluators.

On the other hand, application of the precautionary principle is part of risk management, when scientific uncertainty precludes a full assessment of the risk and when decision-makers consider that the chosen level of environmental protection or of human, animal and plant health may be in jeopardy.

The Commission considers that measures applying the precautionary principle belong in the general framework of risk analysis, and in particular risk management.

5.1. Factors triggering recourse to the precautionary principle

The precautionary principle is relevant only in the event of a potential risk, even if this risk cannot be fully demonstrated or quantified or its effects determined because of the insufficiency or inclusive nature of the scientific data.

It should however be noted that the precautionary principle can under no circumstances be used to justify the adoption of arbitrary decisions.

5.1.1. Identification of potentially negative effects

Before the precautionary principle is invoked, the scientific data relevant to the risks must first be evaluated. However, one factor logically and chronologically precedes the decision to act, namely identification of the potentially negative effects of a phenomenon. To understand these effects more thoroughly it is necessary to conduct a scientific examination. The decision to conduct this examination without awaiting additional information is bound up with a less theoretical and more concrete perception of the risk.

5.1.2. Scientific evaluation

A scientific evaluation of the potential adverse effects should be undertaken based on the available data when considering whether measures are necessary to protect the environment, the human, animal or plant health. An assessment of risk should be considered where feasible when deciding whether or not to invoke the precautionary principle. This requires reliable scientific data and logical reasoning, leading to a conclusion which expresses the possibility of occurrence and the severity of a hazard's impact on the environment, or health of a given population including the extent of possible damage, persistency, reversibility and delayed effect. However it is not possible in all cases to complete a comprehensive assessment of risk, but all effort should be made to evaluate the available scientific information.

Where possible, a report should be made which indicates the assessment of the existing knowledge and the available information, providing the views of the scientists on the reliability of the assessment as well as on the remaining uncertainties. If necessary, it should also contain the identification of topics for further scientific research.

Risk assessment consists of four components - namely hazard identification, hazard characterisation, appraisal of exposure and risk characterisation (Annex III). The limits of scientific knowledge may affect each of these components, influencing the overall level of attendant uncertainty and ultimately affecting the foundation for protective or preventive action. An attempt to complete these four steps should be performed before decision to act is taken.

5.1.3. Scientific uncertainty

Scientific uncertainty results usually from five characteristics of the scientific method : the variable chosen, the measurements made, the samples drawn, the models used and the causal relationship employed. Scientific uncertainty may

also arise from a controversy on existing data or lack of some relevant data . Uncertainty may relate to qualitative or quantitative elements of the analysis.

A more abstract and generalised approach preferred by some scientists is to separate all uncertainties into three categories of – Bias, Randomness and True Variability. Some other experts categorise uncertainty in terms of estimation of confidence interval of the probability of occurrence and of the severity of the hazard's impact.

This issue is very complex and the Commission launched a project "Technological Risk and the Management of Uncertainty" conducted under the auspices of the European Scientific Technology Observatory. The four ESTO reports will be published shortly and will give a comprehensive description of scientific uncertainty.

Risk evaluators accommodate these uncertainty factors by incorporating prudential aspects such as :

– relying on animal models to establish potential effects in man;

– using body weight ranges to make inter-species comparisons;

– adopting a safety factor in evaluating an acceptable daily intake to account for intra- and inter-species variability; the magnitude of this factor depends on the degree of uncertainty of the available data;

– not adopting an acceptable daily intake for substances recognised as genotoxic or carcinogenic;

– adopting the "ALARA" (as low as reasonably achievable) level as a basis for certain toxic contaminants.

Risk managers should be fully aware of these uncertainty factors when they adopt measures based on the scientific opinion delivered by the evaluators.

However, in some situations the scientific data are not sufficient to allow one to apply these prudential aspects in practice, i.e. in cases in which extrapolations cannot be made because of the absence of parameter modelling and where cause-effect relationships are suspected but have not been demonstrated. It is in situations like these that decision-makers face the dilemma of having to act or not to act.

Recourse to the precautionary principle presupposes:

– ***identification of potentially negative effects resulting from a phenomenon, product or procedure;***

– ***a scientific evaluation of the risk which because of the insufficiency of the data, their inconclusive or imprecise nature, makes it impossible to determine with sufficient certainty the risk in question.***

5.2. Measures resulting from reliance on the precautionary principle

5.2.1. The decision whether or not to act

In the kind of situation described above - sometimes under varying degrees of pressure from public opinion - decision-makers have to respond. However, responding does not necessarily mean that measures always have to be adopted. The decision to do nothing may be a response in its own right.

The appropriate response in a given situation is thus the result of an eminently political decision, a function of the risk level that is "acceptable" to the society on which the risk is imposed.

5.2.2. Nature of the action ultimately taken

The nature of the decision influences the type of control that can be carried out. Recourse to the precautionary principle does not necessarily mean adopting final instruments designed to produce legal effects that are open to judicial review. There is a whole range of actions available to decision-makers under the head of the precautionary principle. The decision to fund a research programme or even the decision to inform the public about the possible adverse effects of a product or procedure may themselves be inspired by the precautionary principle.

It is for the Court of Justice to pronounce on the legality of any measures taken by the Community institutions. The Court has consistently held that when the Commission or any other Community institution has broad discretionary powers, notably as regards the nature and scope of the measures it adopts, review by the Court must be limited to examining whether the institution committed a manifest error or misuse of power or manifestly exceed the limits of its powers of appraisal.

Hence the measures may not be of an arbitrary nature.

Recourse to the precautionary principle does not necessarily mean adopting final instruments designed to produce legal effects, which are subject to judicial review.

6. GUIDELINES FOR APPLYING THE PRECAUTIONARY PRINCIPLE.

6.1. Implementation

When decision-makers become aware of a risk to the environment or human, animal or plant health that in the event of non-action may have serious consequences, the question of appropriate protective measures arise. Decision-makers have to obtain, through a structured approach, a scientific evaluation, as complete as possible, of the risk to the environment, or health, in order to select the most appropriate course of action

The determination of appropriate action including measures based on the precautionary principle should start with a scientific evaluation and, if necessary, the decision to commission scientists to perform an as objective and complete as possible scientific evaluation. It will cast light on the existing objective evidence, the gaps in knowledge and the scientific uncertainties.

The implementation of an approach based on the precautionary principle should start with a scientific evaluation, as complete as possible, and where possible, identifying at each stage the degree of scientific uncertainty.

6.2. The triggering factor

Once the scientific evaluation has been performed as best as possible, it may provide a basis for triggering a decision to invoke the precautionary principle. The conclusions of this evaluation should show that the desired level of protection for the environment or a population group could be jeopardised. The conclusions should also include an assessment of the scientific uncertainties and a description of the hypotheses used to compensate for the lack of the scientific or statistical data. An assessment of the potential consequences of inaction should be considered and may be used as a trigger by the decision-makers. The decision to wait or not to wait for new scientific data before considering possible measures should be taken by the decision-makers with a maximum of transparency. The absence of scientific proof of the existence of a cause-effect relationship, a quantifiable dose/response relationship or a quantitative evaluation of the probability of the emergence of adverse effects following exposure should not be used to justify inaction. Even if scientific advice is supported only by a minority fraction of the scientific community, due account should be taken of their views, provided the credibility and reputation of this fraction are recognised.[2]

The Commission has confirmed its wish to rely on procedures as transparent as possible and to involve all interested parties at the earliest possible stage[3]. This will assist decision makers in taking legitimate measures which are likely to achieve the society's chosen level of health or environmental protection

An assessment of the potential consequences of inaction and of the uncertainties of the scientific evaluation should be considered by decision-makers when determining whether to trigger action based on the precautionary principle.

All interested parties should be involved to the fullest extent possible in the

[2] *cf The WTO Appellate Body report on hormones, paragraph 124 : « In some cases, the very existence of divergent views presented by qualified scientists who have investigated the particular issue at hand, may indicate a state of scientific uncertainty »*

[3] *A considerable effort has already been made notably as regards public health and the environment. As regards the latter, the Community and the Member States have demonstrated the importance they attach to access to information and justice by signing the Aarhus Convention of June 1998.*

- The measures must be periodically reviewed to take account of new scientific data. The results of scientific research should make it possible to complete the risk evaluation and if necessary to review the measures on the basis of the conclusions.

- Hence the reasonable period envisaged in the SPS Agreement includes the time needed for completion of the necessary scientific work and, besides, the time needed for performance of a risk evaluation based on the conclusions of this scientific work. It should not be possible to invoke budgetary constraints or political priorities to justify excessive delays in obtaining results, re-evaluating the risk or amending the provisional measures.

Research could also be conducted for the improvement of the methodologies and instruments for assessing risk, including greater integration of all pertinent factors (e.g. socio-economic information, technological perspectives).

The measures, although provisional, shall be maintained as long as the scientific data remain incomplete, imprecise or inconclusive and as long as the risk is considered too high to be imposed on society.

Maintenance of the measures depends on the development of scientific knowledge, in the light of which they should be reevaluated. This means that scientific research shall be continued with a view to obtaining more complete data.

Measures based on the precautionary principle shall be reexamined and if necessary modified depending on the results of the scientific research and the follow up of their impact.

6.4. The burden of proof

- Community rules and those of many third countries enshrine the principle of prior approval (positive list) before the placing on the market of certain products, such as drugs, pesticides or food additives. This is one way of applying the precautionary principle, by shifting responsibility for producing scientific evidence. This applies in particular to substances deemed "a priori" hazardous or which are potentially hazardous at a certain level of absorption. In this case the legislator, by way of precaution, has clearly reversed the burden of proof by requiring that the substances be deemed hazardous until proven otherwise. Hence it is up to the business community to carry out the scientific work needed to evaluate the risk. As long as the human health risk cannot be evaluated with sufficient certainty, the legislator is not legally entitled to authorise use of the substance, unless exceptionally for test purposes.

- In other cases, where such a prior approval procedure does not exist, it may be for the user, a private individual, a consumer association, citizens or the public authorities to demonstrate the nature of a danger and the level of risk posed by a product or process. Action taken under the head of the

ANNEX III

THE FOUR COMPONENTS OF RISK ASSESSMENT

An attempt to complete as far as possible these four components should be performed before action is taken.

Hazard identification means identifying the biological, chemical or physical agents that may have adverse effects. A new substance or biological agent may reveal itself through its effects on the population (illness or death), or on the environment and it may be possible to describe the actual or potential effects on the population or environment before the cause is identified beyond doubt.

Hazard characterisation consists of determining, in quantitative and/or qualitative terms, the nature and severity of the adverse effects associated with the causal agents or activity. It is at this stage that a relationship between the amount of the hazardous substance and the effect has to be established. However, the relationship is sometimes difficult or impossible to prove, for instance because the causal link has not been established beyond doubt.

Appraisal of exposure consists of quantitatively or qualitatively evaluating the probability of exposure to the agent under study. Apart from information on the agents themselves (source, distribution, concentrations, characteristics, etc.), there is a need for data on the probability of contamination or exposure of the population or environment to the hazard.

Risk characterisation corresponds to the qualitative and/or quantitative estimation, taking account of inherent uncertainties, of the probability, of the frequency and severity of the known or potential adverse environmental or health effects liable to occur. It is established on the basis of the three preceding and closely depends on the uncertainties, variations, working hypotheses and conjectures made at each stage of the process. When the available data are inadequate or non-conclusive, a prudent and cautious approach to environmental protection, health or safety could be to opt for the worst-case hypothesis. When such hypotheses are accumulated, this will lead to an exaggeration of the real risk but gives a certain assurance that it will not be underestimated.