# Environment Committee

# Meeting 2020-02

## October 28, 2020

People. Goods.
Canada moves by rail.

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Railway Association of Canada

Railway Association of Canada | RIGHT TRACK

**RAC Environment Committee Meeting Agenda**

**Wednesday October 28th, 2020**
**1:00 pm – 3:00 pm, Eastern Daylight Time**

[Video conference link]

| | Items for Discussion | Lead | Time | D/I | Enc. |
|---|---|---|---|---|---|
| 1. | Introductory Remarks and Administrative Issues: | | | | |
| 1.1 | Welcome: Call to Order | Ben | 1:00 | - | - |
| 1.2 | Approval of Meeting Minutes | Chantale | 1:02 | D | 📎 |
| 1.3 | Review Terms of Reference | David | 1:10 | D | 📎 |
| Standing Items | | | | | |
| 2. | Regulatory Affairs Issues | | | | |
| A. | Regulatory Amendments: | | 1:15 | I | 📎 |
| | Federal Monitoring: | | | | |
| | 1) *Cross-Border Movement of Hazardous Waste and Hazardous Recyclable Material Regulations* | Ben | | I | 📎 |
| | 2) Federal Carbon Pricing System Update | Ben | | I | 📎 |
| | 3) *Canadian Environmental Protection Act* | | | | |
| | • CEPA Reform | Peter | | I | 📎 |
| | • Glyphosate Monitoring | | | | |
| | • Plastic Pollution | Ben | | I | - |
| | 4) Special Review of Pentachlorophenol by Health Canada (PMRA) | Ben | | I | - |
| | 5) Polychlorinated biphenyls (PCBs) Regulations | Ben | | I | - |
| | 6) ECCC Forward Regulatory Plan | Peter | | | |
| | 7) Canadian Chamber of Commerce: CFS Campaign | Ben | | I | 📎 |
| | Provincial Monitoring: | | | | |
| | 8) Port of Vancouver Consultation | Peter | | I | 📎 |
| | 9) Modernizing Hazardous Waste Reporting in Ontario | Ben | | I | 📎 |
| B. | Health Break | ALL | 1:35 | - | - |
| C. | Regulatory Activities: | ALL | 1:40 | I | - |
| | 1) Member reports<br>(i.e. provide update on last 6 months regarding environmental matters, existing and future initiatives, and any lessons learned or best practices to share) | | | | |

| | Items for Discussion | Lead | Time | D/I | Enc. |
|---|---|---|---|---|---|
| 3. | Industry Programs & Studies | | | | |
| A. | Programs & Studies | | | | |
| | 1) Hydrail SOW | Peter | 1:55 | I | ✍ |
| | 2) EMS Pilot Study with Southern Railway of British Columbia Ltd. | E. Mak | 2:00 | I | - |
| | 3) Memorandum of Understanding (MOU) to reduce locomotive emissions. | | | | |
| | a) Rail Pathway Initiative | C. LaRochelle | 2:10 | I | ✍ |
| | b) Clean Fuel Standard | | | I | ✍ |
| | 4) Hydrogen Strategy for Canada | | | | |
| | a) Tabitha Takeda, NRCAN | T. Takeda | 2:40 | I | - |
| 4. | Miscellaneous Items & Closing | | | | |
| A. | Wrap-Up & Look Ahead | Ben | 3:00 | - | - |

**LEGEND:** I = For Information; D = For Decision & Action; ✍ = document enclosed

**RAC Environment Committee Meeting Minutes**

March 9th, 2020
Teleconference & Webinar
12:00 – 14:00 PM EST

**Attendees:**

Brianna Bowman (RAC)
Ben Chursinoff (RAC)
Chantale Despres (CN)
Robert Eager (NBM)
Robert Gaudet (exo)
Michael Gullo (RAC)
David Huck (CP)
Murray MacBeth (GWRR)
Emily Mak (SRY)
Bruno Riendeau (VIA)
Thomas Rolland (exo)
Enrique Rosales (RAC)

**Absent:**

Emily Cosburn (Metrolinx), Keith Dagg (Translink), Stephanie Daneau (exo), Spencer Grecco (Metrolinx), Benoit Gringas (exo), Joe Van Humbeck (CP), Stella Karnis (CN), Arjun Kasturi (Metrolinx), Andre Lapalme (GWRR), Adrian Atena-Russell (GWRR)

1. **Introductory Remarks and Administrative Issues:**

**A. Call to Order**

Chantale Despres called the meeting to order at 12:03 PM EST.

**B. Review of Meeting Minutes and Action Items**

The November 22, 2019 Environment Committee meeting minutes were reviewed and unanimously approved.

Michael Gullo provided updates on the list of action items as follows:

- **Action Item 1:** *Cross-Border Movement of Hazardous Waste and Hazardous Recyclable Material Regulations*: The regulations are currently in Gazette I. A working group has been established to consult with Environment and Climate Change Canada (ECCC) and review the requirements of the regulations. Mr. Ken Roberge has conducted an analysis on the regulatory requirements against consists provided by CN and CP. The results will be shared with industry for comment before filing with ECCC.
- **Action Item 2:** letter to federal government to discuss the rail industry's role in climate change mitigation – item on hold as RAC waits for Minister of Environment to testify before committee and outline priorities.
- **Action Item 3:** Expressions of interest were sent to CN and CP in December 2019 for a meeting with Ontario Minister of Environment, Conservation and Parks. Follow-up with CN and CP will occur to confirm interest.
- **Action Items 4, 5, 7** are complete and can be removed.
- **Action Item 6**: Natural Resources Canada (NRCan) has yet to respond. Correspondence will be shared when available.
- **Action Items 8 and 9** are ongoing.

2. **Standing Items:**

A. **Regulatory Amendments:**

i. **Federal Minister of Environment Mandate Letter**

The mandate letter for the Federal Minister of Environment and Climate Change was reviewed. Priority areas relevant in the mandate letter include net-zero initiatives, flood mapping in Canada, species at risk, and the *Canadian Environmental Protection Act* (CEPA) review process.

A question from CP was raised regarding CEPA and whether it would open the door to reviewing glyphosate or creosote. At this point there is no indication to confirm that the review will address the modifications to the *Cross-Border Movement of Hazardous Waste and Hazardous Recyclable Material Regulation* which touch upon creosote or any emerging issues associated with the use of glyphosate in Canada.

The Committee was informed that Minister Wilkinson is yet to appear before the Standing Committee on Environment and Sustainable Development but is expected to do imminently. His appearance will provide additional insight about the department's priorities.

**Action Item: RAC will reach out to its multi-stakeholder group to determine if glyphosate should be part of the CEPA regulatory review. Michael Gullo to take the lead.**

ii. ***Cross-Border Movement of Hazardous Waste and Hazardous Recyclable Material Regulations***

A review and comparison of CN and CP consists to the regulatory requirements has been completed by Ken Roberge. Upon review and approval from the working group, the results will be shared with ECCC and follow-up will occur to facilitate dialogue between the department and railways.

iii. **Carbon Pricing**

The Court of Appeal of Alberta recently overturned the federal carbon pricing system. It ruled 4-1 that the system is unconstitutional. RAC will continue to monitor and confirm if the decision is appealed by the Supreme Court of Canada.

iv. **Glyphosate**

A conference call was held in December 2019 with rail companies in Canada and the U.S. to discuss the use of glyphosate and any challenges arising from its use. The discussion highlighted that railways use glyphosate on account of its effectiveness and affordable cost, while most railways have received inquiries or expressions of concern from stakeholders. Moving forward glyphosate will continue to be monitored.

The U.S. Environmental Protection Agency published in January 2020 that there are no risks to human health when the product is used in accordance with its label.

The determination is helpful but public opinion continues to question the safety of glyphosate.

In January 2019, Health Canada issued a statement on glyphosate. A re-evaluation of glyphosate in 2017 was completed and determined that it did not present a human health risk. However, numerous objections were raised, and therefore Health Canada completed another scientific review. It concluded that the concerns raised by the objectors could not be scientifically supported when considering the entire body of relevant data. The objections raised did not create doubt or concern regarding the scientific basis for the 2017 re-evaluation decision for glyphosate. The Department's final decision therefore stands.

**Action Item: RAC to correspond with AAR about glyphosate to determine if there are any next steps forward.**

**Action Item: RAC to review federal Minister of Health mandate letter for relevance to glyphosate.**

**v. Clean Fuel Standard (CFS)**

ECCC has indicated that will publish draft regulations in the Canada Gazette I in early 2020. RAC will continue monitoring and will work with members to provide comments and determine the need to develop coalition positions with other industries or supply chain partners.

Concerns were raised over the multi-jurisdictional approach that is occurring in Canada. Some provinces are moving forward with their own clean fuel standards (e.g., Alberta and Manitoba have announced new clean fuel standards) which can potentially create challenges for railways that operate through multiple departments.

**Action Item: RAC to contact ECCC for confirmation of the proposed timelines concerning the CFS.**

**Action Item: RAC to conduct analysis on provincial clean fuel standard initiatives and impacts to railways.**

**vi. Quebec Rail Summit**

The Quebec government engaged with stakeholders in 2019 and has announced a new program for shortline railways in Quebec. This program provides capital support to provincially regulated shortlines for infrastructure investments. The program provides funding for up to 2/3 of the project cost, up to $1M. It does not replace the existing PREGTI program, which supports the increased movements of goods by low-emitting modes such as rail.

In its pre-budgetary submission to the Government of Quebec, RAC recommended that infrastructure programs should also be made available to federally regulated shortlines operating in the province as well.

**Action Item: RAC to reach out to other industry groups to promote the infrastructure funding opportunity in Quebec.**

### B. Regulatory Updates from Committee Members

**VIA Rail – Mr. Bruno Riendeau**

- No environmental inspections or audits to report.
- A derailment in December occurred in Manitoba on CN track. Mr. Riendeau confirmed that CN is leading the clean up effort.

**CP Rail – Mr. David Huck**

- Two derailments occurred in Saskatchewan and the sites are being cleaned up.
- No inspections or audits to report.
- CP has a several propane storage tanks in use across their network. Locations exceeding a total storage volume threshold of 4.5 tonnes per site are required to be registered as per *The Environmental Emergencies Regulations. Sites containing large volumes or tanks may require* emergency plans that need to be developed. An exemption exists for certain substances that are used for heating buildings. Environment and Climate Change Canada is supposed to come out with guidance and CP is waiting to see what comes out as it will impact CP.

**CN Rail – Ms. Chantale Despres**

- No updates to report.

**NBM Rail - Mr. Robert Eager**

- No updates to report.

**SRY - Ms. Emily Mak**

- Transport Canada conducted a Safety Management System (SMS) audit in February.

**exo – Mr. Thomas Rolland**

- exo experienced a misunderstanding with the Ministry of Environment over a disposal facility. The Ministry of Environment determined that exo failed to submit a report but exo clarified that it does not discharge into water systems but rather sewer systems. The case has been resolved.

## 3. Industry Programs & Studies

### A. CUTRIC Presentation

CUTRIC was contracted by Transport Canada to look at the market readiness of various technologies and practices that could be adopted by the rail sector to help them reduce emissions. Dr. Josipa Petrunic, provided an overview of CUTRIC's engagement process and provided a summary of a number of project proposals for the rail sector. The final report will be made publicly available.

**B. EMS Pilot Study**

Mr. Gullo provided background information and context regarding the RAC's sustainability framework, which includes best practices and guidance for Emergency Management Systems (EMS), Safey Management Systems and Emergency Preparedness and Response.  RAC is in the process of piloting its EMS guidance with SRY this year.

Ms. Mak provided background information on SRY and explained that the company is committed to developing and enhancing its EMS through the pilot project with assistance from the RAC and Ken Roberge.

Committee members expressed the potential to develop an EMS Working Group (or meeting) that convenes industry practitioners to discuss best practices and advance knowledge sharing.

**Action Item: RAC to establish an EMS Working Group after completion of the EMS pilot at SRY.**

**C. Transport Canada Update on the Pathway Initiative**

Mr. Daniel Fairbairn, from Transport Canada (TC), provided an update on the Pathway Initiative. The contracting phase has been completed and Pollution Probe is under contract until the end of May, when the final report is due. Delphi will be providing support to the Pollution Probe's project team. A workplan is being developed by Pollution Probe and the RAC will have an opportunity to review it and provide feedback.

TC will undertake the first phase of the initiative which identifies existing programs, policies and technologies. The second phase, lead by the RAC, will identify a roadmap for government and industry to move forward on together.

A kick-off meeting will be held between TC and the RAC. A project steering committee will be established to guide the project and provide an avenue for information sharing. The project steering committee will consist of representatives from TC, ECCC, NRCan, the RAC and rail industry representatives.

**D. LEM Report**

A consultation will occur with the Technical Review Committee to identify opportunities for streamlining the production of the LEM Report. The intent is to develop a proposal for the MOU Management Committee to consider in April.

**Action Item: Enrique Rosales to reach out to members of the Technical Review Committee to schedule a meeting.**

**Meeting adjourned at 2:04 pm EST.**

| New and Remaining Action Items – March 9, 2020 | | |
|---|---|---|
| **Item** | **Owner** | **Due Date** |
| **Regulatory Affairs Issues** | | |
| 1. Follow-up with ECC on *Cross-Border Movement of Hazardous Waste and Hazardous Recyclable Material Regulations* | Ben | Ongoing |
| 2. Write a letter to federal government representatives to discuss the rail industry's role in climate change mitigation. | MG | Seek clarification from EC |
| 3. Send out a proposal to confirm interest in meeting with the Ontario Minister of Environment in Q1 to discuss the rail industry's role in climate change mitigation. | MG | seek clarification from EC |
| 4. RAC to reach out to its multi-stakeholder group to determine if glyphosate should be part of the CEPA regulatory review. | Peter | End of November |
| 5. RAC to correspond with the AAR about glyphosate to determine if there are any next steps forward.<br><br>RAC to review federal Minister of Health mandate letter for relevance to glyphosate. | Ben | Complete |
| 6. RAC to contact ECCC for confirmation of the proposed timelines for the Clean Fuels Standard. | Ben | Complete |
| 7. RAC to conduct analysis on provincial CFS initiatives for impacts to rail sector and report back to the Environment Committee. | Ben | Work with Delphi – in progress |
| 8. RAC to reach out to other industry groups to promote the rail infrastructure funding opportunity in Quebec. | MG | Seek clarification from EC |
| 9. RAC to establish an EMS Working Group after completion of the EMS pilot at SRY. | Ben | Dependent on completion of SRY pilot. |
| 10. RAC to reach out to members of the Technical Review Committee to schedule a meeting to discuss opportunities for streamlining the LEM report. | ER / Ben | Complete |
| **Industry Programs and Studies** | | |
| 11. Distribute notes to fellow committee members following conferences and workshops. | ALL | Ongoing |
| **Ongoing Monitoring Files** | | |
| 12. Monarch butterfly conservation file | AK/EC | Ongoing |
| 13. Share with the Committee NRCan correspondence regarding LNG trucking infrastructure funding once NRCan responds. | MG | On hold as NRCan has yet to respond.<br><br>Seek clarification from committee |

# The Railway Association of Canada – Environmental Committee

## TERMS OF REFERENCE – (last updated November 2012)

### 1. Name
The organization shall be known as the Environmental Committee (hereinafter called the "Committee") of the Railway Association of Canada (RAC).

### 2. General Mandate
The Committee will serve as the focal point for sharing information and ideas that can enhance the environmental performance or sustainability of Canada's rail industry. It will identify and when required, analyze and evaluate environmental problems or issues concerning the RAC members and its affiliates.

The Committee will take appropriate action on the environmental matters referred to it by the other Committees or Subcommittees within the RAC, or as directed by the RAC Board of Directors.

The Committee will provide reports and or recommendations to the RAC Safety and Operations Management Committee (SOMC) on matters within the scope of its mandate and on any action to be taken with respect to those matters.

### 3. Membership
Membership is open to RAC staff and RAC member companies. Persons shall be nominated by their respective companies and shall commit to a two year period. Membership will be of special interest to those with responsibilities in the areas of environment and or sustainability. There shall be a minimum of five (5) members of this Committee.

A member who is unable to attend a meeting should send a representative who shall be counted in deterring a quorum and be permitted to vote in place of the regular member. Decisions will be made on a consensus basis with recommendations being made to SOMC that reflect this consensus. Non-consensus will be addressed by SOMC.

The Committee may select, as affiliate members, individuals whose competence and experience relates to the objectives of the Committee. Affiliate members shall not vote or hold elected office.

### 4. Officers
There shall be a Chairperson, Vice-Chairperson or Co-Chair, and a Secretary.

The Chairperson and Vice-Chairperson or Co-Chair shall be elected by the Committee members. The terms of office shall be two years.

Secretariat services will be provided by the RAC.

**5. Duties of Officers**

The Chairperson shall be responsible for the general supervision of the affairs of the Committee, preside at meetings, and:

a) Call regular meetings, as indicated in Section 8;
b) Be empowered to call special meetings if in his/her judgment, conditions warrant;
c) Appoint Special ad hoc committees or working groups; and
d) Be empowered to act during the interim between Committee meetings after consultation with the membership.

The Vice-Chairperson shall perform the duties of the Chairperson whenever the Chairperson is unable to perform the duties of his/her office. Additionally, the Vice-Chairperson shall perform other duties that may be assigned from time to time by the Chairperson.

The Secretary shall provide administrative and logistical support to the Committee. Responsibilities will include but will not be limited to scheduling meetings, compiling meeting minutes, and addressing specific requests put forward by the Chairperson.

**6. Duties of Committee**

This Committee shall:

a) Elect officers to the Committee every two years or as required;
b) Review the Terms of Reference every two years or as required;
c) Complete and improve an annual work plan that identifies priorities and action items to be addressed over the course of year;
d) Exchange information and data to promote the development of environmental programs within and best practices available to the railway industry;
e) Recommend research and implementation efforts into areas where attention should be directed;
f) Monitor environmental matters of relevant government and non-government entities including but not limited to: municipal, provincial, and federal government organizations, academia, other industry associations (e.g. the Association of American Railroads), and other relevant non-government and government entities; and
g) Consider and recommend positions to be taken on legislative or regulatory matters by the RAC, and formulate recommendations to be considered by SOMC when necessary.

**7. Meetings**

Regular meetings shall be held quarterly and additional meetings may be called by the Chairperson if circumstances requires. The time and place of the meeting with be designated by the Chairperson. Meetings may be held by teleconference at the discretion of the Committee members.

**8. Voting**

A motion shall require the approval of the majority of voting members.

A Committee member unable to attend a meeting may send a representation who shall be counted in determining a quorum and permitted to vote a place of regular member.

**SCHEDULE 4**

(Section 1 and paragraphs 80(1)(a), (c) and (d), (3)(a) and (4)(a))

# Movement Document for Movement Within Canada — Information Required

**1** The information that is required before a shipment is shipped from a consignor to a site that is owned, operated or otherwise controlled by a consignee is the following:

**(a)** the reference number that is provided for the movement document by the Minister or the province from which the shipment is to be shipped or to which it is to be delivered;

**(b)** the consignor's name, telephone number, email address, mailing address and registration number;

**(c)** the name and telephone number of the individual who signs the movement document;

**(d)** the civic address — or, if there is no civic address, the location — of the site from which the shipment is to be shipped;

**(e)** the consignee's name, telephone number, email address and registration number;

**(f)** the civic address — or, if there is no civic address, the location — of the site to which the shipment is to be delivered;

**(g)** for each type of hazardous waste or hazardous recyclable material that is recorded as a line entry in the movement document,

**(i)** the applicable UN number set out in column 1 of Schedule 1, or in column 3 of Schedule 3, to the *Transportation of Dangerous Goods Regulations*,

**(ii)** if no UN number applies to it under those Regulations, UN number 3082 when the hazardous waste or hazardous recyclable material is a liquid, UN number 3077 when it is a solid or either of those UN numbers when it is a sludge,

**(iii)** the shipping name and description set out in column 2 of Schedule 1, or the shipping name or technical name set out in column 1A of Schedule 3, to the *Transportation of Dangerous Goods Regulations* that is associated with the UN number referred to in that line entry,

**ANNEXE 4**

(article 1 et alinéas 80(1)a), c) et d), (3)a) et (4)a))

# Document de mouvement pour un mouvement au Canada — renseignements requis

**1** Les renseignements requis avant qu'un envoi soit expédié par un expéditeur à un site détenu, exploité ou contrôlé par un destinataire sont les suivants :

**a)** le numéro de référence attribué au document de mouvement par le ministre ou par la province à partir de laquelle l'envoi doit être expédié ou à laquelle il doit être livré;

**b)** les nom, numéro de téléphone, adresse de courriel, adresse postale et numéro d'enregistrement de l'expéditeur;

**c)** les nom et numéro de téléphone de la personne physique qui signe le document de mouvement;

**d)** l'adresse municipale ou, s'il n'y en a pas, l'emplacement du site à partir duquel l'envoi doit être expédié;

**e)** les nom, numéro de téléphone, adresse de courriel et numéro d'enregistrement du destinataire;

**f)** l'adresse municipale ou, s'il n'y en a pas, l'emplacement du site auquel l'envoi doit être livré;

**g)** à l'égard de chaque type de déchets dangereux ou de matières recyclables dangereuses inscrit sur une ligne de renseignements dans le document de mouvement :

**(i)** le numéro UN applicable prévu à la colonne 1 de l'annexe 1 du *Règlement sur le transport des marchandises dangereuses* ou à la colonne 3 de l'annexe 3 de ce règlement,

**(ii)** à défaut de numéro UN applicable aux termes de ce règlement, le numéro UN 3082 si le déchet dangereux ou la matière recyclable dangereuse est un liquide, le numéro UN 3077 s'il est un solide ou l'un ou l'autre de ces numéros UN s'il est de la boue,

**(iii)** l'appellation réglementaire et la description prévues à la colonne 2 de l'annexe 1 du *Règlement sur le transport des marchandises dangereuses*, ou l'appellation réglementaire ou technique prévue à la colonne 1A de l'annexe 3 de ce règlement, qui est associée au numéro UN visé à la ligne de renseignements,

**(iv)** the applicable class set out in column 3 of Schedule 1, or primary class set out in column 2 of Schedule 3, to the *Transportation of Dangerous Goods Regulations*,

**(v)** the following codes:

**(A)** the letter L when the hazardous waste or hazardous recyclable material is a liquid, the letter P when it is a sludge, the letter S when it is a solid and the letter G when it is a gas, followed in each case by the applicable code set out in column 1 of Part 2 of Schedule 11,

**(B)** the applicable codes set out in column 1 of Part 4 of Schedule 11,

**(C)** in the case of hazardous waste that is set out in column 2 of Schedule 6 or 12 or column 3 of Schedule 8 or that contains a substance that is set out in column 3 of Schedule 7, the applicable codes, if any, set out in column 1 of the applicable Schedule,

**(D)** in the case of hazardous waste or hazardous recyclable material that produces a leachate that contains one or more environmentally hazardous constituents, the applicable codes, if any, set out in column 1 of Schedule 2 for each environmentally hazardous constituent, and

**(E)** the applicable code, if any, set out in column 1 of Schedule 5 for every persistent organic pollutant that is contained in the hazardous waste or hazardous recyclable material,

**(vi)** the quantity of that type of hazardous waste or hazardous recyclable material, in kilograms or litres, to be shipped, and

**(vii)** the number of containers used, if applicable, and the code 01 when the container is a drum, 02 when it is a tank, 03 when the hazardous waste or hazardous recyclable material is shipped in bulk, 04 when the container is a carton, 05 when it is a bag, 06 when it is roll off or lugger and 07 in any other case; and

**(h)** the date on which the shipment is to be shipped and the date on which it is scheduled to be delivered.

**(iv)** la classe applicable prévue à la colonne 3 de l'annexe 1 du *Règlement sur le transport des marchandises dangereuses* ou la classe primaire applicable prévue à la colonne 2 de l'annexe 3 de ce règlement,

**(v)** les codes suivants :

**(A)** la lettre « L » si le déchet dangereux ou la matière recyclable dangereuse est un liquide, la lettre « P » s'il est de la boue, la lettre « S » s'il est un solide ou la lettre « G » s'il est un gaz, suivie dans chaque cas du code applicable prévu à la colonne 1 de la partie 2 de l'annexe 11,

**(B)** les codes applicables prévus à la colonne 1 de la partie 4 de l'annexe 11,

**(C)** dans le cas d'un déchet dangereux figurant dans la colonne 2 des annexes 6 ou 12 ou dans la colonne 3 de l'annexe 8 ou d'un déchet dangereux contenant une substance figurant dans la colonne 3 de l'annexe 7, les codes applicables, le cas échéant, prévus à la colonne 1 de l'annexe pertinente en cause,

**(D)** dans le cas d'un déchet dangereux ou d'une matière recyclable dangereuse qui produit un lixiviat contenant un ou plusieurs constituants dangereux pour l'environnement, les codes applicables, le cas échéant, prévus à la colonne 1 de l'annexe 2 pour chacun des constituants dangereux pour l'environnement,

**(E)** pour toute substance polluante organique persistante contenue dans le déchet dangereux ou la matière recyclable dangereuse, le code applicable, le cas échéant, prévu à la colonne 1 de l'annexe 5,

**(vi)** la quantité, exprimée en kilogrammes ou en litres, de ce type de déchets dangereux ou de matières recyclables dangereuses qui doit être expédiée,

**(vii)** le nombre de contenants, le cas échéant, et le code 01 si le contenant utilisé est un baril, 02 s'il s'agit d'une citerne, 03 si les déchets dangereux ou les matières recyclables dangereuses sont expédiés en vrac, 04 si le contenant utilisé est un carton, 05 s'il s'agit d'un sac, 06 s'il s'agit d'un conteneur sur châssis ou 07 s'il s'agit d'autre chose;

**h)** la date à laquelle l'envoi doit être expédié et la date prévue de sa livraison.

**2** The information that is required from an authorized carrier is the following:

**(a)** the authorized carrier's name, telephone number, email address, mailing address and registration number;

**2** Les renseignements requis de la part du transporteur agréé sont les suivants :

**a)** ses nom, numéro de téléphone, adresse de courriel, adresse postale et numéro d'enregistrement;

**(b)** the name and telephone number of the individual who signs the movement document; and

**(c)** a confirmation

**(i)** that the authorized carrier has received the shipment, and

**(ii)** if applicable, that they are to deliver the shipment to the site that is owned, operated or otherwise controlled by the consignee.

**3** The information that is required after the shipment is delivered is the following:

**(a)** the consignee's name, telephone number, email address, mailing address and registration number;

**(b)** the name and telephone number of the individual who signs the movement document;

**(c)** the civic address — or, if there is no civic address, the location — of the site to which the shipment was delivered;

**(d)** the date on which the shipment was delivered; and

**(e)** for each type of hazardous waste or hazardous recyclable material that is recorded as a line entry in the movement document, the quantity of the hazardous waste or hazardous recyclable material, in kilograms or litres, that was received.

**b)** les nom et numéro de téléphone de la personne physique qui appose sa signature sur le document de mouvement;

**c)** une confirmation :

**(i)** que le transporteur agréé a reçu l'envoi,

**(ii)** les cas échéant, qu'il livrera l'envoi au site détenu, exploité ou contrôlé par le destinataire.

**3** Les renseignements requis après la livraison de l'envoi sont les suivants :

**a)** les nom, numéro de téléphone, adresse de courriel, adresse postale et numéro d'enregistrement du destinataire;

**b)** les nom et numéro de téléphone de la personne physique qui appose sa signature sur le document de mouvement;

**c)** l'adresse municipale ou, s'il n'y en a pas, l'emplacement du site auquel l'envoi a été livré;

**d)** la date à laquelle l'envoi a été livré;

**e)** à l'égard de chaque type de déchets dangereux ou de matières recyclables dangereuses inscrit sur une ligne de renseignements dans le document de mouvement, la quantité, exprimée en kilogrammes ou en litres, de déchets dangereux ou de matières recyclables dangereuses reçue.

Climate Change and Carbon Pricing

The Supreme Court of Canada resumed hearings on September 22, 2020 based on constitutional challenges brought against the Government of Canada.

The Government of Alberta, Government of Ontario, and the Government of Saskatchewan are challenging whether *the Greenhouse Gas Pollution Pricing Act* constitutes a valid exercise of Parliament's jurisdiction for peace, order, and good government.

The RAC will continue to monitor progress and provide updates to the Environment Committee.

The table below provides a summary of provincial extensions to reporting deadlines due to the COVID-19 pandemic.

| **Annex – Summary of Updates to Carbon Pricing Schemes in Canada – as of September 11, 2020** | | |
|---|---|---|
| **Regulation, Act, or Program** | **Government** | **Change** |
| Greenhouse Gas Reporting Program | Federal | Reporting deadline moved to July 31, 2020 |
| Output-Based Pricing System (OBPS) | Federal | • The deadline for submitting annual and verification reports related to the 2019 compliance period was postponed from June 1, 2020 to October 1, 2020<br>• Regular rate compensation deadline under the OBPS for compliance year 2019 moved from December 15, 2020 to April 15, 2021<br>• Increased rate compensation deadline under the OBPS for compliance year 2019 moved from February 15, 2021 to June 15, 2021 |
| BC Carbon Tax | British Columbia | • The tax measure announced in Budget 2020 aligning the carbon tax rates with the federal carbon pricing backstop was originally schedule for April 1, 2020 and April 1, 2021 are now delayed a full year until April 1, 2021 and April 1, 2022.<br>• Carbon tax rates will remain at their current levels until March 31, 2021. |

| | | |
|---|---|---|
| Carbon Competitiveness Incentive Regulation (CCIR) | Alberta | • Submission deadlines for the following CCIR reports were extended from March 31, 2020 to June 30, 2020:<br>○ 2019 Compliance Report<br>○ Emissions Reduction Plan |
| Specified Gas Reporting Regulation | Alberta | • 2019 reporting requirement deadline extended from June 1, 2020 to July 31, 2020 |
| Methane Equivalency Agreement | Saskatchewan | • Equivalency agreement reached between Government of Saskatchewan and the Government of Canada.<br>• Over 10 years (January 1, 2020 to December 31, 2029), the federal regulations achieve higher methane emission reductions (29.21 Mt of CO2e for the Saskatchewan regulations and directives compared to 37.87 Mt of CO2e for the federal regulations). This means that the Saskatchewan regulations and directives will not be equivalent to the federal regulations.<br>• As the equivalency agreement between the Government of Canada and the Government of Saskatchewan terminates at the end of 2024, the Government of Saskatchewan will have to put in place additional regulatory measures in order for a new equivalency agreement to be concluded beyond 2024. |
| Made-in-Manitoba Carbon Price | Manitoba | • Implementation of provincial $25 tCO2e is indefinitely delayed<br>• Federal backstop implemented |
| | | |

| | | |
|---|---|---|
| Cap and Trade | Quebec | • Covered emitters were to submit the required information no later than July 31, 2020 |
| Gasoline and Motive Fuel Taxes | New Brunswick | • Reduction of gasoline and motive fuel taxes by 4.63 cents per litre and 6.05 cents per litre respectively |
| Cap and Trade | Nova Scotia | • Auction note for the 2020 auction was published and will go ahead as planned |

Environment and Climate Change Canada
Environnement et Changement climatique Canada

# A proposed integrated management approach to plastic products to prevent waste and pollution

DISCUSSION PAPER

Canada

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# Purpose

The Government of Canada is taking steps toward eliminating plastic pollution in Canada, including potentially banning or restricting certain harmful single-use plastic products, where warranted and supported by science. This discussion paper is seeking input on a proposed integrated management approach to plastics to take a number of actions, including regulations which would be developed under the provisions of the *Canadian Environmental Protection Act, 1999* (CEPA).

# Introduction

Plastic plays an important part in the lives of Canadians and in the Canadian economy, including in helping Canadians protect themselves from the spread of COVID-19. Plastic is low-cost, durable, and useful in a wide range of applications, including packaging, clothing, medical and personal protective equipment (PPE) and construction materials. However, the way plastic waste is managed in Canada is an issue of growing concern. According to a recent study conducted by Deloitte,[1] over 3 million tonnes of plastics were discarded as waste in Canada in 2016, and only 9% was recycled. Plastic waste burdens our economy, representing a $7.8B lost opportunity. When leaked into the natural environment, plastic threatens the health of our wildlife, ecosystems, rivers, lakes and oceans. In 2016, 29,000 tonnes of plastic waste entered the Canadian environment as pollution.

## Achieving zero plastic waste

Action is needed to eliminate plastic pollution at its source by reducing the amount of plastic waste that ends up in landfills or the environment. This can be achieved through greater prevention, collection, innovation and value recovery of plastic waste and transitioning to a more circular economy for plastics. The development and scaling up of new forms of plastic and new technologies provides opportunities to incentivize and support improved recovery of resources from products and packaging at the end of their useful life. Retaining materials and products in a circular economy not only reduces greenhouse gases emissions and pressure on the environment, but also has significant economic benefits. The transition to a more circular economy would save costs, increase competitiveness, stimulate innovation, support prosperity by creating new jobs and reduce the amount of plastic entering the environment.

Under Canada's G7 presidency in 2018, the Government of Canada championed the development of the Ocean Plastics Charter,[2] which commits to a more resource-efficient and lifecycle approach to plastics stewardship, on land and at sea. The Charter establishes targets to improve management of plastics, including:

- working with industry towards 100% reusable, recyclable, or, where viable alternatives do not exist, recoverable, plastics by 2030;

[1] *Economic Study of the Canadian Plastic Industry, Markets and Waste* (2019), available at: http://publications.gc.ca/collections/collection_2019/eccc/En4-366-1-2019-eng.pdf
[2] Available at: https://www.canada.ca/en/environment-climate-change/services/managing-reducing-waste/international-commitments/ocean-plastics-charter.html.

- working with industry towards increasing recycled content by at least 50% in plastic products where applicable by 2030;
- working with industry and other levels of government, to reuse and/or recycle at least 55% of plastic packaging by 2030 and recover 100% of all plastics by 2040; and
- working with industry towards reducing the use of microbeads in personal care products, and addressing other sources of microplastics.

In November 2018, through the Canadian Council of Ministers of the Environment (CCME), the federal, provincial and territorial governments approved in principle a Canada-wide Strategy on Zero Plastic Waste.[3] Building on the Ocean Plastics Charter, the strategy takes a circular economy approach to plastics and provides a framework for action in Canada. Federal, provincial and territorial governments are collaborating on implementing the Strategy via an Action Plan[4] by developing, among other things:

- guidance to facilitate consistent extended producer responsibility policies for plastics;
- national performance requirements and standards for plastics, including targets and timelines for increasing recycled content; and
- assessing infrastructure needs for improved plastic lifecycle management.

## Science assessment of plastic pollution

In October 2020, the Government of Canada released a Science Assessment of Plastic Pollution.[5] The Science Assessment presents a thorough scientific review of the occurrence and potential impacts of plastic pollution on human health and the environment. Information included in this assessment indicates that:

- plastic pollution, in both macroplastic and microplastic form, is everywhere in the environment;
- macroplastics have been shown to cause physical harm to individual animals and to have the potential to negatively affect the habitat of animals;
- exposure to macroplastics is not expected to be of concern for human health;
- the evidence is less clear and requires more research for potential effects of microplastics on individual animals and the environment; there is also limited information about the potential human health effects of microplastics, and while a concern for human health has not been identified at this time, further research is needed in this area; and
- there are a multitude of sources that contribute to plastic pollution

The Science Assessment recommends pursuing actions to reduce macroplastics and microplastics that end up in the environment, in accordance with 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".[6]

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[3] Available at: https://www.ccme.ca/en/resources/waste/waste/plastic-waste.html.
[4] *Ibid*.
[5] Available at: https://www.canada.ca/en/environment-climate-change/services/evaluating-existing-substances/science-assessment-plastic-pollution.html
[6] *Canadian Environmental Protection Act, 1999*, SC 1999, c 33, preamble.

## Managing plastics using CEPA

In order to take action as recommended in the Science Assessment, the Government of Canada has proposed using enabling authorities under CEPA to regulate certain plastic manufactured items[7]. This will allow the Government to enact regulations that target sources of plastic pollution and change behaviour at key stages in the lifecycle of plastic products, such as design, manufacture, use, disposal and recovery in order to reduce pollution and create the conditions for achieving a circular plastics economy.

## Rationale and objectives for an integrated management approach to plastics

Currently, Canada's large, complex and important plastics economy is mostly linear, which results in a significant amount of plastics waste being landfilled or released into the environment. The report prepared by Deloitte estimates that in 2016, 86% of plastic waste ended up in landfills, while 1% or 29,000 tonnes entered the environment as pollution.[8] Actions across the value chain or that promote innovation most likely will result in the systemic changes necessary to achieve zero plastic waste and eliminate plastic pollution.

While various governments, industry, scientists, civil society groups and others are working hard to move towards a circular plastics economy, a number of key challenges stand in the way. These include:

- **primary and secondary plastics compete:** competition is difficult for the recycling industry because of inconsistent feedstock composition and a more labour-intensive cost structure compared to primary resin production which can take advantage of economies of scale;
- **weak end-markets for recycled plastics**: in some cases, recycled resins are a cheaper alternative for product manufacturers, for example for use in less demanding applications, but overall the inconsistent supply of quality feedstock at a competitive price undermines the establishment of viable and lasting end-markets;
- **collection rates are low**: only 25% of plastics are collected and sent to a sorting facility (e.g., through curbside collection, recycling depots, or deposit-refund systems),[9] and only a fraction of collected plastics is recycled because of contamination, infrastructure deficiencies, and lack of markets;
- **insufficient recovery options**: current near absence of high volume recovery options, losses from existing processes, and competition from low cost disposal alternatives, such as landfills, point to the need for investments in innovation and infrastructure, in particular to commercialize and scale up new technologies; and
- **cost of plastic pollution is shouldered by individuals and communities**: the responsibility for preventing and managing land-based sources of plastic pollution, such as urban and

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[7] Available at: [LINK]
[8] *Supra* note 1.
[9] *Ibid*.

roadside litter, is largely shouldered by municipalities, civil society organizations and volunteers, at great cost.

No one measure can overcome these challenges. As part of its comprehensive agenda, the Government of Canada is developing an integrated management approach to plastics, which over time would seek to achieve the following objectives:

- **eliminate certain sources of plastic pollution**: reduce environmental harm caused by plastic products, in particular single-use plastics, by managing or, where necessary, prohibiting their use;
- **strengthen domestic end-markets for recycled plastics**: stimulate demand for recycled plastic that can drive the development of sustainable and resilient recycling markets and spur the investment in recovery infrastructure;
- **improve the value recovery of plastic products and packaging**: raise collection and recycling rates of plastic products and packaging, reduce the amount of plastic waste that ends up in landfills or the environment, and incentivize investment in infrastructure that can supply secondary end-markets with sufficient quantities of high-quality recycled plastics; and
- **support innovation and the scaling up of new technologies:** provide the incentives and regulatory space for businesses and researchers to develop, test and scale up technologies that help prevent plastic waste and pollution, such as new forms of plastic, new technologies for recovering value from plastic waste, and innovative business practices to improve the management of plastics throughout the value chain.

This integrated management approach to plastics will involve regulatory and non-regulatory actions. Non-regulatory instruments could be used by governments, industry and civil society to improve the management of plastics within their jurisdictions or control. Regulatory instruments are intended to ensure that rules are in place at key stages of the plastics lifecycle to drive the change necessary to achieve the objectives described above.

## Choosing the best instruments

A broad range of regulatory and non-regulatory instruments is available, allowing the government to choose the type of intervention.  A number of considerations factor into the choice of instrument or mix of instruments that are best suited to help achieve the management objective on a sustained basis while supporting innovation. These include environmental effectiveness, economic efficiency, health and safety, and distributional impacts across sectors, regions, and segments of the Canadian population.

Regulations and voluntary instruments (for example, guidelines) will be developed using CEPA or another effective mechanism. These will seek to:

- **manage single-use plastics**, including banning or restricting certain single-use plastics that cause harm, where warranted and supported by scientific evidence;
- **establish performance standards** for plastic products to reduce (or eliminate) their environmental impact and stimulate demand for recycled plastics; and
- **ensure end-of-life responsibility**, so that companies that manufacture or import plastic products or sell items with plastic packaging are responsible for collecting and recycling them.

These instruments and measures will be designed to complement each other as well as other policies, programs and actions implemented by federal, provincial, territorial and local governments. The success of one instrument will enhance the outcomes of all the others and contribute to achieving zero plastic waste. All instruments and measures are the subject of consultation and in-depth socio-economic analysis. A regulatory instrument is also always accompanied by a comprehensive Regulatory Impact Analysis Statement that is posted on the *Canada Gazette*, and which includes a cost-benefit analysis, as well as estimates of the administrative burden on regulated entities and impacts to small businesses.

## Roles and responsibilities

The integrated management approach to plastics proposed in this discussion paper recognizes that everyone has a role to play in achieving zero plastic waste and eliminating plastic pollution, including:

- **Government of Canada**: Environment and Climate Change Canada (ECCC), as well as other federal departments and agencies, will design regulatory instruments and other measures, work with other levels of government to avoid duplication, promote and ensure compliance, monitor outcomes, and be receptive to feedback in implementing programs, as well as monitor and work with other governments to address any trade implications.
- **provinces and territories**: the Government of Canada recognizes the leadership role provided by provinces and territories in developing, regulating and overseeing waste management systems, including recycling programs, and will support provincial and territorial governments in working to increase diversion rates for plastics, among other things.
- **local governments**: in response to provincial and territorial regulations, waste management services in Canada have traditionally been delivered or coordinated by cities, towns and regional authorities, which includes curbside or depot collection, sorting and separation operations, disposal facilities (landfills or incinerators), plus public education and promotion. Local authorities also deal with litter issues and street cleaning. In all cases, plastics waste is present and must be managed appropriately.
- **indigenous Peoples**: Indigenous peoples have an important role to play as traditional stewards of lands affected by plastic pollution, rights holders, and decision-makers for waste management issues in Indigenous communities, including on reserve land.
- **plastic producers and product manufacturers**: industry leadership and innovation is essential for better management of plastics. Producers of plastic resins and manufacturers of plastic products and packaging are best-placed to innovate and develop new solutions to address plastic waste in addition to meeting obligations established by regulatory instruments.
- **recyclers**: the Government of Canada will look to recyclers to support and enable systemic change in the plastic economy by effectively and efficiently recycling all the plastics collected and providing high-quality recyclable plastics to use as feedstock for new and innovative products.
- **Canadians**: all Canadians can do their part by reducing the amount of plastic waste they create, correctly sorting and binning recyclable plastics, and avoiding littering.

## Working with provinces and territories

The integrated management approach to plastics recognizes the central role played by provinces and territories in reducing plastic waste, eliminating plastic pollution and managing waste more generally. This is why the Government of Canada worked with its provincial and territorial counterparts in the CCME to develop the Canada-wide Strategy on Zero Plastic Waste. All jurisdictions must work together to drive the change necessary to move to a more circular economy for plastics across Canada. Among other things, a circular economy for plastics will:

- help businesses use resources and capital assets more efficiently;
- create new revenue streams through improved value recovery, and markets for new technologies and materials; and
- support the transition to a low-carbon economy by moving Canada away from linear models of resource use.

The Government of Canada will align measures developed under the integrated management approach to plastics with the guidance, standards and targets being developed in support of the CCME Strategy and Action Plan on Zero Plastic Waste.

Consideration of measures and programs already in place and complementarity with the roles of provincial, territorial and municipal governments will also be an important factor in the choice and design of instruments. The Government will work with its partners and stakeholders in Northern, remote and Indigenous communities to take into account their unique circumstances. Where appropriate, the Government of Canada will also seek agreements with provincial and territorial governments to minimize or eliminate duplication or overlapping rules.

# Managing single-use plastics

Canadians and businesses rely on single-use plastics and packaging for various purposes, from convenience to essential health and safety applications, and their use is increasing. Many of these plastic products are poorly managed at their end-of-life and have low recycling rates. Some single-use plastics that end up in the environment cause harm to ecosystems and wildlife, and those that are not recycled are a lost resource for the economy. The Government of Canada has committed to banning or restricting certain harmful single-use plastics, where warranted and supported by science.

## Scope

Single-use plastics have been be defined in recent work as "designed to be thrown away after being used only once".[10] These items include, among others:

- **packaging**: primary packaging (for example, food wrappers, retail product packaging, beverage and shampoo bottles), secondary or short lived packaging (for example, shopping bags, fruit & vegetable bags, containers), and sanitary packaging for sterile items (for example, syringes);
- **convenience items**: utensils, hot and cold drink cups and lids, straws, stir sticks, disposable wipes, and quick-serve containers; and
- **essential items**: masks and latex gloves in the dental and medical field, sterile packaging.

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[10] United Nations Environment Programme, *Single-use Plastics: a Roadmap for Sustainability: Fact-sheet for Policymakers* (2018), https://wedocs.unep.org/bitstream/handle/20.500.11822/25523/singleUsePlastic_sustainability_factsheet_EN.pdf

In addition to single-use plastics, there is a category of **short-lived disposable products or their components**, which includes pens, toothbrushes and their parts such as cotton swabs stems, cigarette butts and bottle caps.

The growing use of these items can present different challenges, such as:

- pollution in the environment and harm to wildlife through litter or accidental releases from commercial and industrial facilities or during transport;
- hampering of recycling, composting or wastewater treatment processes, due to small format, material choice and contamination; and
- inefficient use of material resources when cost-effective and low-impact alternatives are available.

Management of single-use plastics should also reflect the vital functions some single-use plastics play in keeping Canadians safe and healthy, assisting people with accessibility needs, and preserving food. For example, personal protective equipment includes some single-use plastics, such as masks and gloves. These are necessary to keep Canadians safe from the transmission of disease, in particular COVID-19. The Government of Canada will consider whether products that play vital roles such as these should be exempted from management measures, or whether measures should be designed to avoid limiting supply and accessibility (for example, by focusing on areas such as end-of-life management or litter prevention and clean-up) or stipulate acceptable alternatives.

The Government also recognizes the potential for new and innovative technologies to improve the environmental outcomes of some single-use products. For example, the use of compostable, bio-based or biodegradable plastics may in some cases improve a product’s environmental footprint or increase recovery rates of single-use items when they become waste. The Government will consider how the ban or the restriction on certain harmful single-use plastics might be designed to support the growth of new and innovative technologies that further the goals of environmental protection and the transition to a circular economy.

## Banning or restricting certain harmful single-use plastics as early as 2021

ECCC has conducted an analysis of available data to determine which items meet the requirements for a proposed ban or restriction. Sources of data include:

- Canadian citizen science and civil society data on which single-use plastics are most commonly found on Canadian beaches and shorelines;[11]
- ECCC-commissioned reports, Single-use Plastics in Canada (Cheminfo, 2018) and Economic Study of Canada’s Plastics Industry, Markets and Waste (Deloitte, 2019);
- sector-specific research on commonly used single-use plastics in Canada;
- work on single-use plastics prioritized for reduction actions by other jurisdictions within Canada; and
- work on single-use plastics prioritized for reduction by international organizations.

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[11] https://www.shorelinecleanup.ca/impact-visualized-data

In addition, while there is little data currently available on the plastic waste impacts of COVID-19, ECCC is aware of the potential increase in plastic waste and pollution caused by essential personal protective equipment.

Items were identified using the information sources above to provide a preliminary list of products that may be environmentally or value-recovery problematic, and which merited further analysis through a Management Framework for Single-use Plastics:

- Bags, including
  - checkout bags,
  - produce and bulk food barrier bags,
  - garbage bags, and
  - dry cleaning bags
- Packaging not necessary for the protection of food or goods, including:
  - multi-packaging,
  - produce stickers, and
  - some films
- Cosmetic and personal care products and packaging, including
  - cotton swab sticks
  - flushable wipes, and
  - disposable personal care items
- Plastic packaging used in aquaculture and coastal industries (for example., strapping bands)
- Food packaging, including:
  - beverage bottles and caps,
  - snack food wrappers, and
  - some films
- Food packaging and service ware (for example., takeout containers and lids, plates, bowls and cups) made from problematic plastics, including:
  - foamed plastics,
  - black plastic,
  - polyvinyl chloride (PVC),
  - oxo-degradable plastic, or
  - multiple (composite) materials including one or more plastics
- Coffee pods
- Plastics used in medical applications, including personal protective equipment such as:
  - masks,
  - gowns, and
  - gloves
- Cigarette filters
- Contact lenses and packaging
- Food service ware, including:
  - hot and cold drink cups and lids
  - straws
  - stir sticks
  - cutlery, and
  - condiment portion cups and sachets

The Management Framework for Single-use Plastics establishes a three-step process to determine if management is needed, and identifies the options for meeting management objectives:

*Management framework approach for single-use plastics*

| Steps | Details |
| --- | --- |
| **1. Categorize:** | Group single-use plastic items into categories and identify considerations for exemptions:<br>1. environmentally problematic<br>2. value recovery problematic |
| **2. Set management objectives:** | For priority categories, determine which objective in the waste management hierarchy should be pursued: (1) eliminate or reduce from the Canadian market, or (2) increase recycling or recovery rate. |

| Steps | Details |
| --- | --- |
| **3. Choose an instrument:** | Based on the objective chosen for each product, choose the appropriate instrument to achieve the goal informed by the *Instrument Choice Framework for Risk Management under the Canadian Environmental Protection Act*. |

**Step 1: Characterizing single-use plastics**

The first step is to categorize single-use plastics as environmentally problematic, value-recovery problematic, or both. In addition, considerations should be identified for possible exemptions to management action. This is done using the following criteria:

*Table 1: Criteria for the characterization of single-use plastics*

| Categories of single-use plastics | Criteria |
| --- | --- |
| **1) Environmentally problematic** | • Prevalent in natural and/or urban environments, according to citizen science, civil society and/or municipal litter audit data<br>• Known or suspected to cause environmental harm (for example., ingestion by wildlife or entanglement risk to wildlife, etc.) |
| **2) Value recovery problematic** | • Hampers recycling systems or wastewater treatment (nutrient or additive contamination, material or size/shape incompatible with recycling technology, etc.)<br>• Low to very low recycling rate (lower than average recycling rate for packaging, from 0-22%)<br>• Barriers to increasing their recycling rate exist |
| **Considerations for exemptions** | • Perform an essential function (for example., accessibility, health and safety, security)<br>• No viable alternative exists that can serve the same function<br>• Specification of acceptable & available alternative material |

A single-use plastic can be considered environmentally problematic and/or value-recovery problematic if it meets the criteria in the above table. Table 2 illustrates how ECCC categorized select single-use plastics, drawing from the best available information listed above:

*Table 2: Analysis of information of selected single-use plastic products*

| | **Environmentally problematic** | | **Value recovery problematic** | | | **Exemption considerations** | |
| --- | --- | --- | --- | --- | --- | --- | --- |
| | **Prevalent in environment** | **Known or suspected to cause environmental harm** | **Hampers recycling and/or wastewater treatment** | **Non-recyclable, low or very low recycling rate** | **Barriers to increasing recycling rate** | **Performs essential function** | **No viable alternatives** |
| Plastic checkout bags | ✓ | ✓ | ✓ | ✓ | ✓ | | |
| Stir sticks | ✓ | ✓ | ✓ | ✓ | ✓ | | |
| Six-pack rings | ✓ | ✓ | ✓ | ✓ | ✓ | | |

| | Environmentally problematic | | Value recovery problematic | | | Exemption considerations | |
|---|---|---|---|---|---|---|---|
| | Prevalent in environment | Known or suspected to cause environmental harm | Hampers recycling and/or wastewater treatment | Non-recyclable, low or very low recycling rate | Barriers to increasing recycling rate | Performs essential function | No viable alternatives |
| Cutlery | ✓ | ✓ | ✓ | ✓ | ✓ | In some cases, for security | |
| Straws | ✓ | ✓ | ✓ | ✓ | ✓ | In some cases, for accessibility | |
| Food packaging and service ware made from problematic plastics | ✓ | ✓ | ✓ | ✓ | ✓ | | |
| | | | | | | | |
| Other bags (for example., garbage) | | | ✓ | ✓ | ✓ | | |
| Snack food wrappers | Some kinds | | Some kinds (for example., bioplastics) | ✓ | ✓ | ✓ | |
| Multi-packaging | | | ✓ | ✓ | ✓ | | |
| Disposable personal care items | | | ✓ | ✓ | ✓ | | |
| Beverage bottles and caps | ✓ | ✓ | | | | | |
| Contact lenses and packaging | ✓ | | | ✓ | ✓ | ✓ | ✓ |
| Hot and cold drink cups and lids | ✓ | | ✓ | ✓ | ✓ | | |
| Cigarette filters | ✓ | ✓ | | ✓ | ✓ | | ✓ |

### Step 2: Setting management objectives

The proposed environmental objectives of the Management Framework for Single-use Plastics are to:

1) eliminate or significantly reduce single-use plastics entering Canada's environment;
2) reduce the environmental impact of plastic products overall; and
3) conserve material resources by increasing the value recovery of plastics.

### Step 3: Instrument choice

When there are multiple possible actions to achieve the management objectives, the *Instrument Choice Framework for Risk Management under the Canadian Environmental Protection Act* will inform the selection of appropriate instruments. The Instrument Choice Framework uses several criteria to guide these decisions:

1. environmental effectiveness and the achievement of the management objective;
2. economic efficiency including minimizing costs and maximizing benefits;
3. distributional impacts on groups and segments of society;
4. acceptability and compatibility, including stakeholder acceptability and compatibility with other programs in Canadian jurisdictions; and
5. international obligations, with a focus on international protocols and agreements as well as trade obligations.

The Government of Canada has committed to ban or restrict certain harmful single-use plastic items, where warranted and supported by science. This means that:

- for products to be considered “harmful” and for a ban or a restriction to be considered “warranted”, the criteria for both environmentally problematic and value recovery must be met;
- assessing a single-use plastic item using these criteria requires scientific evidence of both environmental prevalence and value recovery challenges; and
- in cases where a product meets all criteria but performs an essential function, exemptions to a ban or a restriction may be recommended in some cases.

Table 3 illustrates how the Management Framework for Single-use Plastics can be applied to choose instruments appropriate to meeting management objectives.

*Table 3: Proposed instruments and the scope of their potential application*

<table>
<tr><th rowspan="2"></th><th colspan="2">Management Objective:<br>Eliminate or reduce from the Canadian market, or restrict use</th><th colspan="2">Management Objective:<br>Increase recycling / recovery rate of single-use plastics and packaging</th></tr>
<tr><th>CEPA instruments:<br>Ban, restrictions in use</th><th>Instruments:<br>Incentives to encourage reusable products or systems</th><th>Instruments:<br>Material specifications (for example., recyclable)</th><th>Instruments:<br>Extended producer responsibility or other collection, recycling requirements</th></tr>
<tr><td>Environmentally problematic</td><td rowspan="2">• Plastic Checkout Bags<br>• Stir sticks<br>• Six-pack rings<br>• Food service ware made from problematic plastics<br>• Straws<br>• Cutlery</td><td>• Food service ware</td><td>• Hot and cold drink cups and lids</td><td>• Beverage bottles and caps<br>• Cigarette filters</td></tr>
<tr><td>Value recovery problematic</td><td>• Personal care product bottles<br>• Hot and cold drink cups and lids</td><td>• Food wrappers<br>• Other bags (for example., garbage)<br>• Multi-packaging</td><td>• Disposable personal care items</td></tr>
</table>

The analysis above generated **six plastic items that meet the requirements of a ban or a restriction**, supported by sufficient scientific evidence, data gathered from the Great Canadian Shoreline Cleanup and socio-economic considerations:

*Table 4: Single-use plastic items that meet the requirements for a ban*

| **Certain single-use plastic items being considered for a ban or a restriction:** |
| --- |
| • plastic checkout bags<br>• stir sticks<br>• six-pack rings<br>• cutlery<br>• straws<br>• food service ware made from problematic plastics |

For other single-use plastics, currently available data on the use, management and prevalence in the environment do not support a recommendation for a ban or a restriction at this time. The results of additional information gathering and consultations, as well as further analysis using the proposed Management Framework for Single-use Plastics, will indicate whether management action is needed and which measure should be considered.

The Government of Canada will continue to work with provinces, territories, industry and other stakeholders to implement this framework over time. How measures are chosen, designed and implemented will take into account factors such as best-placed jurisdiction, the potential for voluntary agreements and other industry-led actions, and the *Instrument Choice Framework for Risk Management under the Canadian Environmental Protection Act*. They will also be the subject of consultation and in-depth socio-economic analysis. A regulatory instrument is also always accompanied by a comprehensive Regulatory Impact Analysis Statement that is posted on the Canada Gazette. As a first step in this process, ECCC welcomes comments on the categorization and the proposed management approach described here.

# Establishing performance standards

The proliferation of different types of plastics, formats, labelling, collection schemes and processing technologies together impede the transformation of waste plastics into materials that are cost-competitive with primary materials.  This, in turn, hampers the establishment of viable markets for secondary and alternative materials. The introduction of new products across value-chains outpaces the deployment of regulations or programs to ensure collection and new technologies to process the growing variety of plastic products on the market. Recyclers need certainty that there will be buyers for the plastic they recycle to secure investments. To begin addressing some of these issues, the Government of Canada is considering how product performance standards for plastic products and packaging can contribute to generating a sufficient, stable and predictable supply of materials in order to support viable secondary plastics markets and investments in the recovery infrastructure in Canada.

## Recycled content requirements

Recycled content requirements establish a market demand for recycled plastics  which lessens the pressures for recyclers to compete with the cost of virgin resin. Robust domestic demand for recycled plastics would also drive investments in recycling operations, innovations in material separation and technologies, and opportunities to scale up emerging technologies. Recycled content requirements can also spur companies to reconsider the design of their products. The use of recycled plastics delivers environmental benefits, such as extending the life of some resins and reducing greenhouse gas emissions, and contributes to the transition to a circular economy.

Recognizing the importance of recycled content requirements to drive demand for these markets, the Government of Canada has adopted a target of at least 50% recycled content in plastic products by 2030. As part of Phase 1 of the Canada-wide Action Plan on Zero Plastic Waste, the CCME supported this objective and further committed to establishing targets and timelines for increasing recycled content.[12]

Many leading companies are including recycled content in their plastic products and have made voluntary commitments to recycled content performance targets. To further support the development of secondary markets for recycled plastics, the Government of Canada is proposing regulations using CEPA to require recycled content in plastic products and packaging. Regulations and accompanying guidance will establish:

- **a minimum percentage of recycled content** as an outcome-based requirement that producers would need to meet to comply with the regulations;
- **rules for measuring and reporting** to evaluate a product's conformity with claims of recycled content; and
- **technical guidelines and related tools** to help companies meet their requirements, such as standards, specifications and terminologies.

The approach for requiring recycled content is under development. Options considered could be based on:

- **resin**: establish recycled content targets and requirements by resin type;
- **product or sector grouping**: establish recycled content targets and requirements by product category (for example., rigid containers, film packaging) or sector (for example., packaging, electronics); or
- **economy-wide**: establish an economy-wide recycled content target/requirements for plastic products without differentiating between sectors, products or resin types.

In addition, the approach as well as the selection of interim targets and timelines for recycled content requirements will recognize the current technical and regulatory barriers that must be considered when incorporating recycled plastics into new products and packaging. For example, food chemical safety is a consideration when using recycled plastics in food packaging. The use of recycled plastics, as with any other plastic material, in food packaging applications must comply with the safety provisions of the *Food and Drugs Act* and associated regulations. Any other existing requirements in laws and regulations related to product performance (for example., energy efficiency or consumer safety) would also still apply. Factors affecting the ability of recycled plastics to meet performance requirements include the quality of the feedstock, technologies and processing methods, and appropriate performance standards and test methods.

The approach for measuring and reporting on recycled content in products is also under development. Voluntary standards are currently used by industry and some new ones are being developed. Key issues to consider for measurement and reporting include, among others:

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[12] *Supra* note 3.

- **definitions of recycled content**, and the potential applicability of different types (for example., post-consumer resin, pre-consumer resin) in meeting performance standards;
- **method of tracking chain-of-custody**, for example., certifications generated by recyclers based on the mass-balance of material flowing through recycling facilities; and
- **flexibility** in meeting performance standards, for example., applying recycled content requirements on an individual product basis or on an average across a company's product line.

Regulatory approaches to ensuring recycled content performance standards are met, such as reporting protocols and open data rules to create accountability and ensure compliance through transparent information, will be considered.

# Ensuring end-of-life responsibility

As part of the integrated management approach to plastics, the Government of Canada is working to extend the life and improve the value recovery of plastic products and packaging. This means

- raising collection, repair and recycling rates;
- minimizing the amount of plastic sent to landfill;
- bringing more product categories under management frameworks across the country; and
- establishing the conditions for innovation and greater capacity throughout Canada to create a circular economy for plastics and stimulate investments in critical collection and recovery infrastructure.

## Improving and expanding extended producer responsibility in Canada

The Government of Canada has committed to working with provinces and territories to develop consistent, national targets, standards and regulations that will make companies that manufacture plastic products or sell items with plastic packaging responsible for collecting and recycling them. This is known as extended producer responsibility. Federal, provincial and territorial governments agree that extended producer responsibility is one of the most effective and efficient ways of increasing collection and recycling rates and is a cornerstone to achieving our Canada-wide objective of zero plastic waste.

Provinces and territories are taking the lead by developing and implementing extended producer responsibility systems within their jurisdictions. To maximize the recovery of plastic products and packaging, the Government of Canada will work with provinces, territories and industry to advance extended producer responsibility across Canada that is:

- **consistent**: rules need to be consistent across jurisdictions to create a level playing field, reduce administrative burden and allow companies to take advantage of the efficiencies and economies of scale possible in larger markets that transcend provincial and territorial borders;
- **comprehensive**: to help achieve zero plastic waste, extended producer responsibility should extend to all major sectors of the Canadian plastics economy that generate large amounts of plastic waste; and

- **transparent**: companies are made responsible for meeting outcomes such as collection targets, but are given the freedom to decide how best to meet those targets, making accountability dependent on the transparent reporting of key data.

As part of Phase 1 of the CCME's Action Plan on Zero Plastic Waste,[13] the Government of Canada is working with provincial and territorial governments to develop national guidance that will facilitate consistent, comprehensive and transparent extended producer responsibility policies for plastics. This guidance will include:

- common material categories and product definitions;
- performance standards to guide reuse and recycling programs;
- options to encourage innovation and reduce costs; and
- standard monitoring and verification approaches.

The Government of Canada will support provincial and territorial governments as they work to harmonize their extended producer responsibility systems. This will include exploring with provinces and territories how gaps and inconsistencies can be addressed, including through national actions.

# Next steps and sending comments

The Government recognizes the importance of balancing environmental protection and clean growth with the economic importance of plastic and its role in protecting human health, in particular during this COVID-19 public health emergency.

Taking into account lessons from the current pandemic and mindful of continued constraints brought about by the pandemic, Canadians and Canadian businesses will be given the opportunity to participate meaningfully in informing any measures taken.

Next steps for ECCC will include engagement with provincial and territorial governments, Indigenous Peoples and stakeholders on the design of the regulatory instruments and the approaches outlined in this discussion paper.

Parties wishing to comment on any aspect of this paper, including the categorization of single-use plastics and proposed management approaches, are invited to provide written comments to the Director of the Plastics and Marine Litter Division of ECCC by December 9, 2020 at ec.plastiques-plastics.ec@canada.ca.

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[13] *Supra* note 3.

# Questions for discussion

The Government is seeking input to inform the design and implementation of the proposals described in this discussion paper. Businesses, civil society groups, jurisdictions, Indigenous Peoples, and all Canadians are invited to provide their perspectives, expertise and opinions. To help focus input, the Government invites commenters to consider the following questions. Other comments and suggestions related to anything described in this discussion paper are also welcome.

## Managing single-use plastics

1. Are there any other sources of data or other evidence that could help inform the development of the regulations to ban or restrict certain harmful single-use plastics?
2. Would banning or restricting any of the six single-use plastics identified impact the health or safety of any communities or segments of Canadian society?
3. How can the Government best reflect the needs of people with disabilities in its actions to ban or restrict certain harmful single-use plastics?
4. Should innovative or non-conventional plastics, such as compostable, bio-based or biodegradable plastics be exempted from a ban or a restriction on certain harmful single-use plastics? If so, what should be considered in developing an exemption that maintains the objectives of environmental protection and fostering a circular economy for plastics?

## Establishing performance standards

5. What minimum percentage of recycled content in plastic products would make a meaningful impact on secondary (recycled resin) markets?
6. For which resins, products, and/or sectors would minimum recycled content requirements make the greatest positive impact on secondary (recycled resin) markets? Why?
7. Which resins, products or sectors are best-placed to increase the use of recycled plastic and why?
8. Which plastic products are not suitable for using recycled content due to health, safety, regulatory, technical or other concerns?
9. What should be considered in developing timelines for minimum recycled content requirements in different products?
10. What would be the advantages and disadvantages to setting minimum percentage requirements that are distinct for each product grouping, sector, and/or resin?
11. How could compliance with minimum recycled content requirements be verified? How can the Government and industry take advantage of innovative technologies or business practices to improve accuracy of verification while minimizing the administrative burden on companies?
12. Besides minimum recycled content requirements, what additional actions by the government could incentivize the use of recycled content in plastic products?

## Ensuring end-of-life responsibility

13. How can the Government of Canada best support provinces and territories in making their extended producer responsibility policies consistent, comprehensive, and transparent?

Government of Canada Gouvernement du Canada

# Science Assessment of Plastic Pollution

Environment and Climate Change Canada
Health Canada

October 2020

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Aussi disponible en français

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## List of abbreviations

| Abbreviation | Meaning |
|---|---|
| μ-FTIR | Micro-Fourier transform infrared spectroscopy |
| AAB | Adopt-a-BeachTM |
| 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.

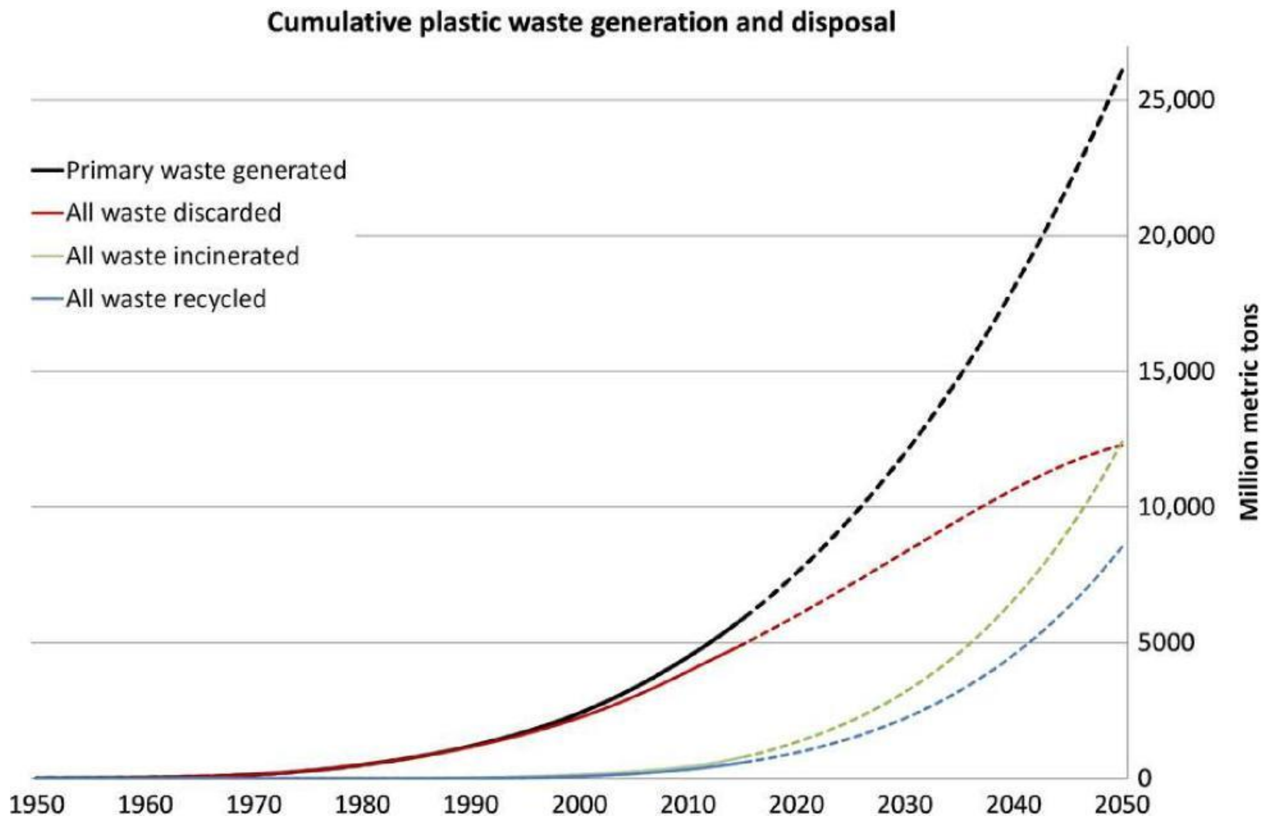


***Figure 1: Global cumulative plastic waste generation and disposal. Solid lines show historical data from 1950 to 2015, and dashed lines show projections of historical trends to 2050 (reproduced with permission from Figure 3 of Geyer et al. 2017)***

**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).

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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<br>Automotive<br>Housewares<br>Electrical insulation |
| PE | Polyethylene | Rigid, semi-rigid and flexible packaging<br>Agricultural film<br>Housewares<br>Electrical insulation<br>Construction (pipes)<br>Self-care products |
| PS | Polystyrene | Packaging (thermoformed containers)<br>Foams |
| PMMA | Poly(methyl methacrylate) | Transparent applications in the automotive and construction industries<br>Medical<br>Electronics |
| PC | Polycarbonate | Transparent applications in the automotive and construction industries<br>Medical<br>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<br>Textile synthetic fibres |
| PVC | Polyvinylchloride | Construction (pipes, profiles, flooring)<br>Sheet and coated fabrics<br>Electrical insulation |
| PTFE | Polytetrafluoroethylene | Anti-adhesive coatings<br>Engineering parts |

[a] Personal communication, email from the Transportation and Manufacturing Oivision, National Research Council Canada, to the Ecological Assessment Oivision, 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 (LOPE), linear low-density polyethylene (LLOPE), and high-density polyethylene (HOPE). These types of PE form the majority of PE production globally. Some examples of LOPE/LLOPE applications are squeeze bottles, toys, carrier bags, and general packaging. LOPE is generally used in heavier duty films, such as high durability bags and protective sheeting, due to its toughness, flexibility, and relative transparency. HOPE 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).

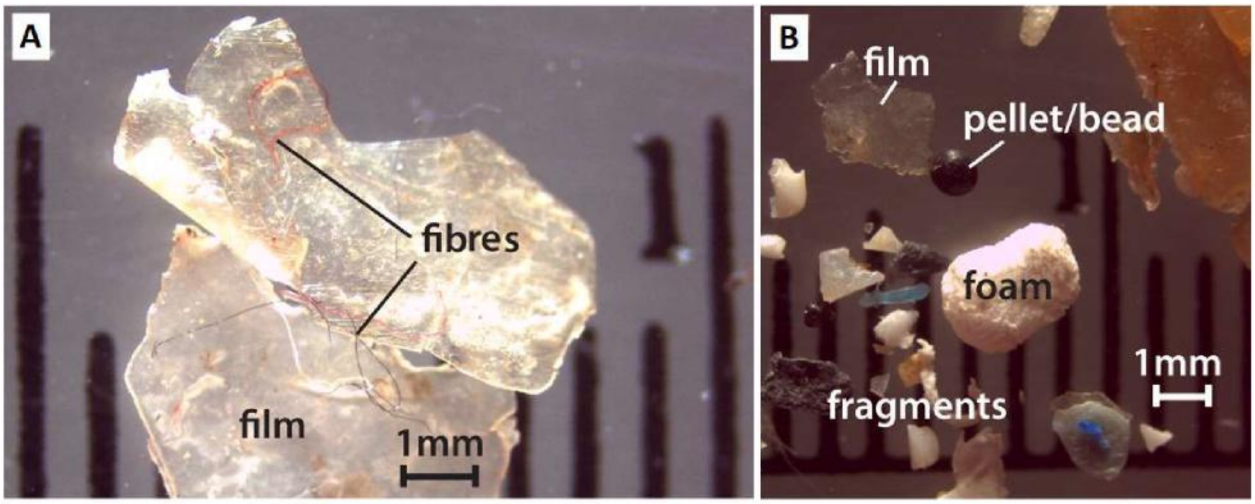


*Figure 2: Microplastics found in the environment (reproduced and adapted with permission from Figure 2 of Baldwin et al. 2016)*

**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/cm³)[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).

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[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).

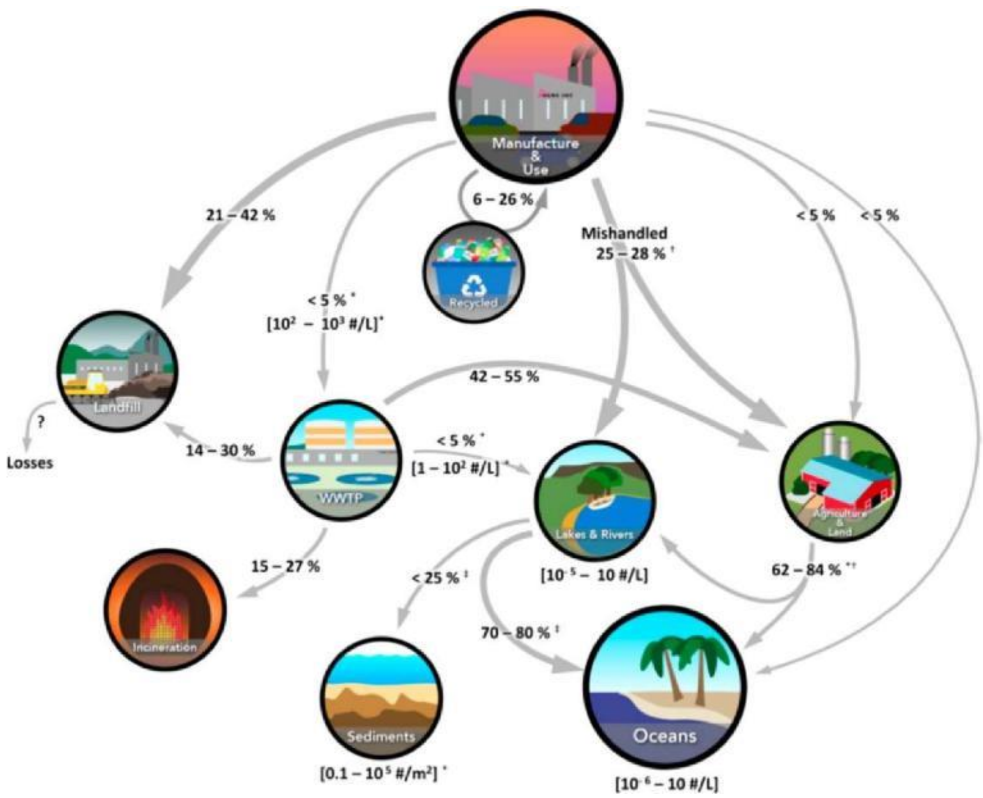


***Figure 3: Estimated plastic loading and transport pathways in the environment (reproduced with permission from Alimi et al. 2018, © 2018 American Chemical Society).***

**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

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[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])

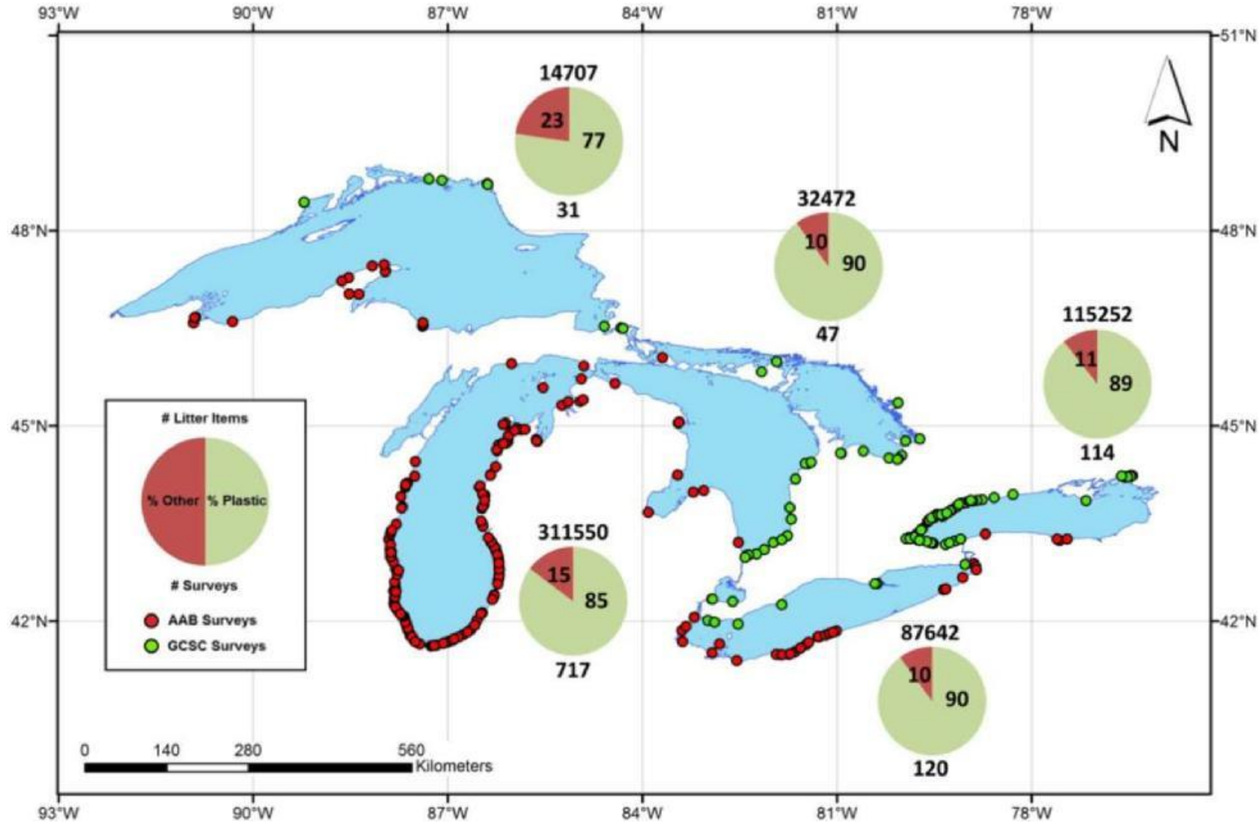


*Figure 4: Great Lakes beach cleanup surveys (reproduced with permission from Figure 2 of Driedger et al. 2015)*

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-BeachTM (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 nonspecific 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 $7 \times 10^{3}$ particles/L and a reported mean of $0.7 \times 10^{-3}$ 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 nonnative 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 x $10^{13}$ 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):583586.

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. 186:1016.

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):8189.

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. Longterm 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(12):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. 251:117127.

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):66346647.

[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.<br><br>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<br>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.<br><br>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 |
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| | 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.<br><br>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.<br><br>A previous study reported microplastics in the digestive tract, head, and gills of shrimp (*Crangon crangon*) collected from the Sea (mean: 1.23 microplastics per shrimp), but not in the abdominal muscle tissue (Devriese et al. 2015).<br><br>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.<br><br>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 |
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| | 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.<br><br>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.<br><br>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).<br><br>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.<br><br>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 |
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| | under conditions that mimicked the tea steeping process.<br><br>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.<br><br>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.<br><br>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).<br><br>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.<br><br>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 |
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| | with mainly fibres detected (range: 70 to 47 40 rim; mean: 850 rim). | |
| 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.<br><br>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<br><br>Bottled water | This study investigated the effects of mechanical stress on the generation of microplastics (≥3 rim) in single-use PET plastic water bottles.<br><br>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.<br><br>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 rim 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 |
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| | 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.<br><br>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 |
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| | 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.<br><br>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%.<br><br>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.<br><br>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 |
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| | 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 |

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| 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 LOPE were egested by the earthworms.<br><br>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:<br>• Sea cucumbers (*Holothuria tubulosa*)<br>• *Alepisaurus ferox Lowe*<br>• *Nephrops norvegicus*<br>• *Zostera marina L.* bed<br>• Common cockle (*Cerastoderma edule*)<br>• 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 |

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| | 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.<br><br>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.<br><br>In the colour experiment, green and black microplastics were ingested at a significantly higher rate than other colours.<br><br>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<br><br>6.2.2. Ecotoxicological effects<br><br>Sediment | This study investigates the uptake and effects of 1 to 5 mm PS fibres on the sea cucumber *Apostichopus japonicus*.<br><br>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.<br><br>For uptake experiments from water, *A. japonicus* were exposed to microplastics in | Mohsen et al. 2020 |

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| | 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.<br><br>There were no significant effects on the velocity or distance travelled by treated *A. japonicus*.<br><br>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<br><br>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.<br><br>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<br><br>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 x $10^5$ particles/L. Microplastics were ingested by larvae and adults throughout the 48-hour exposure, and | Cong et al. 2019 |

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| | 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.<br><br>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<br><br>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.<br><br>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.<br><br>In all cases, increased concentration of microplastics magnified the histopathological alterations in sea bass. | Espinosa et al. 2019 |
| 6.2.2 Ecotoxicological effects<br><br>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.<br><br>Early hatching was observed for embryos under static conditions and survival rate was lower compared to controls for all concentrations. | Malafaia et al. 2020 |

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| | 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<br><br>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*).<br><br>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.<br><br>Fibres and fragments caused the most severe effects in gut microbiota. | Qiao et al. 2019b |
| 6.2.2. Ecotoxicological effects<br><br>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<br><br>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 |

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| 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<br><br>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<br><br>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.<br><br>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<br><br>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 |

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| | 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<br>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<br>Soil | The study exposed the earthworm *Aporrectodea rosea* to HOPE microplastics in soil. *A. rosea* exposed to soil with HOPE lost significantly more biomass than controls. | Boots et al. 2019 |
| 6.2.2 Ecotoxicological effects<br>Soil | Several studies investigated the effects of microplastics on terrestrial plants. Oe Souza Machado et al. (2019) found that microplastics altered biomass and root traits in *Allium fistulosum*. Polyester, PS, HOPE, PET and PE increased root biomass, whereas PA decreased dry biomass. Contrary to Oe Souza Machado et al. (2019) results for HOPE, Boots et al. (2019) found that HOPE microplastics decreased root biomass in *Lolium perenne*, as well as decreased germination and shoot length. | Boots et al. 2019; Oe Souza Machado et al. 2019 |
| 6.2.2 Ecotoxicological effects<br>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 |

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| | communities or related ecosystem functions and processes. | |
| 6.2.2 Ecotoxicological effects<br><br>Soil | The study investigated defence responses in earthworms exposed to microplastics. ***E. fetida*** were exposed to LOPE 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 LOPE after 28-day exposure. | Chen Y et al. 2020 |
| 6.2.2 Ecotoxicological effects<br><br>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<br><br>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<br><br>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 |

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| | dw. There was also noticeable tissue damage in the stomachs and intestines of snails exposed to microfibres. | |
| 6.2.2. Ecotoxicological effects<br><br>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<br><br>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 |

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| | 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.<br><br>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

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| 7.2.1 Effects from oral exposure<br><br>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).<br><br>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.<br><br>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<br><br>Studies in experimental animals | This study evaluated the effects of maternal microplastic exposure on dams and offspring (F1 and F2 generation).<br><br>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 |

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| | 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.<br><br>The study also reported transcriptomic and metabolomic effects; however, these findings are unclear and require further research. | |
| 7.2.1 Effects from oral exposure<br><br>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.<br><br>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.<br><br>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<br><br>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).<br><br>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.<br><br>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 |

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| | 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.<br><br>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<br><br>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.<br><br>All stool samples tested positive for microplastics, with PP and PET being the most commonly detected polymers.<br><br>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<br><br>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).<br><br>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 |
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| | 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 |

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| | 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

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| 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 |

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| | 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.<br><br>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.*<br><br>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.<br><br>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.<br><br>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.<br><br>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.<br><br>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.<br><br>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.<br><br>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 $1.1 \times 10^6$ particles/m$^3$ to $1.2 \times 10^7$ 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 $6.4 \times 10^2$ to $4.2 \times 10^4$ 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)<br><br>For uptake experiments, exposure was 15, 30, 60, 120 and 240 minutes<br><br>For depuration experiments, exposure was 1 hour<br><br>For differential food regime experiments, exposure was 1 hour<br><br>For chronic toxicity tests, exposure was 21 days | Yellow-green fluorescent, carboxylate-modified PS (2 µm) were used for uptake and depuration experiments<br><br>Non-fluorescent PS microplastics (2 µm) were used for toxicity tests<br><br>For uptake and depuration experiments, microplastic concentration was $1.46 \times 10^{2}$ mg/L and algae concentration was $1.00 \times 10^{-1}$ mg/L<br><br>For differential food regime experiments, microplastic concentrations were $6.93 \times 10^{-4}$, $1.39 \times 10^{-3}$, $2.77 \times 10^{-3}$, $5.54 \times 10^{-3}$, $8.31 \times 10^{-3}$, and $1.11 \times 10^{-2}$ mg/L; algae concentrations were $5.00 \times 10^{-2}$, $1.00 \times 10^{-1}$, $2.00 \times 10^{-1}$, $4.00 \times 10^{-1}$, $6.00 \times 10^{-1}$, $8.00 \times 10^{-1}$ mg/L<br><br>Control groups for uptake, depuration, and differential food regime experiments were not exposed to algae<br><br>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.<br><br>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 ($1.00 \times 10^{-1}$ mg/L) with a relatively higher microplastic concentration ($1.11 \times 10^{-2}$ 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.<br><br>In neonate *D. magna*, mortality in those fed a low algal concentration ($1.00 \times 10^{1}$ mg/L) and microplastics was significantly higher than neonates fed only algae. No effect on mortality was found for a high algal concentration ($8.00 \times 10^{-1}$ mg/L) and microplastic uptake. There were no significant differences in reproduction between identical food regimes with and without microplastics. | Aljaibachi and Callaghan 2018 |

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|---|---|---|---|
| | × $10^{-3}$ 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<br><br>Control group for chronic toxicity tests was not exposed to microplastic | | |
| *Xenopus laevis* (African clawed frog) tadpoles<br>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)<br><br>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.<br><br>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)<br>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)<br><br>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<br><br>Control group was given food pellets that contained no microplastic | Various sublethal effects, but no mortality, were observed.<br><br>Fish exposed to plastic fibres, fragments, and pellets showed significant weight loss compared to the control group.<br><br>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 |

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| | | 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.<br><br>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*<br><br>*Daphnia pulex*<br><br>*Ceriodaphnia dubia*<br><br>(Water fleas)<br><br>96 hours | Green fluorescent plastic microspheres (1–5 µm) were used as primary microplastic models<br><br>Irregularly-shaped PE microplastics (approx. 1–10 µm) were used as secondary microplastic models<br><br>Concentrations were $10^3$, $10^4$, $10^5$, $10^6$, $10^7$ particles/mL<br><br>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*.<br><br>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.<br><br>Primary microplastics were found to be more toxic than secondary microplastics in *C. dubia*.<br>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 |

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|---|---|---|---|
| 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*<br><br>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<br><br>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)<br><br>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<br><br>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.<br><br>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.<br><br>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)<br><br>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 |

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| Four sequential generations | Control group was exposed to a clean test medium | mobile juveniles), and population growth rate.<br><br>In two treatment groups, microplastic-exposed populations were extinct in the $F_1$ (2nd) generation. Juveniles produced by microplastic-exposed females were immobile.<br><br>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)<br><br>14 and 21 days | Fluorescent red microspheres (1–5 μm) at 0.02 mg/L and 0.2 mg/L (nominal)<br><br>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.<br><br>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.<br><br>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 |

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| | | 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)<br><br>For distribution experiments, exposure times were 20 hours (4–24 hpf[c]) and 92 hours (4–96 hpf)<br><br>For uptake and qPCR[d] experiments, exposure was 92 hours (4–96 hpf)<br><br>For developmental effects experiments, exposure was 68 hours (4–72 hpf)<br><br>For free swimming and light-to-dark experiments, exposure was 116 hours (4–120 hpf) | Green fluorescent PS microplastics (1 µm)<br><br>For distribution, developmental effects, free swimming, light-to-dark, and qPCR analysis, concentrations used were 100 µg/L and 1 000 µg/L<br><br>For uptake experiments, concentrations used were 10, 100 and 1 000 µg/L<br><br>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.<br><br>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.<br><br>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.<br><br>In the 1 000 µg/L exposure group, *il1b* and *cat* expression were upregulated to 165% and 121%, | Qiang and Cheng 2019 |

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| | | respectively. No significant changes were seen in *sod* expression. | |
| *Danio rerio* (Zebrafish)<br><br>21 days | PS microbeads (5 µm) at 50 µg/L and 500 µg/L<br><br>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.<br><br>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)<br><br>10 days | Uncoated PS particles (1.25 µm) at 2 mg/L, 4 mg/L, and 8 mg/L<br><br>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.<br><br>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).<br><br>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 |

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| In the uptake study, exposure was 24 hours<br><br>In the chronic exposure study, exposure was 48 days | soft drink bottles made from PET<br><br>In the uptake study, concentrations used were 0.8 particles/mL, 40 particles/mL, and 4000 plastics/mL<br><br>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<br><br>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.<br><br>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)<br><br>For uptake experiments, exposure was seven days<br><br>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<br><br>For uptake experiments, a concentration of 40 000 µg/L was used<br><br>For toxicity tests, nominal 40 µg/L ($5.4 \times 10^2$ particles/mL), 400 µg/L ($5.4 \times 10^3$ particles/mL), 4 000 µg/L ($5.4 \times 10^4$ particles/mL) and 40 000 µg/L ($5.4 \times 10^5$ particles/mL) | No significant differences in survival were seen with microplastic exposure.<br><br>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*.<br><br>Acetylcholinesterase, alanine aminotransferase, and catalase activities in all treatment groups were significantly lower than seen in the control.<br><br>The activities of superoxide dismutase, aspartate transaminase, GSH[e], and GPx[f] increased in crabs | Yu et al. 2018 |

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| | 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).<br><br>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 |
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| *Oncorhynchus mykiss*<br>(Rainbow trout)<br>Four weeks | Colourless PS particles (100–400 µm) at approx. 500–700 particles/day/fish<br><br>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.<br><br>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*<br>(Rotifer)<br><br>48 hours<br><br>*Tigriopus fulvus*<br>(Crustacean)<br>48 hours<br>*Acartia clausi*<br>(Marine copepod)<br><br>48 hours<br><br>*Mytilus galloprovincialis*<br>(Mussel)<br><br>48 hours | Non fluorescent LDPE microplastics (1–500 µm)<br><br>Fluorescent green and red PE microplastics were used to examine particle ingestion in rotifers, copepod, and mussel larvae (nominal size of 1–5 µm)<br><br>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<br><br>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.<br><br>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 |

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| *Paracentrotus lividus* (Sea urchin)<br><br>48 hours<br><br>*Oryzias melastigma* (Fish)<br><br>1–13 days post fertilization | | | |
| *Lophelia pertusa* (Cold-water coral)<br><br>For capture rate and polyp activity experiments, exposure was 7, 20, or 47 days<br><br>For coral growth rate experiments, exposure was 69 days | LDPE microbeads (500 µm) at 350 beads/L<br><br>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.<br><br>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)<br><br>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<br><br>Control group was microalgae with surfactant at its highest concentration | Daily growth rate was not affected by microplastic exposure for all test concentrations.<br><br>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)<br><br>*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 |

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| 2 days | 500 rim, and 850–1 000 rim).<br><br>Experiment 2 (Determination of ingestion size ranges and retention): Uncured, PE microbeads (size ranges of 212–250 rim, 425–500 rim, 850– 1000 rim, 1.7–2.0 mm, and 2.4–2.8 mm). Polyps were fed three microbeads from each size class.<br><br>Experiment 3 (Comparing microbeads and microfibres): Uncured, fluorescent, PE microbeads (425– 500 rim) and uncured, fluorescent polyester microfibres (3–5 mm long). Polyps were fed three plastics of each type.<br><br>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 rim, 850–1 000 rim, 1.7– 2.0 mm, and 2.4–2.8 mm microbeads offered. However, a 212–250 rim size class did not elicit a feeding response in either species. No significant differences in egestion were evident in any size classes.<br><br>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)<br><br>3, 5 and 8 days | PS microbeads (90 rim) at 5 particles/mL (nominal)<br><br>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 rim and 6 rim)<br><br>For toxicity tests, concentrations used | Toxicity of beads was size- and concentration-dependent. In the 6 rim treatment group, *B. koreanus* had slightly irregular growth, and no significant | Jeong et al. 2016 |

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| For toxicity tests, exposure was 12 days<br><br>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<br><br>For ingestion, egestion, ROS levels, MAPK activation, and antioxidant enzyme experiments, the concentration used was 10 µg/mL<br><br>Control group was not exposed to microplastic | changes in fecundity and life span.<br><br>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.<br><br>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*<br><br>(Marine copepod)<br><br>24 hours | Non-functionalized PS microbeads (0.5 µm and 6 µm)<br><br>For toxicity tests and ROS levels experiments, concentrations used were 0.1 µg/mL, 1 µg/mL, 10 µg/mL, and 20 µg/mL<br><br>For ingestion, egestion, western blot, and antioxidant enzyme experiments, the concentration used was 10 µg/mL<br><br>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.<br><br>*P. nana* exposed to 0.5 µm microbeads showed delayed molting. No observable effects were seen with 6 µm microbeads.<br><br>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*<br>(Gilt-head seabream)<br><br>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.<br><br>Levels of glucose, aspartate transaminase, alanine transaminase, lactate | Jovanović et al. 2018 |

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| | 36.3 rim), PE (average molecular weight medium density; 54.5 ± 21.3 rim), PVC (low molecular weight; 87.6 ± 16.8 rim)<br><br>Concentration used was 0.1 g/kg body weight/day<br><br>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.<br><br>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)<br><br>95 days post hatching | PS microplastics (2–5 µm)<br><br>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.<br><br>In the second trial, an additional treatment of 10 particles/mL was added<br><br>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.<br><br>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.<br><br>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 |

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| | | in comparison to the control group.<br><br>*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*<br>(Blue mussel)<br>For ingestion and egestion tests, exposure time was Four hours<br><br>For the larval growth tests, exposure time was 15 days | Fluorescent PS beads (2 µm)<br><br>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)<br><br>For the larval growth tests, concentrations used were 0.42 µg/L, 28.2 µg/L and 282 µg/L<br><br>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.<br><br>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*<br>(Marine diatom algae)<br><br>*Dunaliella tertiolecta*<br>(Marine flagellate algae)<br><br>*Chorella vulgaris*<br>(Green microalgae)<br>72 hours | Uncharged PS microbeads (0.5 µm and 6.0 µm) were used on *D. tertiolecta*<br><br>Negatively charged carboxylated PS microbeads (0.5 µm) were used on all three test species<br><br>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.<br><br>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 |

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| | Authors noted that the average measured concentration was up to 9x lower than the nominal concentration in the 6.0 µm treatment group.<br><br>Control group was not exposed to microplastic | rate. Effects were less than 10% for the 6 µm beads. | |
| *Sebastes schlegelii* (Jacopever)<br><br>14 days | Green fluorescent PS microbeads (15 µm) at $1 \times 10^6$ microplastics/L<br><br>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.<br><br>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.<br><br>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.<br><br>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 |

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| | | 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 |
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| *Folsomia Candida*<br>(Soil springtail)<br><br>28 days | PE beads (<500 µm; size distribution of 32% <50 µm, 25% between 50 and 200 µm, and 43% between 200 and 500 µm)<br><br>Concentrations used were 0.005%, 0.02%, 0.1%, 0.5%, and 1% microplastics w/w in dry soil<br><br>Control group was exposed to soil without microplastic | Average survival rates were higher than 80% in all three conditions.<br><br>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.<br><br>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.<br><br>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*<br>(Soil springtail)<br>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 |

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| | 29± 4 µm, and<br>248 ± 14 µm)<br><br>Plastic fragments (average diameters of<br>44 ± 39 µm,<br>282 ± 131 µm, and<br>676 ± 479 µm)<br><br>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<br><br>Concentrations in solution were 10 mg/L and 20 mg/L for the roughly 0.50 µm microbeads<br><br>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)<br><br>Three days | PS microplastics (0.5 µm, 1.0 µm, 2.0 µm, and 5.0 µm) at 1.0 mg/L<br><br>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.<br><br>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 |

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| | | 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.<br><br>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)<br><br>*Abra nitida* (Saltwater clam)<br><br>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<br><br>A low background contamination with perfluorooctane sulfonate was found in microplastics<br><br>Control group was exposed to clean sediment | No significant changes in mortality, condition index, or burrowing behaviour were seen between treatments in both species.<br><br>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.<br><br>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 |

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| | | carbohydrate content, and total energy were seen. | |
| *Perinereis aibuhitensis* (Clamworm)<br><br>Four weeks | PS microspheres (size ranges of 8–12 rim and 32–38 rim) at 100 beads/mL and 1 000 beads/mL (nominal)<br><br>Control group was exposed to 0.45 rim-filtered seawater without microplastic | The presence of microplastics increased mortality in *P. aibuhitensis*, with 8–12 rim microbeads having a significantly higher effect than the other treatments. For example, exposure to 8–12 rim microspheres at 100 beads/mL led to an average survival of 38% compared to over 80% in the control.<br><br>Segment regeneration was size-dependent, with the slowest rate being observed in worms exposed to 8–12 rim (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)<br><br>*Asellus aquaticus* (Isopod)<br><br>*Sphaerium corneum* (Bivalve)<br><br>*Lumbriculus variegatus* (Worm)<br><br>*Tubifex* spp. (Worm)<br><br>28 days | Irregular PS fragments (20–500 rim) mixed with sediment at 0.1%, 1%, 5%, 10%, 20%, 30% and 40% sediment dw<br><br>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).<br><br>No differences in growth were seen in *A. aquaticus*, *S. corneum*, *H. azteca*, *L. variegatus*, and *Tubifex* spp.<br><br>In *H. azteca*, there were no differences in feeding activity at all concentrations. | Redondo-Hasselerharm et al. 2018 |

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| | | In *L. variegatus and Tubifex* spp., microplastic exposure had no negative effect on egestion.<br><br>No microplastics were found in the body and fecal pellets of *H. azteca*. | |
| *Chironomus tepperi* (Sediment dwelling midge)<br><br>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<br><br>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.<br><br>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.<br><br>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.<br><br>It is hypothesized that the 27 µm particles had the greatest effects since they are | Ziajahromi et al. 2018 |

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| | | the ideal size for consumption and to avoid egestion.<br><br>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)<br><br>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<br><br>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.<br><br>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)<br><br>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 |

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| | 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%.<br><br>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<br><br>Dietary<br><br>90 days<br>(7 d/week) | Nonwoven, spunbond polymer fabric made of PE and PET (milled to fine powder)<br><br>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<br><br>i.e., no treatment-related adverse effects on blood parameters, organ weights or histopathology of the liver<br><br>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<br><br>Oral gavage<br><br>28 days<br>(7 d/week) | Fluorescent PS<br><br>5 µm and 20 µm in diameter | $1.46 \times 10^6$ items of 5 µm particles at 0.1 mg/day<br><br>$2.27 \times 10^4$ 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<br><br>(Translocation to the liver and kidney reportedly occurred and particles could be detected one week after cessation of exposure.) | Deng et al. 2017 |
| Mice<br><br>Oral gavage | Virgin PS<br><br>5 µm and 20 µm in diameter | $1 \times 10^5$ 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 |

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| 28 days<br>(7 d/week) | | $2 \times 10^3$ items of 20 µm particles at 0.01 mg/day<br><br>$1 \times 10^6$ items of 5 µm particles at 0.1 mg/day<br><br>$2 \times 10^4$ items of 20 µm particles at 0.1 mg/day<br><br>$5 \times 10^6$ items of 5 µm particles at 0.5 mg/day<br><br>$1 \times 10^5$ items for 20 µm at 0.5 mg/day | Incidence or severity data not reported<br><br>Energy metabolism:<br>Both sizes of PS induced a decrease in ATP level and significant decrease in LDH[b] activity in a dose-dependent matter<br><br>Lipid metabolism:<br>Decreases in all treatments for the levels of total cholesterol and triglycerides<br><br>Biomarkers of oxidative stress:<br>Increased GPx activity (more so in 5 µm group) and SOD;<br>Decrease in catalase activity in almost all the treatment groups<br><br>Potential for neurotoxicity:<br>Decreased acetylcholinesterase activity in liver after exposure to two sizes of PS microplastics, but more so in 5 µm group | |
| Mice<br><br>Oral gavage<br><br>28 days (three times /week) | PS<br><br>1 µm, 4 µm and 10 µm in diameter | Mixture of 1 µm ($4.55 \times 10^7$ particles),<br>4 µm ($4.55 \times 10^7$ particles), and 10 µm ($1.49 \times 10^6$ 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<br><br>Little presence of particles in cells of the jejunum and duodenum<br><br>No particles were found in other organs (liver, spleen and kidney) | Stock et al. 2019 |

| | | | | |
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| Mice<br><br>Drinking water<br><br>Six weeks (continuous exposure) | Virgin and fluorescent PS<br><br>5 µm in | 1.456× $10^6$ particles/L of 5 µm particles at 100 µg/L<br><br>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<br><br>Gut microbiota dysbiosis (change in the composition of the gut microbiota in the cecal contents of the mice) at both doses<br><br>Intestinal barrier dysfunction<br><br>Alterations in amino acid and bile acid metabolism with 1 000 µg/L exposure | Jin et al. 2019 |
| Mice<br><br>Drinking water<br><br>Five weeks (continuous exposure) | PS<br><br>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<br><br>1.456 × $10^{10}$ particles/L of 0.5 µm at 1 000 µg/L in drinking water<br><br>1.456 × $10^4$ particles/L of 50 µm at 100 µg/L in drinking water<br><br>1.456 × $10^4$ particles/L of 50 µm at 1 000 µg/L in drinking water | Altered hepatic lipid metabolism<br>Altered gut microbiota composition | Lu et al. 2018 |
| Mice<br><br>Drinking water<br><br>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 x $10^8$ items/L) and 10 µg/L OPFRs<br><br>2 000 µg/L PS (3.7 x $10^8$ items/L) and 100 µg/L OPFRs<br><br>2 000 µg/L PE (3.7 x $10^8$ items/L) and 10 µg/L OPFRs | Increased oxidative stress, increased neurotoxicity, enhanced disruption of amino acid metabolism and energy metabolism from co-exposure<br><br>No microplastic-only control group; it is unclear what component of the treatment contributed to the effects | Deng et al. 2018 |

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| | | 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<br><br>Nose-only inhalation<br><br>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<br><br>Inhalation in chamber air<br><br>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<br><br>Inhalation in chamber air<br><br>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 |

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| | | | 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<br><br>Inhalation in chamber air<br><br>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<br><br>Nose-only inhalation<br><br>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 |

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| | | | 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<br><br>Inhalation in chamber air<br><br>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 al. 1975 |
| Rats<br><br>Nose-only inhalation<br><br>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 |

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| | 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.<br>NOEC = 10.6 mg/m$^3$<br>$NOEC_{adj}$ = 2.7 mg/m$^3$ | |
| Rats<br><br>Intratracheal<br><br>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.<br>LOEL[g] = 25 mg | Agarwal et al. 1978 |
| Rats<br><br>Intratracheal<br><br>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 | 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.<br>LOEL = 2 mg | Pigott and Ishmael 1979 |
| Rats<br><br>Intratracheal<br><br>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.<br>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 |

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| | | | decreased to control levels at later timepoints; most effects persisted longer in washed PVC than non-washed.<br>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<br><br>Intratracheal<br><br>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.<br>Significant increase in polymorphonuclear leukocyte cell count in BALF. Significant increase in chemiluminescence but not cell count for alveolar macrophage.<br>Significant increase in albumin (indicator of blood-gas barrier | Porter et al. 1999 |

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| | | | deterioration) and matrix metalloprotease activity (indicator of inflammation). LOEL = 10 mg/kg | |
| Rats<br><br>Intratracheal<br><br>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<br><br>Intratracheal<br><br>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<br><br>Intratracheal<br><br>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

Canadian Chamber of Commerce
Chambre de Commerce du Canada

# Canadian Chamber of Commerce Fresh Fuel Standard Campaign (Associations)

Led by: Dr. Aaron Henry, Senior Director of Natural Resources and Sustainable Growth & Jarred Cohen, Policy Advisor for Transportation and Infrastructure

October 7, 2020

Canadian Chamber of Commerce
Chambre de Commerce du Canada

# Context

- CFS is one of many government initiatives used to address serious issue of climate change
- Canadian Chamber of Commerce supports a clean fuel standard in Canada, but not in current form
- Under current design, CFS will hurt Canadian economic recovery, undermine long-term competitiveness, and may fall short of its objective

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# Four Asks

1. Ensure future designs of CFS are aligned with other jurisdictions that have exempted industrial fuels from own standards

2. Design CFS that is tech neutral, with consistent costs & stringency

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# Four Asks (cont'd)

3. Ensure CFS is aligned with OBPS (Out-put based Price System) and design low cost carbon credit and offset options for compliance across both regimes

4. Complete due diligence to ensure all three fuel streams are subject to a regulatory impact assessment before going into force.

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# Objective

Launched in mid-September, the Chamber has activated our initiative using both internal and external services activating both ongoing digital & conventional public campaign on the CFS. This campaign will urge Cabinet and MPs to work with ECCC (Environment and Climate Change Canada) to address key competitiveness concerns. The campaign will likely be active during the 75 day comment period on the liquid stream negotiations, but we intend to use our resources to be active throughout the year.

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# Desired Metrics

1. To have 25% of targeted MPs and 50% of targeted Cabinet ministers speak out publicly on the issue

2. To mobilize 75% of Chambers that operate in the constituencies of key parliamentarians, to engage on this issue through social media and letter-writing campaigns

3. To successfully present our view on the CFS in national media through interviews & paid op-eds

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# Campaign Components

- Develop a compelling narrative with approachable messages that will motivate engagement from the Chamber community

- Full-spectrum media campaigns to support efforts:
  - Paid advertising (e.g. radio, social media, billboards)
  - Earned media (media relations push)
  - Shared media
  - Owned media

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# Campaign Components

- Mobilization of supporters (biz community)
- Additional assets, toolkits for relevant Chambers to further campaign efforts
- Relevant graphic design assets to create distinct campaign brand

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## Option A - Campaign Supporter - (complimentary for association members of the Canadian Chamber of Commerce)

- **Deliverables**
    - Association logo visibility on the campaign landing page with a link back to your organization website at: freshfuelstandard.ca

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## Option B - Campaign Champion - cost to be determined on a case by case basis with your association factoring in size/budget for each organization

- **Deliverables (in addition to logo visibility on campaign landing page at Freshfuelstandard.ca with a link back to your association website you will also receive the following:)**

- A seat on the campaign steering committee (your association can help shape the direction/strategy for the campaign)
- Exclusive invitations to targeted advocacy meetings with government officials and/or parliamentarians
- Opportunity for your association to be interviewed for the campaign video series addressing competitiveness
- Canadian Chamber's 5 Minutes for Business publication on the topic of CFS (e-issue sent to corporate, associations and chambers/boards of trade across the country)
    - Logo visibility for your association on the publication

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# Campaign Champion - con't

- **Deliverables**

- Through your financial support you will also be contributing to:
  - Toolkit for local Chambers to promote this issue with local MPs through social media, virtual roundtables and other tools
  - Mobilize 75% of Chambers that operate in the constituencies of key parliamentarians, to engage on this issue through social media and letter-writing campaigns
  - Efforts towards 25% of targeted MPs and 50% of targeted Cabinet Ministers to speak out publicly on the issue
  - Graphic design assets to create distinct campaign brand
  - Dedicated landing page for campaign at: freshfuelstandard.ca
  - Media campaign to support efforts:
    - Paid advertising (radio, social, media billboards)
    - Earned (media relations push)
    - Shared and owned media

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# Northwest Ports Clean Air Strategy (NWPCAS) Renewal 2020-2030

Round #3 Engagement

October 2020

# Welcome

The Vancouver Fraser Port Authority is currently seeking feedback on the draft the Northwest Ports Clean Air Strategy for 2020 – 2030.

We invite your feedback on the draft strategy and have prepared this presentation to assist you in reviewing the material.

The participating ports plan to finalize the new air strategy at the end of 2020 and are therefore keen to hear your views on this important topic.

This presentation provides an overview of the draft strategy and should be reviewed in conjunction with the draft Northwest Ports Clean Air Strategy available on the consultation website at portvancouver.civilspace.io/en/projects/northwest-ports-clean-air-strategy-renewal

Thank you for taking the time to review the materials and provide feedback!

# Overview

This presentation provides an overview of the following draft content for the 2020 – 2030 Northwest Ports Clean Air Strategy:

- Vision to phase out port-related emissions by 2050
- Guiding principles to inform decisions
- Emission reduction themes identified that apply to each supply-chain sector
- Challenges, roles and responsibilities
- Areas for port authority collaboration
- Objectives for each supply-chain sector (i.e. ocean-going vessels, cargo-handling equipment, trucks, harbor vessels, rail, port administration and tenant facilities)
- Reporting approach
- Actions the Vancouver Fraser Port Authority can take to advance the strategy in the Port of Vancouver

**Please note:**
We have included additional reference slides at the end of this presentation that provide a brief history and background of the Northwest Ports Clean Air Strategy.

You can find more information on our website at www.portvancouver.com

# An ambitious vision to phase-out port related emissions by 2050

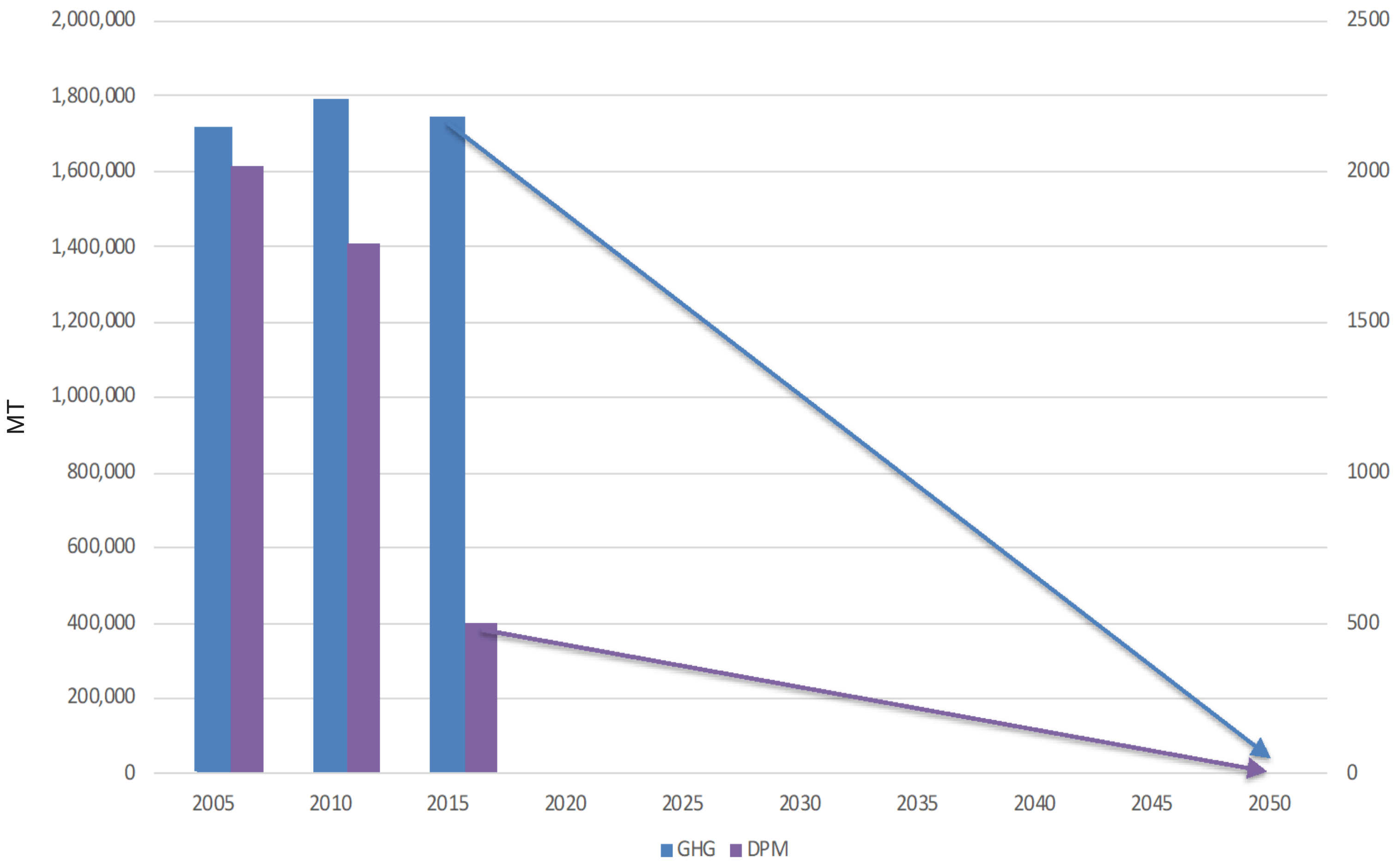
The following is the proposed vision for the new air strategy:

**Phase-out emissions from seaport-related activities by 2050, supporting cleaner air for our local communities and fulfilling our shared responsibility to help limit global temperature rise to 1.5°C.**

The challenge of international shipping:

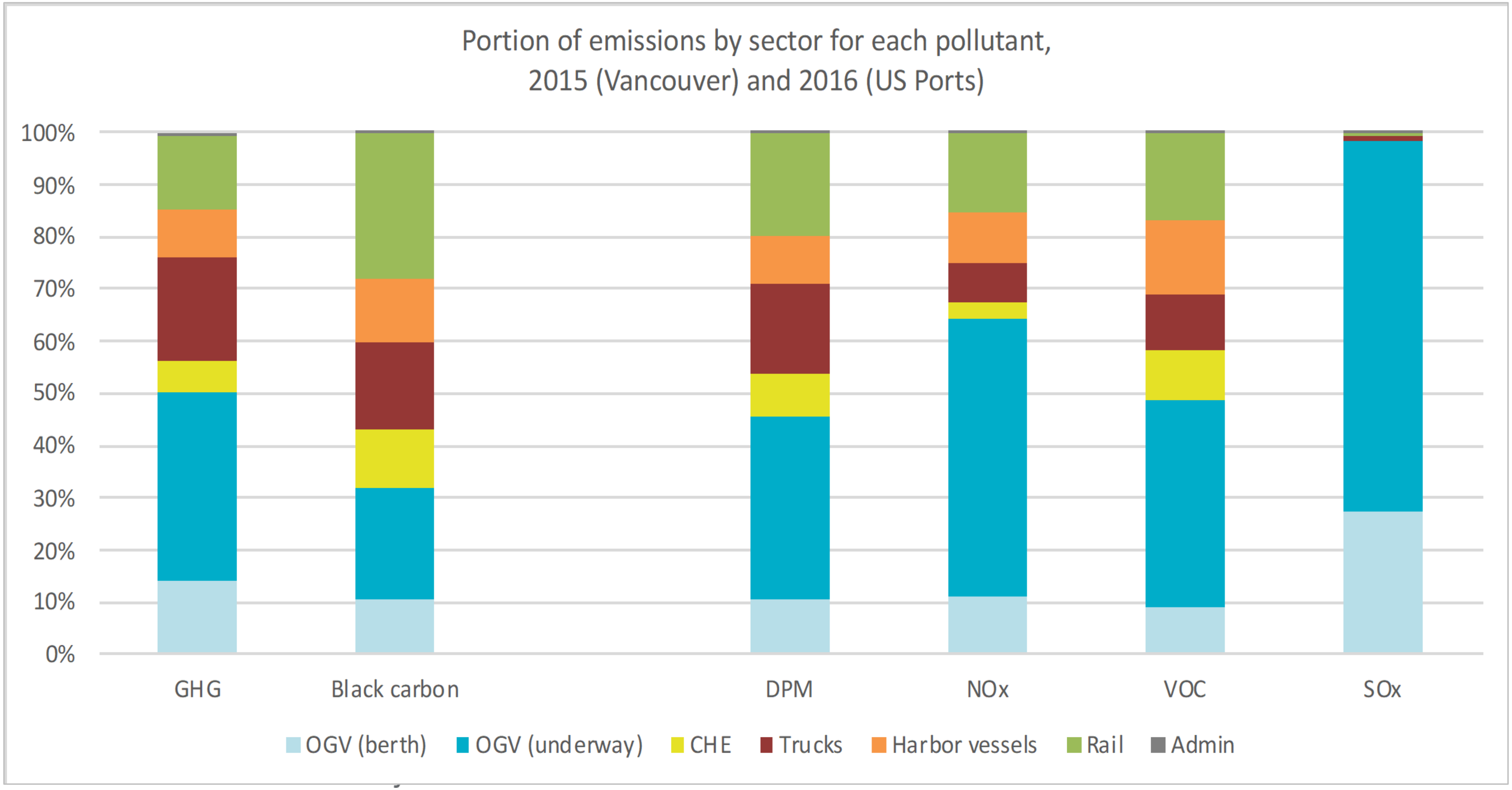
- International Maritime Organization (IMO) targeting 50% reduction in total GHG emissions from shipping by 2050, relative to 2008
- This target is aligned with global efforts to achieve net zero emissions by 2050 and limit global warming to 1.5°C
  - ***Participating ports can encourage vessels to go beyond IMO target, and for zero emission vessels to trade in the Pacific Northwest***

# Vision for 2050




The following provides a simple overview of the enormous challenge presented by the ambitious vision to phase-out port emissions by 2050.

# Emissions by sector



# Guiding principles

The following are proposed guiding principles for the new air strategy:

| Guiding principle | Description |
|---|---|
| Community health | Recognize the importance of reducing the impacts of seaport-related emissions on public health. |
| Climate urgency | Seek early achievement of the vision, recognizing the urgency to act to limit global climate change. |
| Social equity | Prioritize action in communities that have been most impacted by port operations. |
| Innovation | Promote investment in innovative technologies, policies and practices that drive continuous improvement. |
| Evidence-based decisions | Use best available climate change and air quality science to inform decisions. |
| Focused resources | Focus action in areas likely to have the highest environmental, social and economic impact, recognizing the limits of port authority resources, operational control and influence. |
| Leadership | Take a leadership role to facilitate government and industry support for the policy and actions needed to achieve the vision. |
| Accountability | Provide clear, transparent and timely updates on progress toward achieving the vision. |
| Port competitiveness | Deliver the strategy in a way that supports competitiveness of ports and the prosperity of communities. |

# Themes for phasing-out emissions

| Policies to continually improve efficiency and reduce emissions | Infrastructure to support zero-emission equipment | Demonstration and adoption of zero-emission equipment |
| --- | --- | --- |
| Port authorities will continue to advance policies that promote efficiency and the phase out of old, high-emitting equipment, in favour of new equipment | Port authorities will facilitate collaboration among government, utilities, fuel providers and industry to ensure the infrastructure needed to enable zero-emission technologies is in place at the right time | Port authorities will facilitate the collaboration necessary between government and industry to advance demonstration and adoption of zero emissions equipment in each sector |

# Challenges, roles and responsibilities

| Challenge | Port Authority | Other Stakeholders |
|---|---|---|
| **Enabling regulation and funding** is in place to support investment in zero-emission equipment and infrastructure | Advocate for and secure government policy and funding support | Implement supportive policies and funding |
| **Adequate electricity or fueling infrastructure** is available when and where needed | Plan and deliver infrastructure with utilities, industry, tenants, governments and other stakeholders | Plan for and install needed capacity and infrastructure |
| **Suitable zero-emission equipment** is commercially available, demonstrated for port applications, and total cost of ownership is competitive (may require enabling regulation and funding) | Technology assessments and facilitation of demonstration projects<br>Coordination of funding<br>Advocacy for standardization of charging | Fund technology development<br>Design and commercialization<br>Participate in demonstration projects<br>Provide favourable financing options |
| **Industry commitment** to transition to zero emissions through investment and business planning | Technical resources and funding alignment<br>Dissemination of information | Incorporate air strategy into capital plans and operations<br>Funding support |
| **Labor force** is trained to operate and maintain zero-emission equipment | Advocate for and support education and training programs | Provision of training and accreditations |
| **Community and public support** to advance the vision and objectives of the strategy | Increase understanding of required policy, funding, investment and action needed<br>Communicate progress | Where appropriate, provide expertise and support for policy and funding |

# Collaboration between participating ports

The participating ports have identified areas where coordinated action will improve overall outcomes and progress toward the vision.

## Why?

- Improved outcomes
- Sharing of lessons learned
- Elevating collective understanding
- Better engagement with government, industry and communities
- Pooling of resources

## How?

- Continue to advance the phase-out of old, high emitting diesel equipment
- Advance pilot projects together
- Conduct studies to assess infrastructure needs
- Undertake air quality and technology studies
- Engage with industry, government, communities and the non-profits
- Advocate for significant government and industry investment in zero emissions
- Comprehensive emission inventories
- Seek opportunities to better consider GHGs and air pollutants in supply chain efficiency programs
- Publish annual reports

# Sector objectives: ocean-going vessels

The following draft objectives have been identified to advance the vision for ocean-going vessels:

Continually increase vessel efficiency and decrease emissions from vessel operations

- Grow participation in ship incentive programs to promote e.g. wind assist (foils, rotors), operational practices

By 2030, install shore power at all major cruise and container berths

- Requires substantial investment by ports, government, utility providers, coordination with terminal operators
- Ports can establish programs, tariffs or special agreements to encourage use once infrastructure is available

Support international efforts toward phasing out emissions from ocean-going vessels

- Advocate for international policies that align with NWPCAS vision
- Facilitate land-side infrastructure and safety procedures
- Promote increased used of zero-emission fuels
- Participate in pilot studies

# Sector objectives: cargo-handling equipment

The following draft objectives have been identified to advance the vision for cargo-handling equipment:

**Continually advance equipment efficiency and decrease emissions from cargo handling equipment**

- Accelerate efforts to remove old diesel equipment from operation, in favour of Tier 4 diesel or better
- Promote best practices in fuel efficiency plans, peer-to-peer learning on emission reductions and enable pilot projects

**By 2030, sufficient infrastructure is in place to begin transition to zero-emission cargo handling equipment**

- Collaborate with utilities and terminal operators to identify and plan for required network upgrades
- Work with terminal operators to understand feasibility of hydrogen and collaborate on demonstration projects and fueling infrastructure needs

**By 2050, zero-emission cargo handling equipment is adopted**

- Support demonstration projects, advocate for supportive government policies, identify funding strategies and engage with industry and labor on training needs
- Explore lease requirements or fee-based programs to promote transition

# Sector objectives: container (drayage) trucks

The following draft objectives have been identified to advance the vision for container trucks:

**Continually advance vehicle efficiency and decrease emissions from existing trucks**

- Port policies that promote use of new, lower emission diesel trucks, discourage idling, penalize unlawful deletion of emission controls, and that consider access to needed capital
- Programs to reduce congestion and emissions in sensitive/affected neighbourhoods

**By 2030, sufficient infrastructure is in place to begin the transition to zero-emission trucks**

- Participate in regional collaborations to provide infrastructure for zero emissions trucks
- Work with industry and near-port communities where trucks park/idle to understand viable zero-emission technologies and facilitate infrastructure planning

**By 2050, zero emission trucks are adopted**

- Support demonstration projects, advocate for supportive government policies and funding and engage with labour on training needs
- Monitor total cost of ownership to inform policy and program development

# Sector objectives: harbour vessels

The following draft objectives have been identified to advance the vision for harbour vessels:

**Continually advance vessel efficiency and turnover of old, high-emitting vessels**

Accelerate efforts to support turnover to cleaner engines, hybrid vessels and support implementation of efficient vessel operating practices e.g. update port programs to drive use of efficient technologies; conduct outreach about efficient operations

**By 2030, sufficient infrastructure is in place to enable adoption of zero-emission harbor vessels**

Engage operators to understand infrastructure needs, and (engage?) regional partners to support planning and installation of infrastructure

**By 2050, zero-emission harbor vessels are adopted**

Support demonstration projects, advocate for government funding for research and development and supportive policies, identify transition funding strategies and support work force training needs

# Sector objectives: rail

The following draft objectives have been identified to advance the vision for rail:

**Continually advance equipment efficiency and turnover of old, high-emitting engines**

- Continue efforts to remove old, unregulated and Tier 1 engines from operation on port property e.g. update port programs and facilitate use of efficient technologies particularly for tenant-owned switcher locomotives
- For line-haul locomotives, advocate for e.g. lower emission standards for new and re-built engines, adoption of policies supporting use of hybrid engines, and conducting more engine repowers or replacements to Tier 3 or higher

**By 2030, sufficient infrastructure is in place to enable adoption of zero-emission on-terminal rail**

- Coordinate and facilitate planning with utilities, fuel providers and terminal operators to understand technologies, infrastructure needs, grid impacts and timing for terminal-owned switchers
- For line-haul and rail company-owned switchers, engage to understand needs and support regional efforts

**By 2050, zero-emission on-terminal rail is adopted**

- For terminal-owned switchers, support demonstration projects, advocate for supporting government policies and funding and engage operators on training needs

# Sector objectives: Port administration and tenant facilities

The following draft objectives have been identified to advance the vision for port authority administration and tenant facilities:

**Continually advance efficiency in port authority fleets, facilities and lighting**

- Develop retrofit programs, support efficiency improvements, engage with tenants
- Adopt hybrid equipment in interim

**By 2030, port authority passenger fleet is zero-emission vehicles or uses renewable fuels**

- Establish internal purchasing policies to require zero emission vehicles and equipment, install charging infrastructure, use renewable fuels where electric is not feasible

**By 2050, port authorities have adopted zero-emission for remaining vehicles, equipment and vessels**

- Early adoption of zero / near-zero emission technology

**By 2050, zero-emission buildings and high-efficiency lighting are in place**

- Update policies to ensure zero emission building standards incorporated for new buildings
- For existing buildings and lighting, prepare green building policies, establish port programs or lease requirements, engage with tenant

# Reporting approach

The participating ports remain committed to publishing annual reports that track progress toward the vision and objectives of the strategy.

Feedback from the previous round of consultation indicates preference for concise and accessible reporting format.

The following are examples of indicators that will inform reporting:

- Absolute GHG emissions and air pollutants and trends
- Total investment and policy facilitated to support strategy objectives
- Availability and utilization of shore power facilities
- Progress on demonstration and pilot projects
- Per cent trucks meeting 2007 emission standards
- Per cent non-road engines meeting Tier 4 emission standards
- Uptake of low or zero emission technologies

# Implementation plans

As a next step, each participating port will develop an implementation plan to advance the vision and objectives of the air strategy.

Implementation plans will respond to the specific business and operational context of the respective ports.

The following slides provide a simple overview of some of the actions that the Vancouver Fraser Port Authority can take to advance the vision and objectives of the air strategy in the Port of Vancouver.

# Implementation plan: cross-sector actions

| VFPA highlights of actions that apply to multiple supply-chain sectors |
|---|
| Annual review process to assess implementation plan effectiveness |
| Advocate to Government of Canada and Province of B.C. to strengthen policy and regulation to reduce air pollutants and advance zero emission technologies in ports and heavy-duty transportation |
| Work with BC Hydro to deliver electrical distribution network that supports adoption of battery electric technologies in heavy-duty transport |
| Work with Government of Canada and Province of B.C. to deploy clean technology demonstration and pilot projects, in particular in vessels, cargo-handling equipment and drayage trucks |
| Undertake studies to improve understanding of impacts of port-related emissions on air quality |
| Explore how to attract green finance opportunities |
| Incorporate emission reduction strategies into supply-chain efficiency programs and projects |
| Participate in International Association of Ports and Harbours, Climate and Energy Committee |
| Participate in World Ports Climate Action Program |

# Implementation plan: ocean-going vessels

| VFPA highlights for ocean-going vessels |
| --- |
| Expand shore power facilities to all cruise and container berths |
| Continually promote increased participation in EcoAction Program for Ships to promote cleaner and quieter ships |
| Continue to advance initiative to improve global collaboration on ship incentive programs |
| Facilitate LNG bunkering by 2022, and promote engine types with best performance and advocate for OEM improvements in control of methane slip for other engine types |
| Work with fuel providers and government to increase availability of low carbon marine fuel alternatives such as renewable diesel |
| Advance demonstration projects of low carbon marine fuels |
| Participate in World Ports Climate Action Program, Policy Work Group, Power-to-Ship Work Group, and Sustainable Marine Fuels Work Group |
| Participate in Getting to Zero Coalition to accelerate commercialization of a zero emission vessel by 2030 |

# Implementation plan: cargo-handling equipment

| VFPA highlights for cargo-handing equipment |
| --- |
| Participate in World Ports Climate Action Program, Policy Work Group, and Decarbonization of Cargo Handling Facilities Work Group |
| Deliver annual climate change and air quality workshops for tenants to increase awareness and capability of tenants and terminal operators in measuring and reducing air emissions |
| Support tenants and terminal operators in identifying and accessing funding for energy conservation and emissions reduction measures |
| Expand the Non-Road Diesel Emissions Fee to continue to promote phase-out of old, high emission equipment in favour of Tier 4 engines or better |
| Continue to promote and advance opportunities for idle reduction policies, practices and technology |

# Implementation plan: trucks

| VFPA highlights for trucks |
| --- |
| Advocate to Province of B.C. for zero emissions policies for heavy-duty trucks |
| Advocate to Province of B.C. for truck scrappage program |
| Continue with Truck Licensing Environmental Requirements and implementation of 10-year rolling maximum truck age starting in 2022 |
| Continue to improve Truck Licensing System to better address high-emitting trucks and potential deletion of emission controls |
| Advocate to local, regional, provincial governments to implement road congestion reduction mechanisms |

# Implementation plan: harbour vessels

| VFPA highlights for harbour vessels |
| --- |
| Participate in World Ports Climate Action Program, Policy Work Group, and Sustainable Marine Fuels Work Group |
| Participate in Canada - U.S. collaboration to promote decarbonization of domestic vessel traffic |
| Continue to evolve EcoAction Program to better support and recognize harbour vessels |
| Deliver annual climate change and air quality workshops to improve awareness and capabilities in measuring and reducing air emissions |

# Implementation plan: rail

| VFPA highlights for rail |
| --- |
| Explore opportunities to support rail operators in using renewable fuels |
| Deliver annual climate change and air quality workshops to improve awareness and capabilities in measuring and reducing air emissions |
| Expand the Non-Road Diesel Emissions Fee to promote phase-out of old, high emission switch locomotives in favour of Tier 4 engines or better |

# Implementation plan: port administration and tenant facilities

| VFPA highlights for port administration and tenant facilities |
|---|
| Explore opportunity for demonstration of plug-in hybrid patrol vessel for port authority |
| Transition passenger fleet vehicles to zero emission vehicles |
| Assess options for zero emission equipment and vehicles for port authority maintenance department and harbor patrol vessels |
| Continue to advance energy conservation measures for all port authority owned/operated facilities |
| Develop energy performance guidance for tenant developments |

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# We look forward to your feedback!

Please submit your feedback by **Thursday, October 29, 2020**, to:

Christine Rigby, Environmental specialist - air emissions
email: Christine.Rigby@portvancouver.com

Canada

# Background Information

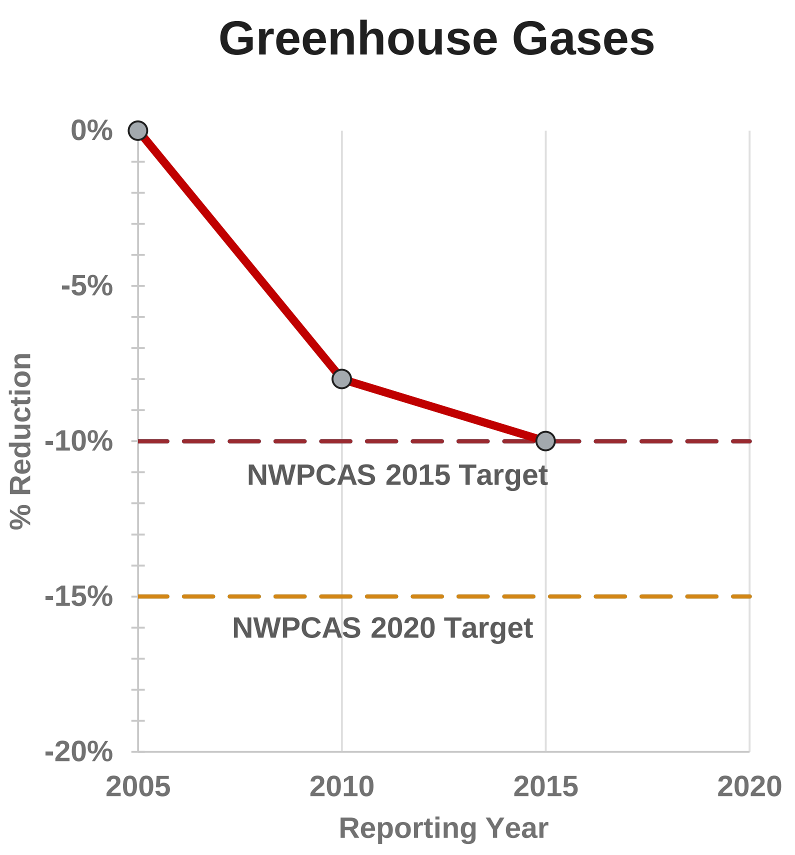
# History

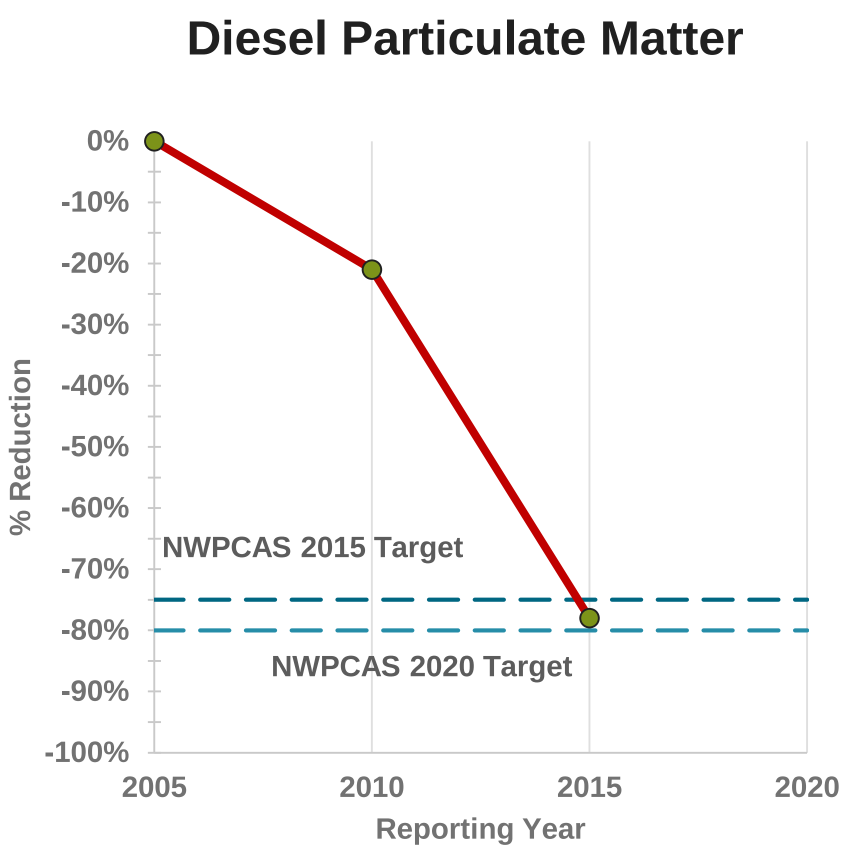

- 2007
  - Ports of Vancouver, Seattle, Tacoma
  - Supported by US EPA, Washington State Dept of Ecology, Puget Sound Clean Air Agency, Environment Canada
  - Sector targets-ocean going vessels, harbour craft, cargo handling equipment, trucks, rail, administration
  - Annual progress reports
  - Diesel particulate matter and greenhouse gases (GHG)
  - 2010, 2015
- 2013
  - + Northwest Seaport Alliance (NWSA)
  - + Province of BC, Metro Vancouver
  - + Airshed targets
  - + 2015 (updated), 2020

# Successes: 2018 Highlights (all ports combined)

- Ocean going vessels
  - 62% of vessel calls participate in port-designed or third party certification programs that promote continuous efficiency improvements
- Harbour vessels
  - 38% of vessels participate in port-designed or third party certification programs
- Cargo handling equipment (CHE)
  - 52% of CHE meets Tier 4 interim emission standards or equivalent
- Container trucks
  - 67% of container trucks meet or surpass EPA emission standards for model year 2007 for particulate matter
- Rail
  - 9% of unregulated switcher locomotive engines upgraded or replaced to Tier 2 or better
- Port Administration
  - 21% of on-road and 48% of non-road vehicles use non-conventional fuels

# 2013 NWPCAS Intensity-based Air shed Targets (All Ports Combined)

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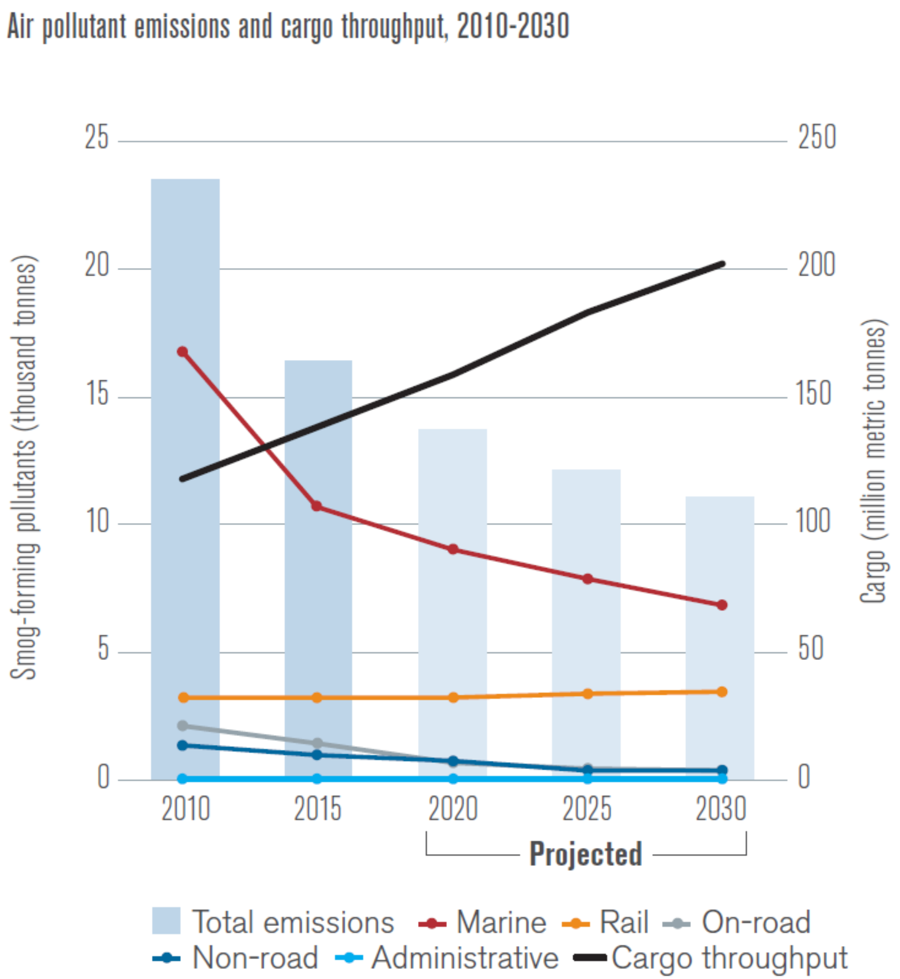
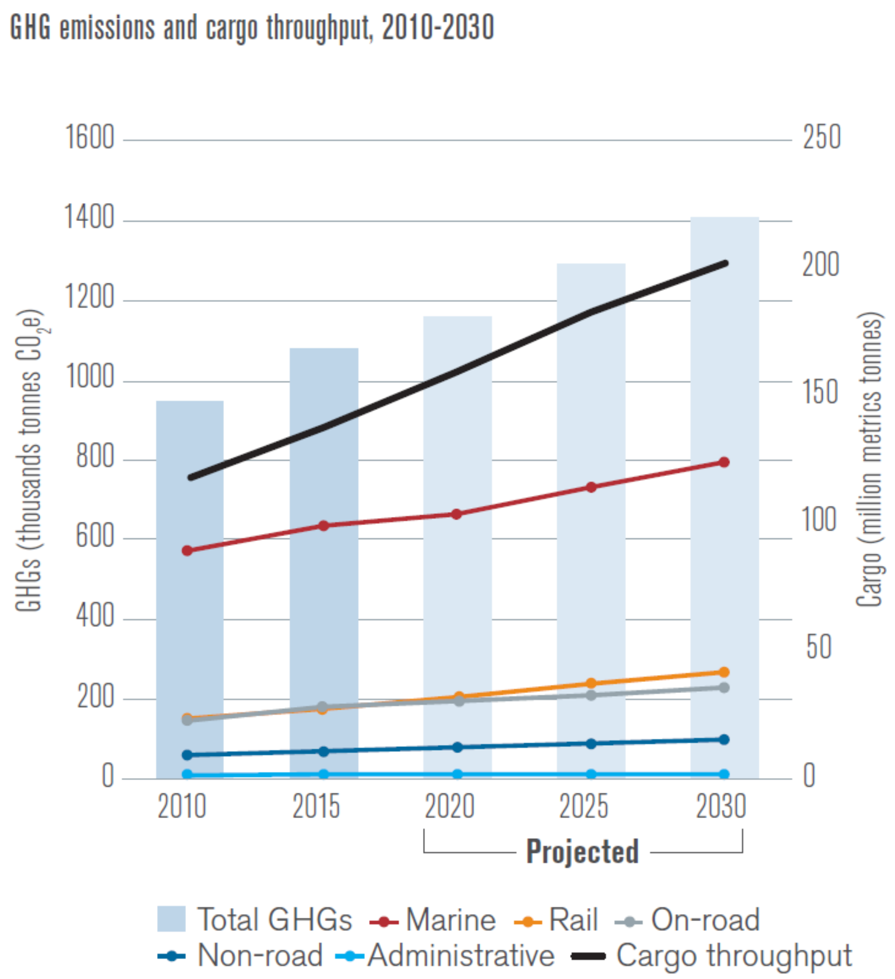
# 2015 Port of Vancouver Emission Inventory

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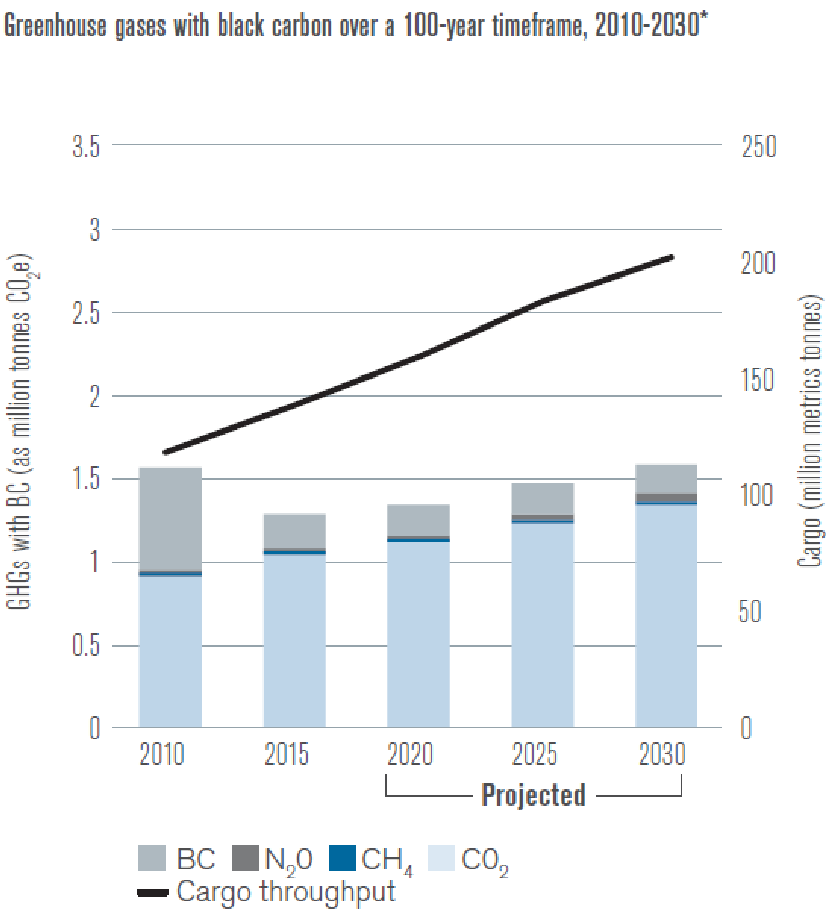
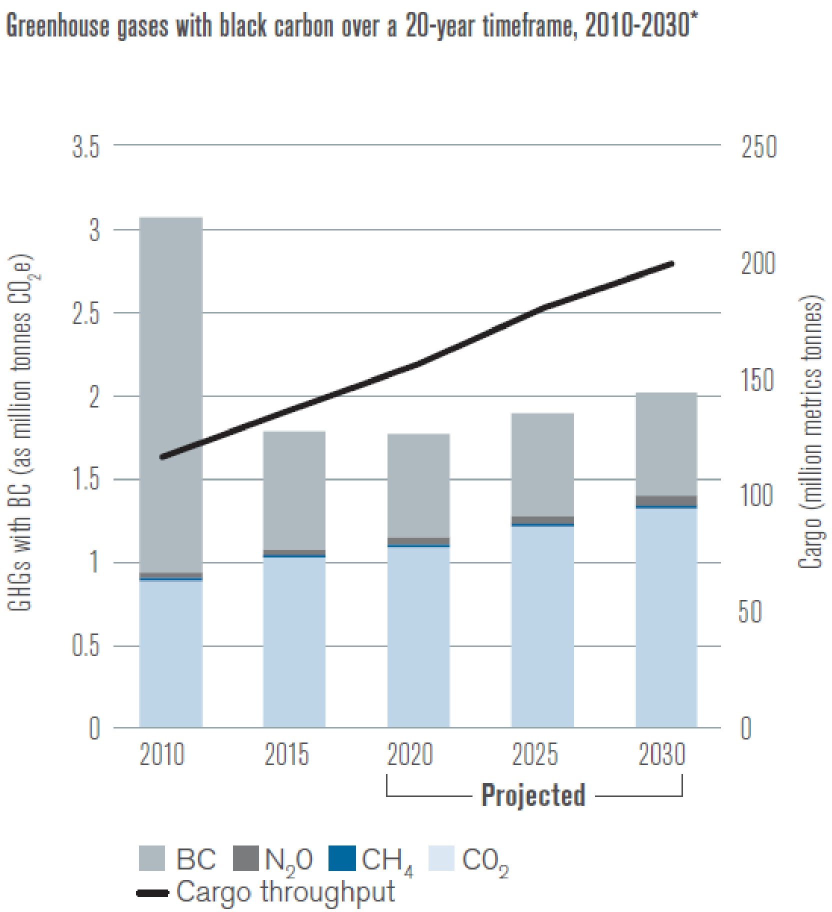
# Black Carbon




*$CO_2e$ 100-year timeframe values: $CH_4$ at 28, $N_2O$ at 265, black carbon at 900.




*$CO_2e$ 20-year timeframe values: $CH_4$ at 84, $N_2O$ at 264, black carbon at 3,200.

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# Review: Engagement Round #1, Summer 2019

- Example actions taken in support of NWPCAS over the last 10 years
- Air quality and climate change emission forecasts
- Why are we creating a renewed strategy, and what does that mean?
  - What have we learned?
  - How has our context changed?
  - Role of port authorities
- Engagement topics:
  - Draft vision
  - Draft guiding principles
  - Preliminary sector technology shifts
  - Port administration and tenant facilities

# Review: Engagement Round #2, Summer 2020

- Vision
- Guiding principles
- Sector objectives
  - Efficiency/interim emissions
  - Equipment
  - Infrastructure
- Conditions for success
- Key actions for ports and other stakeholders
- Preliminary input on monitoring and reporting

Railway Association of Canada | RIGHT TRACK

August 18, 2020

HWIN Modernization
Program Management Branch - Program Oversight
40 St Clair Avenue West
4th Floor
Toronto ON M4V 1M2

**Subject:** ***Discussion paper on modernizing hazardous waste reporting in Ontario***

Dear Sir / Madam:

The Railway Association of Canada (RAC), on behalf of its near 60-member railway companies, welcomes the opportunity to provide feedback on the *Discussion paper on modernizing hazardous waste reporting in Ontario*. In Ontario, the RAC's membership consists of four Class 1 freight railways and 12 shortline railways. In 2018, the total value of goods moved by freight railways in Ontario totaled $7.8 billion respectively and just over one million carloads originated in the province. The RAC applauds the Government of Ontario's efforts to improve the efficiencies and effectiveness of the hazardous waste reporting system while also reducing regulatory compliance costs for businesses and as such, is pleased to provide the following comments.

**Registration and Payment**

1. Generators must maintain records of all data, analysis, and other information used to register subject waste. Now that businesses are moving digital, should we allow supporting records (e.g. data, analysis, and other information used in the preparation of the Generator Registration Report) to be kept electronically? Are paper copies needed?

Overall, the rail sector supports the ability to allow supporting records to be kept electronically. Moving to electronic systems has the potential to increase efficiencies, reduce the use of paper and create further alignment between provinces and the federal government.

3. Currently spills and emergency reporting is managed through the Spills Action Centre. How should we manage registration for emergency situations going forward?

Given the linear nature of the rail sector, a corporate level emergency generator ID would allow for rapid response and clean up in the event of spills, or releases, while assuring traceability of information to the responsible organization.

**Alignment with Federal Government**

The RAC and its federal Class 1 rail carriers that move hazardous waste to, from and across numerous provinces, would like to emphasize the need for a harmonized approach to moving hazardous waste in Canada that is accessible and acceptable to the federal government and provinces. The rail sector is concerned that the development of distinct provincial and federal waste document management systems will introduce further administrative burden for the movement of hazardous waste interprovincially and internationally.

It is our stance that these systems need to communicate with each other so that the multiple authorities receive their respective information through single entries thus avoiding the replication of administrative effort on carriers to meet the obligations of separate systems. This is further complicated by the differences between the provinces with respect to the classification of hazardous waste.

Allowing for data referencing or data transfer between the provincial government and the federal government will no doubt reduce the administrative burden on businesses. Case in point, a railway transporting hazardous waste from British Columbia to Ontario should be able to submit a single manifest that is accepted by all levels of government. This eliminates the need for two different reporting requirements on the same shipment. Furthermore, harmonization with the *Transportation of Dangerous Goods Regulations* (TDG) also needs to be considered. Currently the provincial manifests don't line up with the TDG reporting requirements which results in requiring a separate attestation for TDG that accompanies any TDG-regulated manifests. Correcting this misalignment will further support efficient reporting to authorities and reducing the administrative burden on businesses.

While it is noted in the discussion paper that the Government of Ontario will work closely with Environment and Climate Change Canada (ECC) and Transport Canada (TC) to ensure the new digital reporting service aligns with federal requirements, the RAC would appreciate transparency on what this would entail during the development of the new system in Ontario. Notably, the RAC strongly recommends that changes to Reg 347 under Ontario's *Environmental Protection Act* are adequately harmonized with federal regulations (*Cross-border movements of hazardous waste and hazardous recyclable materials regulations)* to ensure minimal overlap of requirements thus reducing impacts to interprovincial carriers.

Another consideration for the rail sector is the Electronic Data Interchange (EDI) systems used by railways. These EDI systems allow for application to application transfer of business documents and data. In the event the Government of Ontario enacts changes to the type of information required in hazardous waste manifests, it will have a negative impact to the EDI systems. Actioning changes to these EDI systems by railways are both cost prohibitive and time intensive to implement, creating an additional administrative burden rather than reducing red tape.

**Conclusion**

The RAC and its members appreciate the opportunity to provide feedback to the Government of Ontario as it modernizes the hazardous waste reporting system. Please don't hesitate to reach out to the RAC should you require additional information and I look forward to continued consultations.

Sincerely,

PBedrossian

Peter Bedrossian
Director of Regulatory Affairs

# Statement of work

## 1.0 TITLE

*Hydrail Railway Transition in Canada: Technological, Operational, Economical, and Societal (TOES) Barriers and Opportunities*

## 2.0 INTRODUCTION

Hydrogen rail is a relatively new technology that is showing great potential to reduce emissions by using electrical energy generated through fuel cells using hydrogen to power the locomotive's electric engine. In contrast to other more conventional electrified rail systems using overhead catenary lines, hydrogen fuel cells allow for the energy to be generated on-board the vehicle. Recent studies have shown that hydrogen powered trains could be a good alternative for passenger rail -- and in some cases long-haul freight rail -- where infrastructure and fuel supply challenges can be addressed in collaboration with government, industry and utility companies.

## 3.0 BACKGROUND

The Government of Canada is committed to tackling climate change and leading the transformation towards a low-carbon economy. In December 2016, the Government of Canada adopted the Pan-Canadian Framework on Clean Growth and Climate Change, which is a plan to grow the economy, reduce greenhouse gas emissions, and build resilience to the impacts of a changing climate. The plan reflects Canada's international commitments to reduce Canada's greenhouse gas emissions to 30% below 2005 levels by 2030, and is built on four pillars: pricing carbon pollution; taking action to reduce emissions in each sector of the economy; adapting to climate change; and supporting clean technologies, innovation and jobs.

Canada's rail industry accounts for 4% of Canada's transportation GHG emissions; roughly 6.8 Mt, and the sector's share of transportation emissions has fluctuated between 4% and 5% between 2005 and 2016. While the sector has positively improved its emissions intensity, absolute emissions from the sector continue to grow in contrast to Canada's climate commitments.

Hydrail technology which emits no emissions at the point of use, has seen commercial deployment throughout the world, with the most notable example being the Alstom Coradia iLint hydrail passenger trains being operated in Germany. To date hydrail rail freight applications have been very limited. This research project has the goal of acquiring a broad appreciation of the many factors that impede the development of a hydrogen based propulsion system for the railway industry. The impedances and benefits are likely found in technical, operational, economic, and societal barriers and opportunities.

## 4.0 OBJECTIVES

Transport Canada is requesting Professional Services for a consultant to explore the technical, operational, economic, and societal factors (TOES) that would affect the viability of transitioning a diesel driven railway industry in Canada to a railway system centred around hydrogen.

The consultant is tasked to conduct a high level study to identify the technical, operational, societal, and economic determinates for this transition. The outputs of this study will be documented in a report that defines the challenges and the opportunities inherent in ~~the~~ this process. The report will serve as a tool that frames the transition and sequences the necessary changes.

The time horizon for this outlook are immediate (2030) and long term (2050). We can hypothesize that a complete transition would occur by 2050.

## 5.0 RAILWAY INDUSTRY SEGMENTATION

Converting current railway operations in Canada to Hydrail would be a significant undertaking involving many layers of government along with multiple industry stakeholders, such as fuel suppliers, utilities, shippers, transit companies, railways etc. A significant and concerted effort would be required to enable the implementation of this technology.

Collaboration and government leadership are necessary to de-risk the industry up-take and provide guidance related to the development of codes, standards -- and in some cases -- adapt or modernize regulatory frameworks. As result, a realistic progression over a number of years would be necessary and to that end we propose to separate the following railway operations into a logical progression sequence to facilitate Hydrail transition, as follows:

5.1 Commuter Passenger Rail
5.2 Inter-City Passenger Rail Service
5.3 Freight Service, restricted to locomotive switchers in railway/marshalling yards.
5.4 Long Distance Trans-Canada Freight Service and Regional Service

## 6.0 SCOPE OF WORK

To answer the preceding questions, the scope of work will include five principal components:

1. Document a thorough literature review of all Hydrail projects around the world.
2. Identify the technical and/or engineering challenges that hinder the progression of Hydrail for the entire spectrum of possible railway operations in Canada
3. Identify the operational impacts, both positive and negative, that a Hydrail system would impose on an operating railway, whether passenger, freight, or commuter service. To the extent possible, highlight: accompanying modifications in codes and standards that would be required to realize such changes; and new training requirements for railway personnel (e.g. operators, maintenance operations support personnel.

4. Identify all the possible capital and operating cost components that a Hydrail transition would represent to permit interested parties to accelerate the transition, including securing Canadian-wide hydrogen supply and infrastructure.
5. Identify all the societal benefits that would accrue to Canada, should a successfully industry wide Hydrail transition occur, including ones related to investment in Canadian technology, reduction in CAC emissions, etc.

## 6.1 Review of Global Hydrail Projects and Lessons Learned

This initial task is an abbreviated review of all Hydrail development projects around the world:

- Identification of all Hydrail initiatives around the world.
- At a high level, identify the key drivers which triggered decisions to advance the projects beyond the conceptual stage and into development/deployment.
- Discuss the role of government, private industry, investors, and other project parties for initiating each project, and working towards a functioning hydrail system.
- Reflect on key differences between the launch of these projects and the economic, geographic, and political environment in Canada – including the integrated nature of the North American railway systems.

## 6.2 Identity and Assess Technical / Engineering Challenges for Hydrail Transition

The contractor will devise a definition of hydrail systems, identifying the constituent components, and provide an overview of the necessary technical and engineering challenges that will require research and development funding investment to overcome to create a platform that will allow manufacturers to build the locomotive equipment and operators to operate such equipment.

At a high level, evaluate the technological readiness and commercial readiness of hydrail systems including a description about the current status of key components of the hydrogen power system. Discuss further research and development needed to raise the technology readiness to a level that commercial readiness can advance. Identify the differences in the readiness of various components if applicable. Illustrative examples of components to consider are:

- Hydrogen fuel cell
- Hydrogen storage tanks
- Batteries
- Control system for managing electricity production against tractive power demand
- Control system for managing electricity production for auxiliary loads
- Temperature control system

The discussion must show the differences of hydrogen power trains in locomotives for each of the following usage cases;

- Commuter Passenger Rail
- Inter-City Passenger Rail Service
- Freight Service, restricted to locomotive switchers in railway/marshalling yards.
- Long Distance Trans-Canada Freight Service and Regional Service

## 6.3 Identify Operational Impacts to the Rail Industry

The last major transformation in the railway industry in Canada happened after WWII, when steam locomotive engines were replaced with diesel driven locomotives. This transition initiated a major change in the economics of the railway industry. The changed economic structure was the result of operational changes that impacted both the operating costs and the capital costs of the industry. For example:

- Infrastructure changes included the elimination of water and coal depots along the right of way (ROW), and also the elimination of many smaller railway marshalling yards and maintenance centres, such as occurred in cities like St. Thomas in Ontario.
- Operational changes also occurred, such as the obsolescence of coal shoveling manpower. Maintenance practices also changed substantially.
- New codes, standards, safety regulations, and training programs had to be devised and implemented to accommodate this transition.

The rate of these changes was very rapid. When the last steam engine was retired in Canada, it was a scant period of less than 15 years that ushered in the diesel engine. TC is requesting that the contractor anticipate the range of operational changes that would be required for a change from diesel to hydrail.

One very significant change may derive from the need to distribute the locations of hydrogen fuel depots along the ROW. This will have a significant impact on both labour and infrastructure costs: personnel to operate and maintain the hydrogen fuelling equipment; and the number of fuel depots that will be determined by the range limits for hydrail locomotives. Another aspect to be considered is the new skillsets that railway companies will require to operate hydrail locomotives, maintain hydrogen equipment and provide support services for railway operations such as refueling.

- Commuter Passenger Rail
- Inter-City Passenger Rail Service
- Freight Service, restricted to locomotive switchers in railway/marshalling yards.
- Long Distance Trans-Canada Freight Service and Regional Service

## 6.4 Hydrogen Source, Supply and Distribution System

A robust hydrogen production, distribution and fuelling network is essential to unlock the potential of hydrogen as a fuel for rail operations. Locomotives consume significant amounts of power and it is expected that immense volumes of hydrogen (gaseous and in some cases liquid) would be required to sustain hydrail operations. Examine the following parameters with the knowledge that the goal is to have an understanding of the order of magnitude:

- Describe the current hydrogen production capabilities within Canada and profile the source of hydrogen feedstock.
- Estimate the volume of hydrogen required to sustain hydrail operations.
- Discuss the current technological and commercial readiness of hydrogen production systems at a scale needed to supply sufficient hydrogen for hydrail operations.
- Estimate level of effort to increase Canada's hydrogen production capabilities to meet this demand, assuming a logical expansion of Canada's existing hydrogen feedstock.
- Estimate the level of effort to meet the hydrail demand using net-zero carbon hydrogen feedstock only.
- Discuss the scale of the undertaking to expand distribution to supply hydrogen for the rail system and the associated cost.
- Discuss the potential challenges related to shipping and storage of hydrogen (gaseous, liquid) as it relates to the movement of Dangerous Goods.

The analysis described above must be done for each of the following use cases in order to impart understanding of the different scales required.

  - Commuter Passenger Rail
  - Inter-City Passenger Rail Service
  - Freight Service, restricted to locomotive switchers in railway/marshalling yards.
  - Long Distance Trans-Canada Freight Service and Regional Service

### Regulatory Impacts of Operational Changes

The identification of changes in the infrastructure and operations will also enable the researchers to highlight those changes that will require a regulatory review and if needed, amendment to support the wide-scale deployment of the technology (versus a one-off pilot applying for a temporary exemption).

## 6.5 Identify Affected Operational and Capital Cost Components

It is not realistic to expect the contractor to estimate accurately the scope of change in the operational and infrastructure costs for the transition to hydrail. However, the identification and assessment of the cost components is required to allow future researchers to develop reasonable estimates in future studies that may be commissioned by Transport Canada. The list below is very limited and not exhaustive. It is expected that the product of this research will include a more comprehensive list. Furthermore, the client is aware that any estimates will be speculative at best and extremely sensitive to economies of scale.

**Operational:**

The following operating cost components are required:

1. Operations: labour to operate the fueling depots along the ROW
2. Maintenance: labour to maintain the hydrail locomotives
3. Annual consumption fuel hydrogen
4. Annual displacements of diesel resulting from hydrogen uptake

**Capital:**

The following capital cost components are required:

1. Using the costs for existing diesel units as a baseline, hypothesize the change in costs for hydrail locomotives assuming that their design is mature and are generaly available from locomotive manufacturers. Consider the following cost elements:
    a. Depreciation
    b. Life cycle; price of a new unit, the necessary remanufacturing events throughout its useful life, and scrappage.
    c. If these hydrail locomotives can be retrofitted to extend service life, please provide estimated cost and incremental extended service life span\Cost of retrofitting versus new builds, and timelines for fleet turnover

2. Additional train units, if necessary
3. Hydrogen Distribution Network: number of fueling depots along ROW. Expressed as <u># of depots / x kms? </u>Please differentiate the required capacity levels for ROW and major terminal depots, if this will be feature of the transition to hydrail .

## 6.6 Benefits to Environment and to Canadian Society

Hydrail transitionhas as its primary objective, is the reduction of diesel locomotive GHG and CAC reductions. In addition, another societal benefit to society is the development of a Canadian Hydrail industry that can export this technology and become a valuable Canadian industry for the competitiveness of the Canadian economy.

Environmental Benefits

With the assistance of Transport Canada (TC has a data base containing the litres of diesel consumed by the rail industry), the contractor will document and quantify the Greenhouse Gas (GHG) **Emissions** and Criteria Air Contaminants (**CAC**) reductions achieved by converting diesel powered locomotives to hydrail powered locomotives. Current pricing values for carbon emissions will be used to quantify annual GHG emissions.

Industry Benefits

To the extent that data exists, the contractor will provide estimates of global GHG emissions from the railway industry and the reliance on diesel locomotives. TC will assist the contractor in obtaining this value to assess the export potential of Hydrail. While this benefit is not traditionally used in a benefit cost analysis to determine the return investment potential for government, it is a noteworthy benefit for identification for this exercise. Some suggested benefits could include employment generation, GDP growth and Canadian export potential.

## 6.7 Contextual Overview for Hydrail Transition

In this section of the report, the consultant will devise a “Road Map” to explore how the the Hydrail transition could unfold. This is not intended to describe a step by step transition process, but rather, an assessment of how the following inter-dependent transition elements could initiate the process.

Address the rationale to consider hydrogen as a fuel for rail transport in Canada, as partly to meet challenges, and partly to realize opportunities, such as: Climate change (CCG strategy, and international commitments to forestall catastrophic global warming)

- Air pollution and human health
- Industrial stagnation vs. economic diversification and clean growth
- Sustainable development
- Market competitiveness, domestically and internationally

Qualitatively describe the contribution that a shift toward hydrail could make.

Provide a brief review of the history of past energy system transitions, focusing on key drivers of change. Demonstrate that history tends to show that end-use energy innovations are the driver for the development of new energy supply chains.

Deliver a contextual review of past and present energy systems supporting the railway industry, noting historical developments; for example:

- while electric locomotives were first deployed in the 1830s, shortly after steam locomotives, electricity failed to displace coal for steam power;
- by contrast, diesel eliminated coal within a couple of decades of its initial deployment in the 1940s.

Identify the characteristics of transitions that occurred and those that did not within the railway sector.

Assert a value proposition that drives a propulsion energy system transition in Canada's rail transport sector from diesel to hydrogen (leaving existing direct electrification systems unchanged); revised according to industry literature review.

Introduce an ambitious vision of a transition from diesel to hydrogen from 2030 and 2050.

This vision will serve as a reference to guide the various analyses to be scoped in the sections of the report.

## 7.0 IMPLEMENTATION APPROACH

### 7.1 Tasks

The project will be implemented by the Consultant in seven phases with the completion of the following tasks:

Task 1- Literature Review
Task 2- Technical Review
Task 3- Impact on Railway Operations;
Task 4- Impact on Operating and Capital Costs
Task 5- Society Benefits
Task 6- Contextual Overview for Hydrail Conversion
Task7- Delivery of Draft Report
Task 8- Delivery of Final Report

### 7.2 Meetings and Work Plan

After contract award, a project kick-off meeting will be held through teleconference with the Project Steering Committee to review and confirm the project tasks and schedule, and introduce the project participants and their roles. Additional teleconference meetings will be help upon completion of project milestones. For each meeting, the consultant will present the progress made and will prepare minutes. These minutes will be prepared in electronic format and emailed to the Transport Canada Technical Authority.

The consultant will produce an overall work plan and activity schedule within 10 working days of the effective contract commencement date. These are to be submitted to the Transport Canada Technical Authority for review and approval. The work plan is to include the dates for submission and review of milestones and the draft and final reports.

## 8.0 METHODOLOGY AND INVESTIGATION TEAM

8.1 Methodology
The bidder will submit a methodology in his/her proposal that will describe how the following research phases will be conducted:

Literature Review
Technical Review
Impact on Railway Operations;
Impact on Operating and Capital Costs
Society Benefits
Contextual Overview for Hydrail Conversion

8.2 Investigation Team
The bidder is required to assemble an multi-disciplinary team of investigators possessing the techical expertise to perform: an engineering design assessment, a regulatory review with recommendations, and an economic assessment of Hydrail purported savings in capital and operating costs.

The core team of investigators, ideally will be composed of five to seven investigators. It will be led by the Project Director and the Project Manager whose education, professional engineering certification, and experience will be sufficient to direct and manage this project. Supplementing them will be at least three to five other core members: one or two other engineers; one or two experts in transportation safety regulations; and one or two experts in estimating capital and operating costs in transportation systems. The composition of this core group will be evaluated. Additional team members may be drawn in to provide supplmental specialized expertise. If the bidder proposes to include these additional resources, these team members must be identified and their accredications, education, and experience presented.

## 8.0 INTELLECTUAL PROPERTY

The Crown will own the foreground intellectual property arising from work under this in accordance with Appendix C: Section 4.1 of the federal policy on Title to Intellectual Property Arising from Crown Procurement Contracts on the grounds that the main purpose of the Crown procurement Contract, or of the deliverables contracted for, is to generate knowledge and information for public dissemination.

## 9.0 CONSULTANT PROJECT CONTROL

The Consultant must employ a critical scheduling method to monitor the project timelines, cost and resources. Budgets for each project element must be prepared at the start of the project and monitored to ensure that the resources available are compatible with the estimates of what is required to complete the work.

## 10.0 TRANSPORT CANADA SUPPORT

Transport Canada will be responsible for the following during the course of the work:

(i) Creating the Project Steering Committee that will include, among others, project sponsors and principal stakeholders. Other organizations, including those providing specialized expertise, may be invited to join the Committee.
(ii) Convening Project Steering Committee meetings for each phase.

(iii) Distributing to the Committee members necessary reports or other materials delivered by the Consultant.
(iv) Providing feedback, as required, to the Consultant, and accepting and approving Consultant deliverables.
(v) Making available to the Consultant, previous High Speed Rail Studies prepared for Transport Canada. These will be necessary to evaluate the potential public benefits generated by a Hydrail system.

## 11.0 DELIVERABLES

### 11.1 Interim Reports

All, including interim reports must be submitted in electronic Microsoft Word format to the Project Authority for review and acceptance. The interim reports will be submitted following the completion of the following activities:

Draft Literature review
Final draft for review
Final report

The interim reports must include methodology, data, results, conclusions, references, and recommendations.

### 11.2 Progress Reports

Monthly Progress Reports must be submitted to the Project Authority electronically, no later than the 14th day of each month. A monthly progress meeting may also be held either by telephone or at a location to be specified by the Project Authority.

### 11.3 Final Report

The Consultant must must produce and submit a professionally written and edited Final Report that summarizes the findings of the work described in Section 7.0, "Detailed Work Description", to the Project Authority. Consultant

The Consultant must also prepare and submit with the Final Report a Power Point presentation summarizing the contents of the Final Report that will be used for general project dissemination. On acceptance of the Final Report, the Consultant will present the findings of the report to Transport Canada in Ottawa.

Two electronic versions of the report are required by email or flash drive. The first must be produced as or converted to a Microsoft Word (version 2013) document. The second must be an Adobe portable document format (pdf) file.

### 11.4 Delivery Schedule for Final Report

The Consultant must submit the draft Final Report to the Project Authority, who will submit it to the Committee for review. Feedback and comments will be provided to the

Consultant by the Project Authority. The Final Report submission and review schedule will be as follows:

(i) The Consultant must provide one printed copy and one copy in Microsoft Word of the draft Final Report, including the presentation summary, a PDF abstract and key words.

(ii) Technical comments on the draft Final Report will be given to the Consultant three (3) weeks after receipt.

(iii) The Consultant will provide a revised version of the draft Final Report within three (3) weeks of receipt of technical comments.

(iv) Comments on the revised draft Final Report will be provided electronically to the Consultant within three (3) weeks of submission of the revised draft Final Report following the a second technical review.

(v) The Consultant must provide a final version of the draft Final Report within three (3) weeks of receipt of the second set of editorial comments.

(vi) The Consultant must provide two copies of the Final Report within two (2) weeks of receipt from the Project Authority of written authorization to proceed with printing of the Final Report.

### 11.5 Webinar

The consultant will present the finding of the report in Webinar format to invited members of the Federal Government. The Webinar will reflect the results of the Consultant's work and will solicit thoughts, additional information, and feedback from Innovation Centre's collaborators to be considered for inclusion the final version of the report.

For control purposes, all electronic copies of the report will be delivered to Richard Zavergiu, Innovation Centre Project Officer:

Phone Number: 450-928-4387
Email Address: richard.zavergiu@tc.gc.ca

## 12.0 COMMENCEMENT OF WORK AND DELIVERY DATE

The work will commence on Contract Award with the final deliverables due three months after project kick-off.

## 13.0 CONSULTANT PROJECT MANAGER

The consultant will appoint a senior staff member to assume project management responsibilities. That person will be the principal contact person with the Transport Canada Project Authority.

## 14.0 – Schedule and Expected Level of Effort

The duration and schedule for each phase of the project, including startup, are projected as follows:

| **Tasks** | Estimated level of effort from the date of contract award | Estimated delivery date |
|---|---|---|
| **Task 1 - Project Kick-Off**<br><br>• Kick Off Meeting | 10 days | |
| **Task 2 – Introduction and Literature Review** | 1st month | |
| **Task 3 – Draft for review** | 2rd month | |
| **Task 4 - Final draft for review** | 3rd month | |

## 15.0 TRAVEL AND LIVING EXPENSES

There is no requirement for travel during the investigation of the report. It is expected that the consultant will present the report findings to Transport Canada virtually.

## 16.0 SUITABILITY & REPLACEMENT OF RESOURCES

Suitability of Resources

The Consultant will render all services required in the Terms of Reference and the Consultant's proposal. The resource(s) assigned by the Consultant must be capable of performing at a level of competence acceptable to the Project Authority (PA). Should the Consultant's resource(s) be considered unsuitable and upon written notice from the Project Authority, the Consultant must provide suitable replacements. Failure to do so will result in the contract being terminated.

Replacement of Resources

The Consultant must provide the services of the personnel named in the contract to perform the work, unless the Consultant is unable to do so for reasons beyond his/her control. Should the Consultant at any time be unable to provide the services of the resource (s) named in the contract, the Consultant shall be responsible for providing replacement personnel, at the same cost, who shall be of similar or greater ability and attainment and whom shall be acceptable to the Transport Canada PA. In advance of the date upon which replacement resources are to commence work, the Consultant shall notify, in writing, to the Transport Canada PA the reason for the unavailability of the resource(s) named in the contract. The Consultant shall then provide to the Transport Canada PA the name(s) of the personnel and an outline of the qualifications and experiences of the proposed replacement(s). Any replacement personnel will be evaluated in the same time. Under no circumstances shall the Consultant allow performance of the services by the replacement resources that have not been authorized by the Transport Canada PA.

## 17.0 DEPARTMENTAL AUTHORITIES

**Contracting Authority**
Martin Poirier
Contracting Specialist
Transport Canada, Material & Contracting Services
95 Foundry St. - Heritage Court Bldg.
Atlantic region – Moncton NB E1C 5H7
Tel: (506) 377-2561
E-mail : martin.poirier@tc.gc.ca

*The Contracting Authority is responsible for the management of the Contract and any changes to the Contract must be authorized in writing by the Contracting Authority. The Consultant must not perform work in excess of or outside the scope of the Contract based on verbal or written requests or instructions from anybody other than the Contracting Authority.*

**Project authority**
Richard Zavergiu
Innovation Centre Project Officer
Transport Canada
Phone Number: 450-928-4387
Email Address: richard.zavergiu@tc.gc.ca

*The Transport Canada Departmental Representative is the representative of the department or agency for whom the Work is being carried out under the Contract and is responsible for all matters concerning the technical content of the Work under the Contract. Technical matters may be discussed with the Transport Canada Departmental Representative; however the Transport Canada Departmental Representative has no authority to authorize changes to the scope of the Work. Changes to the scope of the Work can only be made through a contract amendment issued by the Contracting Authority.*

# RAIL PATHWAYS INITIATIVE

## PHASE 1: LANDSCAPE DOCUMENT

August 2020

POLLUTION PROBE
CLEAN AIR. CLEAN WATER.

DELPHI GROUP

FINAL DRAFT

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# Executive Summary

Canada's rail sector currently provides a low-carbon means of transporting both goods and people throughout the country. In terms of freight transport, the carbon intensity of rail is roughly four times lower than that of on-road freight movement. Despite being responsible for almost half of all freight movement within the country, the sector only accounts for 4% of total transportation-related greenhouse gas (GHG) emissions. However, as Canada and other nations accelerate transportation decarbonization efforts towards the achievement of climate-related targets, further reductions in GHG emissions intensity from the sector will be necessary.

The purpose of Phase 1 of the Rail Pathways Initiative is to develop a landscape document which delivers on the following objectives:

- Develop a common understanding of the current state of rail sector decarbonization in Canada, which can be used as a tool for collaboration between industry and government;
- Create a repository of current federal, provincial and territorial GHG reduction legislative instruments and activities impacting the rail sector; and
- Contribute to next-phase work on a roadmap to achieving future GHG reductions in Canada's rail sector.

To help achieve these objectives an expert steering committee comprised of public and private rail sector stakeholders was convened. A major role of the Rail Pathways Steering Committee was to review and supplement an inventory of legislative instruments and activities that impact rail carbon intensity in Canada. Specifically, the inventory found in Sections 4 to 7 and in Appendices B to F, includes:

- Federal Instruments: Regulations, policies and programs being led at the federal level;
- Provincial Instruments: Regulations, policies and programs being led at the provincial level;
- Federal and Provincial Research, Development and Demonstration (RD&D) initiatives: specifically, in areas such as technologies, fuels and feasibility assessments;
- Canadian Rail Industry Activities: GHG reduction activities in the areas of efficiency, alternative fuels, alternative propulsion and infrastructure; and
- International rail GHG reduction landscape and best practices.

To help with the identification of gaps and to illustrate areas of strength, GHG reduction mechanisms are categorized into five areas: fuel efficiency, alternative fuels, alternative propulsion, infrastructure and modal shift. Currently available and emerging technologies within these categories are touched upon in this document, although an assessment of the GHG reduction potential and barriers to the implementation of various technologies and practices in a Canadian context will be conducted in Phase 2 of the Pathways Initiative.

In terms of legislative instruments related specifically to rail decarbonization, most provincial and territorial legislation that impacts rail carbon intensity is not rail specific, or even transportation specific, but rather broadly addresses GHG emissions through climate action plans and strategies. At the federal level, however, a series of overarching policies are working to reduce GHG emissions from the rail sector by driving a shift towards lower emitting fuels and technologies, as well as towards lower emitting types of transportation.

These include the Pan-Canadian Framework on Clean Growth and Climate Change (which includes supportive policies such as carbon pricing and the Clean Fuel Standard) and the Canadian Environmental Protection Act (CEPA).

In terms of non-regulatory instruments impacting rail carbon intensity, of greatest impact is the MOU between the Railway Association of Canada and Transport Canada, which lays out GHG intensity reduction targets of 6% from 2017 levels by 2022 for Class I freight and intercity passenger operations, and reductions of 3% in the same timeframe for short line operations.

Direct support for RD&D initiatives within the rail sector in Canada was found to take place predominantly at the federal level. Similar to regulatory approaches, much of this support is not aimed specifically at the rail sector, but at transportation decarbonization efforts broadly. Canada is home to several organizations and bodies that explore and address the environmental performance of the nation's rail sector. These include the Railway Association of Canada (RAC), Transport Canada's Innovation Centre, the Rail Research Advisory Board of Canada (RRAB), and, in a limited capacity, the Canadian Rail Research Laboratory (CaRRL). Research was found to be focused largely on decreasing carbon emissions by improving efficiency and fuel switching. It appears that relatively little attention is being paid to assessing the GHG reduction potential of modal shifts to rail for the movement of goods and people – a gap that can perhaps be addressed in future RD&D efforts.

Canada's rail industry is engaged in decarbonization activities across all five categories explored in this document. The category being most intensively addressed continues to be fuel efficiency. This is being largely driven by fleet renewal and the implementation of software and data analytics related to energy and route optimization. The utilization of alternative fuels continues to advance as well, as does infrastructure expansion to enhance network capacity and fluidity.

Canada's geography, climate, economy and population density make its rail sector unique in a global context, however global rail decarbonization trends can nonetheless provide useful input into future planning. Many global jurisdictions are pursuing modal shifts to rail to support decarbonization efforts, and are conducting RD&D across all decarbonization categories to further reduce the carbon intensity of rail using emerging technologies and best practices. These jurisdictions include the UK, Germany, France (and the EU more broadly), Russia, China, India, Japan and Australia. Specific initiatives taking place in these and other parts of the world are profiled in Section 7.1 and Appendix F. Section 7.2 provides some examples of rail decarbonization roadmapping exercises currently taking place internationally. Elements from these examples may prove useful in informing a comparable exercise in Canada, which is to be the focus of Phase 2 of the Rail Pathways Initiative.

Phase 2 will use this landscape document to inform the development of a roadmap towards deeper GHG reductions. It will continue to involve federal and industry stakeholders and will further engage provincial and territorial governments to collaborate on the development of a common vision and path forward, including setting priorities for the near, medium and longer term.

# 1. Introduction

With over 49,000 route kilometres of track running from coast-to-coast, three national railway companies and numerous regional and shortline railways that carry freight, passengers or a combination of both, Canada's extensive rail network supports both the Canadian economy and the quality of life enjoyed by Canadians. Class I and shortline freight rail moves more than $328 billion worth of goods, and passenger rail moves 88 million people per year – 82.8 million passengers by commuter rail and an additional 5 million by intercity passenger railways.[1]

Close to a quarter of Canada's greenhouse gas (GHG) emissions come from the transportation sector.[2] Once dominated by passenger transportation, emissions from this sector are increasingly as a result of the movement of freight. Emissions from freight are projected to exceed those from passenger transportation by 2030.[3] On a tonne-km basis, rail is the most prevalent method of transporting freight domestically (44%, versus 33% by truck). Yet the rail sector accounts for only 4% of Canada's total transportation-related greenhouse gas (GHG) emissions[4], and 98% of these are as a result of the transportation of freight[5]. This is a testament to the fuel efficiency of this mode.

By consistently investing in efficiency and sustainability, Canada's freight railways have reduced their GHG emissions intensity by over 40% since 1990, and intercity passenger railways have reduced their GHG emissions intensity by about 55%.[6] These efficiency gains have largely been realized through locomotive engine upgrades. As Canada and the world moves towards deep carbon reductions, however, all sectors will need to look beyond efficiency to decarbonize the sources of energy they consume.

Since 1995, Transport Canada and the Railway Association of Canada (RAC) have signed four Memoranda of Understanding (MOU) to establish voluntary reduction targets for emissions produced by locomotives in Canada. The most recent of these, the 2018 – 2022 MOU, includes a commitment to collaborate on a "comprehensive pathway document for aligning government and industry efforts to reduce emissions produced by the railway sector." The two-phase Rail Pathways Initiative is intended to build off the successes achieved to date by the MOU via collaborative public-private efforts to explicitly target GHG reductions from Canada's rail sector. Phase 1 will catalogue ongoing and potential activities related to rail sector decarbonization that are led by industry and government, or collaborations between the private and public sectors. Phase 2 will develop a roadmap to deep GHG reductions in the rail sector through development of a comprehensive pathway document for aligning government and industry efforts to reduce emissions produced by the railway sector.

This landscape document represents the culmination of Phase 1 of the Pathways Initiative. Developed collaboratively with government and industry, it explores the current emissions-

1 RAC
2 In 2017, 24% of Canada's GHG emissions came from the transportation sector, per the National Inventory Report 1990-2017: Greenhouse Gas Sources and Sinks in Canada (Annex 10, Table 3).
3 Government of Canada. Canada's Second Biennial Report on Climate Change. (2016). https://www.canada.ca/content/dam/eccc/migration/main/ges-ghg/02d095cb-bab0-40d6-b7f0-828145249af5/3001-20unfccc-202nd-20biennial-20report_e_v7_lowres.pdf
4 Transport Canada
5 National Inventory Report 1990-2017: Greenhouse Gas Sources and Sinks in Canada
6 Canada's Railways - Helping Canada Stay on Track to Fight Climate Change, RAC, 2020 https://www.railcan.ca/wp-content/uploads/2020/05/RailCan_EnvironmentalBrief_Final.pdf

reduction landscape through an inventory and analysis of current policies, regulations, programs, operations practices, research and technology related to GHG reductions that impact the rail sector. It fulfils the following objectives:

1. Develop a common understanding of the current state of the rail GHG reduction landscape between industry and government that can be used as a tool for collaboration.
2. Create a repository of current GHG reduction federal and provincial instruments, programs and activities impacting the rail sector; rail sector activities to reduce GHG emissions; and the current research and technology landscape for reference by officials, researchers and technologists.
3. Contribute to next-phase work on a roadmap to achieving future GHG reductions, both in the rail sector and enabled by the rail sector.
4. Phase 2 of the Pathways Initiative will continue to engage stakeholders in government and industry in order to develop a roadmap to deeper GHG reductions in the rail sector. This will entail establishing a common vision, developing a framework for assessing GHG reduction opportunities and planning for the deployment of these opportunities over time to meet established targets.

# 2. Methodology

The Rail Pathways Initiative is a collaboration between the federal government (Transport Canada; Natural Resources Canada; Innovation, Science and Economic Development Canada; and Environment and Climate Change Canada) and the rail sector. It was spearheaded by Transport Canada in conjunction with the Railway Association of Canada and is supported by a project Steering Committee. The latter is comprised of public and private sector stakeholders along with members of the MOU Management Committee. A list of Steering Committee members is included in Appendix A.

The Steering Committee was first convened in April 2020. This initial meeting focused on finalizing a methodology for creating a repository of federal and provincial regulatory instruments, programs and activities impacting the rail sector; rail sector activities to reduce GHG emissions; and the current research and technology landscape. The following information was flagged for collection:

- Federal Instruments: Regulations, policies and programs being led at the federal level;
- Provincial Instruments: Regulations, policies and programs being led at the provincial level;
- Federal and Provincial Research, Development and Demonstration (RD&D) initiatives: specifically, in areas such as technologies, fuels and feasibility assessments;
- Canadian Rail Industry Activities: GHG reduction activities in the areas of efficiency, alternative fuels, alternative propulsion and infrastructure; and
- International rail GHG reduction landscape and best practices.

This Landscape Document is based on information provided by Steering Committee members from federal departments and from the rail industry; and from rail safety and environmental personnel in provincial and territorial governments with rail sector activity. Information collected through these means was supplemented by a literature review to ensure that what is reflected here is comprehensive, and to characterize the international landscape.

# 3. GHG Reduction Mechanisms for the Rail Sector

GHG reductions for the rail sector typically fall into one of five categories: fuel efficiency, alternative fuels, alternative propulsion, infrastructure and modal shift.

**Fuel efficiency** is commonly measured in litres of fuel consumed per passenger km for passenger rail, or per revenue tonne-kilometre (RTK) for freight rail. Measures around fuel efficiency may address infrastructure and equipment including engines, railcars, components or tracks; and software, such as for route and speed optimization and automation.

**Alternative fuels** are low-carbon alternatives to petroleum diesel. These include liquefied natural gas (LNG), which must be used in purpose-built LNG locomotives or existing locomotives that have been retrofitted; and biofuels such as biodiesel and renewable diesel, which can be blended with petroleum diesel up to the limits imposed by OEMs[7] and used to power regular locomotives. In some cases, alternative fuels require the use of specialized refueling infrastructure such as compressors and pressurized storage tanks and dispensers.

**Alternative propulsion** systems avoid the use of combustion engines and are instead powered by electricity derived from hydrogen fuel cells, local electrical grids, or batteries. These may provide motive power to trains on their own or in tandem with combustion technologies in the case of hybrid systems. Some alternative propulsion technologies necessitate specialized infrastructure such as overhead catenary wires or electrified third rails, or hydrogen distribution pipelines, compressors and storage tanks.

**Infrastructure** relates to Class I and shortline tracks, bridges, tunnels, road crossings, signalling devices and signage, spurs and sidings, along with stations, depots, terminals and container yards. Infrastructure may impact the value proposition of rail relative to other, higher-carbon modes of transport. For example, double-tracking can enhance network fluidity and shortlines and rail spurs can enable the more direct delivery of goods and mitigate the need for multi-modal last mile options such as on-road freight trucks.

**Modal shift** refers to the shift of freight and passenger transportation away from higher emitting modes including trucks, cars and air, to the more efficient rail. In particular, shifting freight from truck to rail represents an opportunity for significant GHG emissions savings, as trucks represent the greatest source of transportation emissions in the movement of goods, and GHG emissions from the movement of freight trucks has more than doubled in the past three decades.[8] While such a shift will be expected to raise the overall GHG emissions from the rail sector as a result of increased usage, because the emissions intensity of rail is lower than on-road and air, overall transportation emissions will be reduced. In addition to GHGs, it is also necessary to account for impacts on CAC emissions in the context of modal shift.

This report applied the above categorizations when cataloguing activities that contribute to the reduction of GHG emissions from the rail sector, including RD&D, actions on the part of the Canadian rail industry, and international activities. Prior to addressing GHG reduction activities, however, it examines the regulatory and non-regulatory federal and provincial instruments that overlay them.

7 OEM warranties limit blending to 5% for biodiesel and 30% for renewable diesel.

8 Modal Optimization as a Contributor to Reducing GHG Emissions in Canada. The Delphi Group and Pollution Probe for Natural Resources Canada, March 2020.

With regard to specific technologies that can be utilized by rail stakeholders to further reduce the carbon intensity of this mode, the table below provides a high-level overview of available and emerging options for rail decarbonization.

*Table 1: Technology-Related Measures to Reduce Rail Carbon Intensity*

| Technology | Category | Description | Readiness Level* |
|---|---|---|---|
| **Engine and drivetrain upgrades** | Fuel Efficiency | Fleet renewal and retrofitting using higher efficiency engines have been the primary means of realizing significant fuel efficiency gains and GHG emissions reductions in Canada's rail sector. | High |
| **Energy Management Software** | Fuel Efficiency | Software (e.g., Trip Optimizer) related to fuel and energy management systems, route and speed optimization, traffic management and scheduling, track monitoring, and cargo and container optimization can use data analytics to significantly enhance fuel efficiency within individual trainsets and across rail networks. | High |
| **Hybridization** | Fuel Efficiency | Diesel-electric hybrid locomotives containing energy storage systems such as batteries, flywheels, and/or supercapacitors properly sized to capture braking energy have been shown to significantly reduce fuel consumption through the use of regenerative braking. | Medium |
| **Lightweighting** | Fuel Efficiency | Lightweighting of railcars and components (e.g., through the use of composite materials, lightweight alloys or 3D printed components) can significantly decrease trainset weight, thereby enhancing fuel economy. | Medium |
| **Aerodynamics** | Fuel Efficiency | Efficiency enhancements through enhanced aerodynamics (e.g., streamlined noses, surface microstructures, railcar gap reductions, enhanced coupling systems, bogie and nose fairings) are most pronounced in high speed rail applications, however efficiency gains can also be realized within lower speed networks. | Medium |
| **Passenger Hydrail** | Alternative Propulsion | Passenger hydrail offers zero operational GHG or air pollutant emissions through the use of compressed hydrogen fuel coupled with a fuel cell stack and batteries. In essence it is a mode of electric traction in which electricity is generated onboard and used to charge batteries which in turn power electric motors. While light passenger hydrail has begun commercial operation in Germany, key barriers include equipment costs due to low production volumes, achieving power output levels required for heavy passenger rail applications, and fuel availability. | Medium |

| Technology | Category | Description | Readiness Level* |
|---|---|---|---|
| **Wired Electrification** | Alternative Propulsion | Electrification using overhead catenary lines negates the need for large batteries or other types of energy storage to power electric motors. While this technology has no operational emissions, the carbon intensity of regional electrical grids must be factored into its GHG reduction potential. There are significant cost and geospatial barriers to implementation in a Canadian context. | Medium |
| **Battery Electrification** | Alternative Propulsion | Battery electric locomotives, which can operate independent of a continual source of power, are in the early stages of RD&D globally, and are not ready for commercial applications. However the use of battery electric switcher locomotives in railyards is actively being explored globally. | Low |
| **Freight Hydrail** | Alternative Propulsion | Freight hydrail utilizes the same suite of technologies as passenger hydrail. Like battery electric trains, the power produced by hydrogen locomotives is currently inadequate to move freight trains the distances required for commercial viability. However switcher locomotive applications are being explored in many jurisdictions including Canada. | Low |
| **Biodiesel** | Alternative Fuels | Biodiesel can be produced from a wide variety of plant and animal biomass, but lacks sufficient chemical similarity with petroleum diesel to serve as a drop-in fuel. Due to its chemical properties, the typical blend limit for biodiesel use in diesel locomotives is 5%, and this is even lower in extreme cold weather operations. Higher blends have been tested successfully but are not approved for use by engine manufacturers. | High |
| **Renewable drop-in biofuels** | Alternative Fuels | Renewable drop-in biofuels are exchangeable with diesel as they have sufficient chemical similarity with it. They are typically produced through upgrading oils derived from a wide variety of organic feedstocks. They are capable of being used in existing diesel locomotives, either on their own or in blends (although most engine manufacturers limit the blend rate to 30%). The major barrier to their implementation is high fuel costs stemming from a lack of production capacity. | Medium |
| **Compressed and Liquefied Natural Gas** | Alternative Fuels | CNG and LNG have been explored as alternative fuels in the rail sector over the past two decades. They can be used in purpose-built spark-ignition natural gas engines or in retrofitted dual fuel diesel engines as blends. | Medium |

**For the purposes of this document, technology readiness level refers to a given technology's current ability to contribute to the deep decarbonization of the rail sector from the perspective of both global technological development and the Canadian experience with the technologies. The GHG reduction potential and barriers to the implementation of various technologies will be assessed in Phase 2 of the Pathways Initiative.*

Regarding the five categories of rail decarbonization activity, it should be noted that there can be significant overlap between actions within a given category. For example, infrastructure enhancements can result in enhanced fuel efficiency through reducing the need for braking or idling.

Modal shift is closely related to the infrastructure category, as rail infrastructure significantly impacts the potential timeliness and geographic reach of rail transport. Neither of these categories encompass discrete technologies per se, but are influenced by extraneous factors such as physical geography, pre-existing transportation networks, and public policy. With regard to modal shift, one of the major advantages of on-road trucking relative to rail is the speed and precision with which goods can be delivered. If rail infrastructure is available which allows for shortened delivery timelines, the business case for shipment by rail improves. Further, if rail infrastructure is adequate to allow railway companies to deliver goods as close to their destinations as possible, this will also help to provide an environment in which increased modal shift to rail can occur.

# 4. Federal, Provincial and Territorial Instruments

Federal, provincial and territorial governments offer a variety of instruments that impact GHG emissions from rail. These include high-level policies; regulations targeting various aspects of railway operations; non-regulatory programs that incent emissions reductions from railway operations or a shift from more emitting modes to rail; and other instruments. These primarily consist of broad, overarching climate change policies and their associated regulations and initiatives that work through multiple avenues to reduce GHG emissions from rail via all five categories described in Section 3.

Key federal, provincial and territorial instruments are summarized here and described at greater length in Appendices B and C.

## 4.1 Federal Instruments Matrix

Of the federal legislative instruments that are most impactful on GHG emissions reductions from railway operations, only one is specific to the rail sector: the Railway Safety Act's Locomotive Emissions Regulations. While this acts primarily to target criteria air contaminant (CAC) emissions, it aims to reduce GHG emissions largely through idle-reduction measures. Most federal measures target transportation more broadly, or act at an even higher level to address fuels and emissions.

In terms of non-regulatory federal instruments that have an impact on the rail sector's GHG intensity, the 2018 MOU between the Railway Association of Canada and Transport Canada lays out GHG intensity reduction targets of 6% from 2017 levels by 2022 for Class I freight and intercity passenger operations, and reductions of 3% in the same timeframe for short line operations. Progress towards these targets is summarized and made public annually through the Locomotive Emissions Monitoring (LEM) Program.[9]

Transportation-specific policy impacts the rail sector via actions to enhance rail safety standards, support innovation to reduce emissions, and catalyze investment in efficiency within trade corridors through infrastructure funding and modal optimization. Innovation is also supported through non-transportation-specific instruments, such as Sustainable Development Technology Canada's (SDTC's) Sustainable Development Tech Fund.

Overarching policies such as the Pan-Canadian Framework on Clean Growth and Climate Change (PCF, which includes supportive policies such as carbon pricing and the forthcoming Clean Fuel Standard) and the Canadian Environmental Protection Act (CEPA) reduce GHG emissions from the rail sector by driving a shift towards lower emitting fuels and technologies, as well as towards lower emitting types of transportation.

### CEPA

The 1999 Canadian Environmental Protection Act spawned the Renewable Fuels Regulations which mandated an average renewable fuel content of 2% in diesel fuel. This will be replaced by the Clean Fuel Standard (CFS) under the PCF by 2022. The CFS will govern GHG emissions from fuel production and combustion by requiring reductions in life-cycle carbon intensity and will further incent a transition away from petroleum-based fuels.

9 Railway Association of Canada. Locomotive Emissions Monitoring. (2018). https://www.railcan.ca/rac-initiatives/locomotive-emissions-monitoring-program/

## PCF

The PCF has set out an approach for reducing emissions from the transportation sector, including a commitment for the federal government to "improve efficiency and support fuel switching in the rail sector." The Pathways Roadmap that will be developed in Phase 2 will contribute towards satisfying this commitment. The PCF also included among its four transportation pillars "support the shift from higher to lower-emitting types of transportation, including through investing in infrastructure,"[10] which would include investments in rail infrastructure as described in Section 3.

Carbon pricing is a central component of the PCF that will impact on all of the GHG reduction mechanisms addressed in Section 3 by dis-incentivizing the consumption of petroleum-based fuels. This includes both measures intended to reduce overall fuel use, such as fuel-efficiency, infrastructure and modal-shift related measures; and also measures that promote non-petroleum based fuels including alternative fuels and alternative propulsion. Under the 2018 Greenhouse Gas Pollution Pricing Act, provinces had an opportunity to design and implement their own carbon pricing plans. British Columbia, Québec, Nova Scotia, Prince Edward Island, and Newfoundland and Labrador did so. For those provinces that did not – Alberta, Saskatchewan, Manitoba, Ontario and New Brunswick – the federal fuel charge is in effect.

*Table 2: Federal Regulatory and Non-regulatory Instruments*

| Federal Department/ Organization | Policies | Regulations | Programs | Other |
|---|---|---|---|---|
| **Environment and Climate Change Canada** | ❖ Pan-Canadian Framework on Clean Growth and Climate Change<br>❖ Canadian Environmental Protection Act | ❖ Clean Fuel Standard<br>❖ Renewable Fuels Regulations | | ❖ Greenhouse Gas Pollution Pricing Act (Carbon Pricing)<br>❖ PCF Initiatives:<br>▶ Setting emissions standards and improving efficiency<br>▶ Shifting from higher- to lower-emitting modes and investing in infrastructure |
| **Natural Resources Canada** | | | ❖ SmartWay Transport Partnership | |
| **Transport Canada** | ❖ Transportation 2030: A Strategic Plan for the Future of Transportation in Canada | ❖ Locomotive Emissions Regulations (Railway Safety Act) | ❖ National Trade Corridors Fund | |
| **Sustainable Development Technology Canada** | | | ❖ Sustainable Development Tech Fund | |

10 https://www.canada.ca/en/services/environment/weather/climatechange/pan-canadian-framework/complementary-actions-reduce-emissions.html#3_3

## 4.2 Provincial and Territorial Instruments Matrix

Similar to the federal landscape, most provincial and territorial instruments that impact GHG reductions from the rail sector are not rail specific, or even transportation-specific, but more broadly address climate change. These include climate change strategies, carbon pricing, and renewable and clean fuel standards. Together with the associated regulations and programs, these policies work to shift the transportation sector, including the rail industry, towards lower emitting fuels and technologies in much the same way that the CEPA and PCF do, as outlined in Section 4.1.

Québec and British Columbia have been the most active in addressing GHG reductions from transportation through targeted policies and programs. Québec in particular is directly supporting both the development and uptake of low-carbon technologies, fuels and infrastructure in the transportation sector, along with a modal shift to rail through a number of initiatives, outlined in Table 3, below, and further described in Appendix C. BC is taking a broader, non-sectoral view of GHG reductions through its CleanBC strategy and addressing rail efficiency specifically in the context of trade corridors (including ports).

*Table 3: Provincial and Territorial Regulatory and Non-regulatory Instruments*

| Province | Policies | Regulations | Programs | Other |
|---|---|---|---|---|
| BC | ❖ Carbon Target<br>❖ CleanBC<br>❖ B.C. on the Move: A 10-Year Transportation Plan (2015) | ❖ Renewable & Low Carbon Fuel Requirements Regulation | ❖ Trade Corridors Initiative | ❖ Carbon Tax |
| AB | ❖ Transportation Business Plan 2019–23 | ❖ Renewable Fuel Standard | | |
| SK | ❖ Climate Change Strategy | | | |
| MB | ❖ Climate and Green Plan Implementation Act | ❖ Clean Fuel Standard | | |
| ON | ❖ Environmental Protection Act | ❖ O. Reg. 97/14: Greener Diesel Regulation | | |
| QC | | ❖ Renewable Fuel Standard | ❖ Electrification and Climate Change Fund<br>❖ Marine, Air and Rail Transportation Efficiency Assistance Program (PETMAF)<br>❖ Greenhouse Gas Emissions Reduction / Avoidance Program through Intermodal Transportation Development (PREGTI)<br>❖ Technoclimat | |
| NB | | | | |
| NS | | | | ❖ Cap and Trade Program |
| PE | | | | |
| NL | | | | |
| YK | | | | |
| NWT | | | ❖ GHG Grant Program for Buildings and Industry | ❖ Carbon Tax |

# 5. Canadian Research, Development and Demonstration Activities

Research, Development and Demonstration, or RD&D, facilitates the determination of achievable GHG reduction levels for the rail sector and the identification of actions and policies that will help the sector contribute to broader decarbonization efforts and targets.

Canada's federal, provincial and territorial governments conduct rail RD&D initiatives to support continual improvement in terms of environmental performance while ensuring the sector's continued economic competitiveness. Within Canada there also exists a robust ecosystem of organizations and bodies that explore and address the environmental performance of the nation's rail sector. These include the Railway Association of Canada (RAC), Transport Canada's Innovation Centre, the Rail Research Advisory Board of Canada (RRAB), and, in a limited capacity, the Canadian Rail Research Laboratory (CaRRL).

Direct support for RD&D initiatives in Canada takes place predominantly at the federal level. Much of this support is not aimed specifically at the rail sector, but at transportation decarbonization efforts broadly. Provincial support for RD&D, where applicable, tends to happen through non-regulatory programs (Section 4.2) and partnerships rather than being led directly by government. Provincial engagement on efforts related to transportation decarbonization also tends not to be aimed specifically at the rail sector, but at either the movement of freight or people.

Although the federal government and several provinces have identified shifting to higher efficiency modes such as rail as an effective means to reduce GHG emissions, there is little RD&D occurring focused on assessing and quantifying the GHG reduction potential of modal shifts to rail for the movement of goods and people. Increased engagement in this area could help to situate rail transport more concretely into climate-related plans, targets and programming, and could help to provide guidance on funding priorities to reduce the carbon intensity of the transportation sector.

*Table 4: Canadian RD&D Initiatives Related to Rail Decarbonization*

| Department, Jurisdiction | Fuel Efficiency | Alternative Fuels | Alternative Propulsion | Infrastructure | Modal Shift |
|---|---|---|---|---|---|
| Transport Canada (TC) | Transport Canada Innovation Centre (TCIC) | | | | |
| | Rail Research Advisory Board of Canada (RRAB) | | | | |
| | Clean Transportation System - Research and Development Program (CTS-RD) | | | | |
| | MOU between TC and the RAC for Reducing Locomotive Emissions | | | | |
| | | Higher-Concentration-Blend Lignin-Derived Diesel Fuels for Rail Applications (partnership between TCIC and NRCan's CanmetENERGY) | Electrification of the Freight Rail Sector in Canada: Review of the Feasibility, Costs and Benefits | Canadian Rail Research Laboratory (consortium led by TC) | |
| | | | Hydrail Switcher Locomotive Project (partnership between TCIC and ECCC) | | |
| Natural Resources Canada (NRCan) | | Economic and Environmental Benefits of Natural Gas Fuel for the Rail Sector in Canada | | | Modal Optimization as a Contributor to Reducing GHG Emissions in Canada |
| | | Evaluation of Liquefied Natural Gas Infrastructure Build-up for Supplying Fuel to the Rail Market in Canada | | | |
| Sustainable Development Technology Canada (SDTC) | Sustainable Development Tech Fund | | | | |
| | Energy Efficient Transit Propulsion Pilot Program Project | | | | |
| Ontario Ministry of Transportation | | | High Speed Rail in Ontario: Report by the Special Advisor for High Speed Rail | | |
| Metrolinx (Ontario) | Tier 4 locomotives | | | | |
| | Throttle Control Program | | | | |
| | Excess Idle Reduction | | | | |
| Canadian Urban Transit Research and Innovation Consortium (CUTRIC) | National Rail Innovation Initiative (partnership with TCIC) | | | | |
| Government of British Columbia (with industry and academic partners) | | The British Columbia Sustainable Marine, Aviation, Rail and Trucking (BC-SMART) Biofuels Consortium | | | |
| TransPod | | | TransPod Hyperloop | | |

# 6. Canadian Rail Industry Activities

Canada's freight rail industry is dominated by two Class 1 freight rail companies, Canadian National Railway (CN) and Canadian Pacific Railway (CP). CN and CP represent more than 95% of Canada's annual rail tonne-kilometres and more than 75% of the nation's tracks.[11] There are also numerous shortline and regional railways in operation throughout the country. Shortline railways feed traffic to and from the mainlines, and regional railways provide local service in areas that are not serviced by the mainline railways.

VIA Rail Canada is Canada's dominant intercity rail passenger service operator. Commuter rail service is provided by TransLink in Metro Vancouver, GO Transit in the Greater Toronto and Hamilton Area, and exo in the Greater Montreal Area.

Rail industry GHG reduction activities are summarized here and described at greater length in Appendix E.

Canada's Class 1 railways, both freight and passenger, have been active in the fuel efficiency space for well over two decades,[12] implementing strategies to enhance operating practices and employing fuel saving technologies such as telemetry, energy and route management software, and distributed power. Fleet renewal has included both refurbishment of existing locomotives and railcars, and the purchase of more fuel efficient trains, including Tier 4 locomotives[13] and new generation railcars.

In the past decade, technology has changed across the transportation sector at a rapid pace, with new innovations such as electric and hydrogen cars, buses, and medium-duty trucks gaining market share. This change is likely to accelerate if climate change mitigation continues to be a priority for industry and government.

In line with this broader shift, emission reduction measures in the rail sector have expanded beyond fuel efficiency. Alternative fuels have been and continue to be explored. Testing of renewable fuels has occurred, though the scope of these efforts has been limited by engine manufacturers, as some warrantees only cover use of blends of up to 5% for biodiesel and 30% for renewable diesel. One LNG demonstration project was also undertaken.

More recently, these activities have broadened to include alternative propulsion. Uptake of alternative propulsion technology has largely been limited to corporate fleet vehicles, though there is indication of interest in testing rail-based electric and hydrogen technologies. While most regional and shortline railways have been less active in pursuing GHG reduction initiatives, there are a few exceptions as noted in the table below.

11 Transport Canada https://www.tc.gc.ca/eng/policy/anre-menu-3020.htm

12 TC and RAC signed the first MOU to establish voluntary reduction targets for emissions produced by locomotives in Canada in 1995, and since then have signed three additional MOUs on this issue. Emissions from locomotives have been gradually falling since 1995.

13 While Tier 4 emissions standards are focused on reducing criteria air contaminant (CAC) emissions, Tier 4 locomotives also improve fuel efficiency. (https://www.aar.org/article/new-locomotive-technology-makes-freight-rail-more-efficient-environmentally-friendly/)

*Table 5: Canadian Rail Industry Activities*

<table>
<tr><th>Company</th><th>Fuel Efficiency</th><th>Alternative Fuels</th><th>Alternative Propulsion</th><th>Infrastructure</th><th>Modal Shift</th></tr>
<tr><td colspan="6">MAINLINE FREIGHT</td></tr>
<tr><td rowspan="3">CN</td><td rowspan="2">❖ Fuel saving technologies:<br>▶ Trip optimization<br>▶ Distributed power<br>❖ Fleet Renewal:<br>▶ Tier 4 locomotives<br>❖ Data collection to improve performance and fuel conservation:<br>▶ Horsepower Tonnage Analyzer (HPTA)<br>❖ Enhanced operating practices:<br>▶ Precision Scheduled Railroading</td><td rowspan="2">❖ Use of biodiesel blends<br>❖ LNG demonstration project:<br>▶ Retrofit of two 3,000-horsepower locomotives with engines that run on a fuel mix of 90-per-cent liquefied natural gas and 10-per-cent diesel.</td><td rowspan="2">❖ Investment in hybrid and electric vehicles</td><td>❖ Fleet Renewal:<br>▶ New-generation railcars</td><td>❖ Providing low carbon transportation and logistics solutions to customers</td></tr>
<tr><td colspan="2">❖ Double-tracking into the Port of Vancouver</td></tr>
<tr><td colspan="5">❖ Investments in rail infrastructure, network fluidity and efficiency</td></tr>
<tr><td>CP</td><td>❖ Fuel saving technologies:<br>▶ Trip optimization<br>▶ Automatic engine start stop systems<br>❖ Enhanced operating practices:<br>▶ Precision Scheduled Railroading</td><td>❖ Use of biofuel blends</td><td></td><td>❖ Fleet renewal:<br>▶ 8,500-foot High Efficiency Product Trains (HEP)<br>▶ Locomotive Modernization Program – upgrades to locomotive fleet</td><td>❖ Deployment of CP Fast Pass Technology</td></tr>
<tr><td colspan="6">REGIONAL AND SHORTLINE</td></tr>
<tr><td>Southern Railway of British Columbia (SRY)</td><td></td><td></td><td>❖ Supporting UBC research on hydrogen-fueled and battery hybrid switching locomotives</td><td></td><td></td></tr>
</table>

| Company | Fuel Efficiency | Alternative Fuels | Alternative Propulsion | Infrastructure | Modal Shift |
|---|---|---|---|---|---|
| Quebec North Shore & Labrador Railway (QNS&L) | ❖ Anti-idling devices installed on four locomotives | | | | |
| **PASSENGER** | | | | | |
| VIA Rail | ❖ Enhanced operating practices:<br>▶ Locomotive Engineer Scorecard including mapping optimal operating parameters, use of telemetry and simulator training to improve fuel efficiency<br>▶ Optimization of cycling, including deployment of most fuel efficient engines on most demanding routes<br>▶ Idle reduction policy, including no idling in yards<br>❖ Fleet renewal:<br>▶ Tier 4 locomotives with electrified rail functionality<br>▶ Fleet refurbishment of older F-40 locomotives with separate HEP engine<br>▶ Replacement of electrical systems in cars with more efficient options | ❖ Use of Biodiesel (B5)<br>❖ Test of B20 on one engine | ❖ Green strategy for fleet vehicle replacement: zero-emissions vehicles<br>❖ Future initiatives that are being explored include electrification | | ❖ Promotion of rail over less efficient modes<br>❖ Intermodal partnerships |

# 7. Literature Review: State of GHG Emissions Reduction Technology and International Best Practice

## 7.1 State of GHG Emissions Reduction Technology and Practices

When considering Canadian initiatives in the context of global rail sector GHG reduction initiatives, it is critical to be cognizant of geographic, climatic and demographic realities that make Canada unique on the global stage. Canada's massive size, rugged Rocky Mountain topography, harsh winters, low population density, and natural resource-based economy mean that low-carbon rail solutions being tested and implemented elsewhere may not be feasible here. While these factors may not preclude Canadian adoption of certain global practices and technologies, they may increase both the complexity and cost. A review of global trends nonetheless provides useful insights into potential areas for Canadian engagement. Likewise, Canadian initiatives may contribute to the evolution of global rail decarbonization efforts.

In recent years, the Government of Canada has led several national assessments focused on determining the technological readiness, environmental performance and costs of alternative fuels and propulsion technologies for both passenger and freight rail. These assessments, which are overviewed in Appendix D, have looked at both alternative fuels and alternative propulsion technologies: liquefied natural gas, biodiesel, renewable diesel, as well as electric and hydrogen-powered propulsion. This is largely consistent with fuels and technologies being examined outside of Canada as strong candidates to replace or reduce the use of diesel in the rail sector. Additional alternative fuels being assessed outside of Canada (primarily in the USA) include dimethyl ether (DME) and compressed natural gas (CNG). Canada's Renewable Fuels Regulations already require that all diesel fuel used south of 60°N latitude contain at least 2% renewable content. The Regulations will be replaced by the Clean Fuel Standard (CFS) in the near future, which will require increasingly higher blends of renewable content or shifts to other fuels.

Many global jurisdictions are pursuing the expansion of passenger and freight rail networks to reduce the carbon intensity of their transportation sectors and the number of cars and trucks on their roads.[14] The UK is targeting a diesel-free rail network by 2040, as part of its broader efforts to completely decarbonize its transportation sector by 2050. It is also aiming to significantly enhance rail's modal share for both passenger and freight movement through the construction of a high speed rail network connecting cities and regions where most freight and passenger movement is currently confined to trucks and cars. Germany is likewise pursuing an increase in freight modal share for its rail sector through its Freight Rail Masterplan and a series of related, targeted initiatives which delve into policy and economic measures as well as technological innovations. France and Russia have a long history of low-carbon national rail networks which leverage domestic energy sources, and both are currently in the process of major rail modernization efforts. China and India have prioritized the expansion of existing electric rail networks to facilitate the movement of their massive populations and reduce the carbon intensity of their transportation and industrial

14 See Appendix F for details on recent rail initiatives in global jurisdictions.

sectors. Trends like these allude to the fact that many national governments foresee an expanded role for rail in the decades ahead, having determined that it provides a low-carbon and cost-effective alternative to other modal options such as on-road transport and aviation. A key facilitator of increased modal shift to rail globally has been the expansion of high speed rail networks – a mechanism that may be worth exploring for certain regions of Canada with high rail traffic volumes.

A common challenge for rail networks throughout the world, especially in the face of increased modal shifts to the sector, is allocating an appropriate amount of rail infrastructure to accommodate high speed passenger rail while ensuring it does not conflict with the delivery of time-sensitive freight. Common measures being taken to address this globally include the construction of dedicated tracks and infrastructure for high speed rail lines, tunnels and bridges to minimize road crossings, and physical barriers to keep people, animals and other obstructions off of tracks. Double-tracking is a common measure globally to minimize delays and idling, providing right-of-way to high speed and/or time-sensitive trains and facilitating the bi-directional flow of rail traffic. This measure is becoming more common in Canada and is rolling out along busy stretches of its Class 1 railroads.[15] It can be complemented by the addition of sidings and loops, which also serve to mitigate downtime and create more efficient networks. While these measures help rail traffic flow more smoothly, spur construction can also be used to help rail access more destinations directly, thereby expanding its potential role in the movement of goods and passengers.

While foundational infrastructure like that referenced above is essential and should not be overlooked in terms of its direct and indirect GHG reduction potential, cloud-based and "smart" infrastructure is uniformly playing a bigger role in rail logistics around the world. Software related to fuel and energy management systems, route and speed optimization, and cargo and container optimization are leading to trains and rail networks that are increasingly autonomous and interconnected. As transport hubs continue to grow spatially and with regard to the volume of goods and passengers transported, software, big data and connectivity will allow the rail sector to play a bigger role in multi-modal transport systems around the world.

A comparable scope of measures are being taken in Canada and globally related to rail fuel efficiency. Engine and drivetrain upgrades, hybridization, lightweighting, energy management software, friction reduction and wheel-track lubrication, eco driving practices, higher capacity freight cars and even enhanced coupling systems and aerodynamics are being explored in multiple jurisdictions to yield incremental efficiency benefits. Numerous studies, consultations and demonstrations have recently been conducted within Canada (Appendices D and E) and globally (Appendix F) that focus on efficiency and operational measures to reduce the carbon footprint of rail transport. Global organizations such as the International Union of Railways (UIC), Union Of European Railway Industries (UNIFE), European Commission (EC), The European Rail Research Advisory Council (ERRAC), The Community of European Railway and Infrastructure Companies (CER), the Australasian Railway Association (ARA), Association of American Railroads (AAR), Federal Railroad Administration (FRA), and California Air Resources Board (CARB) have all conducted work and promoted measures related to efficiency enhancements to rail operations. Harmonized regulations and standards, enhancing rail interoperability across jurisdictional boundaries,

15 Trains Magazine. CN envisions slowly double-tracking main line from Edmonton to Winnipeg. (2018). (https://trn.trains.com/news/news-wire/2018/07/25-cn-envisions-slowly-double-tracking-main-line-from-edmonton-to-winnipeg)

and improving access to intermodal transport hubs can also indirectly serve to enhance efficiency and the environmental performance of rail operations.[16] In Canada, a key focus of recent industry-led decarbonization activities (Appendix E) has been fuel efficiency enhancements to further improve rail's carbon intensity advantage over on-road shipping. These efforts primarily revolve around fleet renewal and retrofitting using higher efficiency engines, and the implementation of software that optimizes energy, routes, and scheduling to reduce fuel consumption and time spent idling. These voluntary fuel efficiency efforts are in some cases expected to lead to fleet-wide GHG emissions reductions in excess of 20%.

The table below provides a high-level glimpse into recent global initiatives related to rail sector decarbonization. For further information on these and additional initiatives please refer to Appendix F.

***Table 6: International Rail GHG Emissions Reduction Activities***

<table>
<tr><th>Country</th><th>Fuel Efficiency</th><th>Alternative Fuels</th><th>Alternative Propulsion</th><th>Infrastructure</th><th>Modal Shift</th></tr>
<tr><td rowspan="5">USA</td><td>AAR - Transportation Technology Center</td><td></td><td>AAR - Transportation Technology Center</td><td></td><td></td></tr>
<tr><td>Wabtec - Sustainability improvements</td><td></td><td>BNSF - Battery electric high-horsepower freight locomotive prototype</td><td></td><td></td></tr>
<tr><td colspan="5">CARB - Technology Assessment: Freight Locomotives</td></tr>
<tr><td></td><td></td><td>Norfolk Southern - 999 battery electric switcher locomotive</td><td></td><td></td></tr>
<tr><td></td><td></td><td>Port of Los Angeles - Zero-Emission Track-Miles Locomotive Project</td><td></td><td></td></tr>
<tr><td rowspan="2">Germany</td><td></td><td></td><td>Alstom - Coradia iLint Hydrogen Powered Train</td><td></td><td></td></tr>
<tr><td colspan="5">Rail Freight Masterplan</td></tr>
<tr><td rowspan="3">UK</td><td></td><td colspan="4">National Rail - High Speed 1 (HS1)</td></tr>
<tr><td></td><td colspan="4">National Rail - High Speed 2 (HS2)</td></tr>
<tr><td colspan="5">Decarbonising Transport: Setting the Challenge</td></tr>
<tr><td>France</td><td></td><td></td><td>RATP - Battery-overhead electric maintenance locomotives</td><td></td><td></td></tr>
<tr><td>Japan</td><td>The Association of Japanese Private Railways – Commitment to a Low Carbon Society</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Russia</td><td></td><td></td><td>Electrification program: Trans-Siberian Railway</td><td></td><td></td></tr>
</table>

16 See Appendix E for information on specific initiatives led by these groups.

| Country | Fuel Efficiency | Alternative Fuels | Alternative Propulsion | Infrastructure | Modal Shift |
|---|---|---|---|---|---|
| China | | | Rail and transport hub electrification | | |
| India | | | Broad gauge rail network electrification | | |
| EU | UNIFE - RAILENERGY | | | | |
| | ERRAC - Rail 2050 Vision | | | | |
| | UIC and EC - CAPACITY4RAIL | | | | |
| | EC - SHIFT2RAIL<br>EC - White Paper: Roadmap to a Single European Transport Area | | | | |
| | ERRAC - Rail 2050 Vision | | | | |
| | CER and UIC - Moving Towards Sustainable Mobility | | | | |
| Global | UIC - Energy Efficiency Best Practice Workshops | | | UIC - Door to Door | |

## 7.2 International Roadmapping Initiatives

While initiatives addressing the environmental performance of rail operations are many and varied, comprehensive rail decarbonization roadmapping exercises are not. A review of global literature revealed only three recent examples of such exercises: Germany's Rail Freight Masterplan, the UK's Decarbonising Transport: Setting the Challenge, and The European Rail Research Advisory Council's (ERRAC) *Rail 2050 Vision*.[17]

Germany's Rail Freight Masterplan aims to increase the competitiveness and market share of rail freight transport while improving environmental performance. These efforts will ensure that the German freight rail sector is future-proofed and sustainable. The Masterplan contains discrete milestones for different elements of the freight rail network categorized under the following themes: infrastructure, digitalization, automation, innovative technologies (e.g., electric locomotives with batteries for wire-free last mile travel capabilities), multimodality enhancement, electric haulage expansion, reductions in infrastructure charges (i.e., charges for track access and service facilities – key inhibitors of increased rail freight transport), reducing the levy and tax burden on freight carriers, ensuring that labour, social and safety standards are harmonized across all transport modes, and improved employee training and professional development. Germany's current coalition government has committed to electrifying 70% of the freight rail network by 2025. It has also committed to additional measures, some of which are put forward in the Masterplan, to enhance the rail's share of domestic freight movement. While the Masterplan does not include any sector-specific targets for emissions reductions, it does stress that a low-carbon freight transport sector will help to achieve national targets. It also suggests potential perks for rail companies that use low-emissions technologies, such as preferential access to certain multimodal hubs.

While it lacks an exclusive focus on the rail sector, the UK's Decarbonising Transport: Setting the Challenge roadmap addresses rail in detail. This pan-modal initiative is actually a precursor to a more fulsome transportation roadmap – The Transport Decarbonisaton Plan (TDP) – which is currently under development. Setting the Challenge states that the primary means to decarbonize rail is through government intervention to expand electrification. TDP measures will also be informed by an industry-led effort which will provide information on the possible scale and pace of decarbonization out to 2050, the government-mandated timeline for net zero carbon transportation. The UK government recently issued a challenge to its rail sector to develop a pathway to eliminate the use of diesel-only trains by 2040. Subsequent industry recommendations were focused on increased electrification using overhead wires for freight, complemented by hydrogen and battery electric trains for passenger transport. It was stated that current hydrogen and battery powered trains lacked sufficient power for freight haulage.

The European Rail Research Advisory Council (ERRAC) released its Rail 2050 Vision roadmap in 2017. The Vision highlights the critical roles that rail plays in Europe's current and future economy and sustainability efforts, as well as key challenges (e.g., climate change) and opportunities (e.g., the creation of a Single European Rail Area (SERA), smart and low-carbon transport technologies). It states that rail is to "become the backbone of an intermodal 'Mobility as a Service' for passengers and 'Delivery as a Service' for goods" throughout Europe. The vision touts rail as both the safest and greenest mode of transport and discusses the many potential synergies between rail and increasingly digitalized, data-

17 For further information and links to the original documents please refer to Appendix F.

driven economies as well as smart technologies and cities. The technology-driven Vision looks beyond low-carbon fuels and propulsion and foresees the decarbonization of entire supply chains – not only within the rail sector but for the goods shipped by the rail sector. It envisions a role for trains not only in the movement of goods and people, but for electricity distributed by means such as dynamic wireless power transfer. It foresees automated, connected train operations, robotics, lightweight materials, and smart infrastructure playing major roles in improving rail efficiency. Being based in Europe, ERRAC sees rail becoming the default mode of transport for people within cities and for trips up to 1,000 km. To achieve these goals ERRAC calls for a seamless European rail research and innovation system, increased cooperation within the rail sector and between rail and other transport modes, and increased contributions from experts outside of the transportation sector.

# 8. Conclusion and Next Steps

Phase 1 has developed a comprehensive landscape document that includes: regulatory and non-regulatory instruments that support GHG emissions reductions from the rail sector; and activities on the part of government and industry to both study and implement GHG emissions reductions in five areas: fuel efficiency, alternative fuels, alternative propulsion, infrastructure and modal shift.

Phase 2 will use this landscape document to inform the development of a roadmap towards deeper GHG reductions. It will continue to involve federal and industry stakeholders and will further engage provincial governments to collaborate on the development of a common vision and path forward, including setting priorities for the near, medium and longer term.

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Stephan Wehr, Vice President, The Delphi Group

# Appendix B – Federal, Provincial and Territorial Legislation Related to GHG Reductions from Rail

| Department | Legislation | Regulation | Year | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|---|
| Alberta Ministry of Environment and Parks | Emissions Management and Climate Resilience Act | Renewable Fuels Standard Regulation | 2010 | The RFS requires a minimum annual average of 5% renewable alcohol in gasoline and 2% renewable diesel in diesel fuel sold in Alberta by fuel suppliers. | High - will encourage development and uptake of low carbon technologies in transportation sector | Link 1<br>Link 2 |
| BC Ministry of Environment & Climate Change Strategy | Climate Change Accountability Act | | 2007 | Targets a 40% economy-wide GHG reduction from 2007 levels by 2030, a 60% reduction by 2040, and an 80% by 2050. | Medium - will encourage development and uptake of low carbon technologies in the transportation sector. | Link 1 |
| | Carbon Tax Act | Carbon Tax Regulation | 2008 | Tax is currently $40 per tonne on fossil fuels (gasoline, natural gas and diesel). To be increased to $45 in September 2020, and to $50 in April, 2021. | High - this directly impacts rail companies (though currently only large industrial emitters can access funding based on revenues). | Link 1<br>Link 2 |
| | Greenhouse Gas Reduction (Renewable and Low Carbon Fuel Requirements) Act | Renewable & Low Carbon Fuel Requirements Regulation | 2008 | The previous regulation required fuel suppliers to decrease the average carbon intensity of their fuels by 10% by 2020. The updated regulation requires a 20% decrease by 2030. | High - will encourage development and uptake of low carbon technologies in the transportation sector. | Link 1<br>Link 2 |
| Environment and Climate Change Canada | Canadian Environmental Protection Act | Renewable Fuels Regulations | 2013 | These regulations require petroleum producers and importers to have an average of 5% renewable fuel content in gasoline and 2% renewable fuel content in diesel fuel and heating distillate oil based on volume. There is a system that can be utilized to trade renewable fuel compliance units. Will be replaced with Clean Fuel Standard in the near future. | High – renewable fuels are low-carbon relative to diesel, with the potential of being zero carbon. | Link 1<br>Link 2 |

| Department | Legislation | Regulation | Year | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|---|
| **Environment and Climate Change Canada** | Greenhouse Gas Pollution Pricing Act | Output-Based Pricing System Regulations | 2019 | An Act to mitigate climate change through the pan-Canadian application of pricing mechanisms to a broad set of greenhouse gas emission sources and to make consequential amendments to other Acts. The regulations cover reporting requirements for facilities that produce hydrogen, as well as carbon intensity standards for hydrogen produced via SMR and hydrogen used in various industrial applications.<br><br>As part of the Pan-Canadian Framework on Clean Growth and Climate Change, carbon pricing is central to Canada's plan to meet our emissions reductions targets. The federal carbon pollution pricing system applies in provinces and territories that request it or that do not have a system that meets the federal stringency requirements.<br><br>The federal carbon pollution pricing system consists of an output-based pricing system for large emitters and a regulatory charge on fossil fuels ("fuel charge"). The fuel charge applies to a broad range of fuels, including light fuel oil (e.g., diesel) used in locomotives in backstop jurisdictions. The federal fuel charge currently applies in Alberta, Saskatchewan, Manitoba, Ontario, Yukon and Nunavut. | High—incentivizes lower GHG emissions throughout supply chain, including fuel switching to lower carbon fuels. | Link 1<br><br>Link 2 |

| Department | Legislation | Regulation | Year | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|---|
| **Environment and Climate Change Canada** | Clean Fuel Standard (proposed) | | TBD | The objective of the Clean Fuel Standard is to achieve up to 30 million tonnes of annual reductions in greenhouse gas emissions by 2030, making it an important contribution to the achievement of Canada's target of reducing national emissions by 30% below 2005 levels by 2030. The CFS will be a performance-based approach designed to incent the innovation and adoption of clean technologies in the oil and gas sector and the development and use of low-carbon fuels throughout the economy.<br><br>The Clean Fuel Standard regulations will cover all fossil fuels used in Canada, but will set separate requirements for liquid, gaseous and solid fossil fuels. It is being developed in a phased approach, with liquid fuel class regulations being developed first followed by gaseous and solid fuel class regulations.<br><br>Publication of the draft liquid class regulation is expected in Canada Gazette, Part I, in fall 2020, with a final publication in 2021. The first year of required reductions in carbon intensity of fuel would be 2022. | High — Rail operators who import diesel fuel into Canada would be subject to requirements to reduce the carbon intensity of those fuels. The CFS will result in the diesel pool for rail having a lower carbon intensity (e.g. reduced upstream emissions) and increasingly higher blends of renewable content. Rail operators could also electrify their transportation fleet and create CFS credits for compliance or sale into the credit market, or could purchasing CFS credits from others. It may also incent fuel switching for locomotives. | Link 1 |
| **Manitoba Department of Conservation and Climate** | The Climate and Green Plan Act | Clean Fuel Standards | 2020 | Manitoba will increase the ethanol content requirement of gasoline to 10% from 8.5%, and the biodiesel content of diesel to 5% from 2% - pending consultations. These numbers represent the highest renewable fuel content requirements in Canada. | High - will encourage development and uptake of low carbon technologies in transportation sector | Link 1 |

| Department | Legislation | Regulation | Year | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|---|
| **Northwest Territories Department of Finance** | Petroleum Products and Carbon Tax Act | | 2019 | Rates for diesel will be $0.082/L as of July 2020, rising to $0.137/L by 2022. Applicable on fuel sold in NWT. | Low - based on low levels of fuel consumption by the rail sector in this region. | Link 1<br>Link 2 |
| **Nova Scotia Environment** | Environment Act | Cap-and-Trade Program Regulations | 2018 | 2020 price is $20 per emission allowance, but that will increase by 5% plus inflation annually. Revenue from program will be reinvested into climate change initiatives | High - Incentivizes lower GHG emissions throughout supply chain, including fuel switching to lower carbon fuels. Program revenues may be available for transportation RD&D. | Link 1<br>Link 2 |
| **Ontario Ministry of Environment, Conservation and Parks** | Environmental Protection Act | Greener Diesel Regulation (O. Reg. 97/14) | 2015 | Ontario has set minimums for the amount of bio-based diesel in the diesel fuel distributed, used, and/or sold in Ontario. As of 2017, 4% of the total volume of diesel fuel must be bio-based. The bio-based diesel component of this blend must have 70% lower greenhouse gas emissions than standard petroleum diesel. | High - will encourage development and uptake of low carbon technologies in transportation sector | Link 1<br>Link 2 |
| **Québec Ministry of the Environment and the Fight Against Climate Change** | Bill 44 - An Act mainly to ensure effective governance of the fight against climate change and to promote electrification | | 2019 | The fight against climate change includes all measures to reduce, limit or prevent GHG emissions, in particular by electrification, to remove GHGs from the atmosphere, to mitigate the economic and social consequences of such measures and to promote adaptation to the impacts of global warming and climate change, as well as Québec's participation in regional or international partnerships in these areas and the development of such partnerships.<br><br>The Minister shall ensure the coherence and coordination of policies, action plans, programs, consultation processes and other measures of the Government that concern the fight against climate | | |

| Department | Legislation | Regulation | Year | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|---|
| | | | | change and shall be involved in their preparation. Each minister or public body concerned continues to be responsible for choosing and implementing the means to achieve the results.<br><br>The Minister must be consulted when measures that could have a significant impact in the fight against climate change are developed. The Minister shall give the other ministers and the public bodies any opinion they considers appropriate to promote the fight against climate change, in particular when a proposed measure does not, in their opinion, comply with the principles and objectives set out in the climate change framework policy provided for in section 46.3 of the Environment Quality Act (chapter Q-2) or with the GHG reduction or limitation targets set under section 46.4 of that Act, and may recommend to them the adjustments required to ensure such compliance. | Medium – Act includes nothing specific to rail sector yet applies to all modes of transport. The Minister must approve all monetary transfers to the Land Transportation Network Fund, which supports programs aimed at providing low-carbon shared mobility options (including passenger rail) in Québec. | Link 1<br><br>Link 2 |
| **Québec Ministry of the Environment and the Fight Against Climate Change** | Petroleum Products Act | Minimum volume of renewable fuel in gasoline and diesel fuel (proposed) | 2019 | The Regulation sets standards for the integration of renewable fuels into gasoline and diesel fuel. As of July 2021, 2% minimum bio-based diesel fuel (annual average blend) is required, increasing to 4% in 2025. | High - will encourage development and uptake of low carbon technologies and fuels in transportation sector | Link 1 |
| **Transport Canada** | Railway Safety Act | Locomotive Emissions Regulations | 2017 | To reduce CAC emissions, the Locomotive Emissions Regulations require federal railway companies to: meet emissions standards set out for new locomotives, carry out emissions testing, follow labelling and anti-idling requirements, keep records, file reports with Transport Canada. | Medium – the Locomotive Emissions Regulations do not explicitly apply to GHGs, yet actions to limit certain CAC emissions are likely to have positive impacts on GHG emissions | Link 1<br><br>Link 2 |

# Appendix C – Federal, Provincial and Territorial Non-regulatory Programs Related to GHG Reductions from Rail

| Department | Program | Years In Effect | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|
| Alberta Ministry of Transportation | Transportation Business Plan 2019–23 | 2019 - 2023 | Outcomes and objectives that could impact rail sector include:<br><br>Outcome #1: Competitiveness, Market Access and Economic Growth: Alberta has a safe and efficient multi-modal transportation system that supports the economy.<br><br>Key objectives: Implementing transportation initiatives that promote economic growth, and supporting opportunities in multiple sectors. None of the plan's initiatives supporting key objectives focus on rail.<br><br>Outcome #6: Environmental Stewardship: Alberta has a transportation system that is managed in an environmentally responsible and sustainable manner.<br><br>Key objectives: Advancing clean transportation technologies to support environmental stewardship and economic development; investigating opportunities and partnerships to develop public transportation systems through grant funding programs that support more sustainable, energy-efficient forms of transportation. | Low - no explicit focus on rail but outcomes/objectives are somewhat relevant | Link 1 |
| BC Ministry of Energy, Mines and Petroleum Resources (in collaboration with industry and academic stakeholders) | The British Columbia Sustainable Marine, Aviation, Rail and Trucking (BC-SMART) Biofuels Consortium | 2018 - ongoing | Encourages the production and use of low carbon intensity drop-in biofuels for long-distance transport to support CleanBC targets. Consortium states that: "While electrification and other technologies can decarbonise urban transport, they are not viable short-term (2030-horizon) options for long-distance transport sectors. Marine, aviation, rail and long-distance trucking require sector-compatible, low carbon intensity renewable fuels." | Medium – focus is on decarbonizing freight transport, however the exclusive area of focus is drop-in biofuels | Link 1 |

| Department | Program | Years In Effect | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|
| **BC Ministry of Environment & Climate Change Strategy** | CleanBC | 2018 - 2030 | The CleanBC plan outlines the province's overarching climate strategy. It provides a blueprint for reducing GHG emissions and fostering sustainable development and economic growth. Priority areas for the transportation sector are: cleaner fuels, cleaner vehicles and more support for measures that get people out of their cars. | Medium – a blueprint document to outline actions that will lead to achievement of provincial climate targets (no actions specific to rail sector) | Link 1 |
| **BC Ministry of Transportation and Infrastructure** | Trade Corridors Initiative | 2019 - ongoing | As containerized traffic increases, the Initiative is increasing safety and efficiency through partnerships and focused investments. Examples of rail-related programs supported are: enhancing container capacity at shipping ports and adding road underpasses to reduce the number of level road-rail crossings. | High - addresses rail efficiency specifically | Link 1 |
| | B.C. on the Move: A 10-Year Transportation Plan (2015) | 2015 - 2025 | This plan sets out 12 priorities that will be key to BC's transportation investments and strategic policy actions from 2015 to 2025. One of the priorities has relevance for efforts to reduce emissions from freight: Enabling Efficient Ports and Rail: Invest in infrastructure that enhances efficiency of goods movement by rail. The Province will work with private sector partners to explore road/rail grade separations on key trade corridors to support the efficient movement of goods by rail. Multi-modal corridor analysis of infrastructure will be carried out, so growing trade from resource sectors can move efficiently through the Pacific Gateway.<br><br>The plan states that safeguarding the environment and taking measures to reduce environmental impacts is a key priority in all of the projects and programs that are delivered to improve transportation. However, there is no discussion regarding the GHG emissions of increased freight activity by rail or GHG reduction measures specific to the rail sector. | Low – no focus on reducing GHG emissions from rail, however activities within scope may serve to enhance rail fuel efficiency | Link 1 |

| Department | Program | Years In Effect | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|
| **Environment and Climate Change Canada** | Pan-Canadian Framework on Clean Growth and Climate Change (PCF) | 2016 - 2030 | The Framework is Canada's plan to meet its emissions reduction targets, grow the economy, and build resilience to a changing climate. The plan includes a pan-Canadian approach to pricing carbon pollution, and measures to achieve reductions across all sectors of the economy. It aims to drive innovation and growth by increasing technology development and adoption to ensure Canadian businesses are competitive in the global low-carbon economy. It also includes actions to advance climate change adaptation and build resilience to climate impacts across the country. There are four areas for action related to rail emissions identified in the PCF [relevant areas included as separate rows below]. | High – will encourage development and uptake of low carbon technologies in transportation sector | Link 1 |
| | Setting emissions standards and improving efficiency (PCF) | 2016 - 2030 | Includes standards for: increasingly stringent emissions requirements for LDVs and HDVs, fuel-efficient tires, HDV fuel-saving devices (e.g., aero), fuel switching in the rail, aviation, marine and off-road sectors. | High – will encourage development and uptake of low carbon technologies and fuels in transportation sector | Link 1 |
| | Shifting from higher- to lower-emitting modes and investing in infrastructure (PCF) | 2016 - 2030 | Work with other levels of government to enhance investments in public transit upgrades and expansions. Build more efficient trade and transportation corridors including investments in transportation hubs and ports. Consider opportunities with the private sector to support refueling stations for alternative fuels for LDVs and HDVs, including natural gas, electricity and hydrogen. | High – will encourage development and uptake of low carbon technologies, fuels and infrastructure in transportation sector | Link 1 |
| | Using cleaner fuels (PCF) | 2016 - 2030 | Develop a Clean Fuel Standard to reduce emissions from fuels used in transportation, buildings and industry. | High – see "Clean Fuel Standard" in Appendix B | Link 1 |
| | Low Carbon Economy Fund (PCF) | 2016 - 2030 | The $2 billion Fund supports the PCF by leveraging funding in projects that will: generate clean growth, reduce GHG emissions, and help to meet or exceed Canada's Paris Agreement commitments. The Fund has two components: the Low Carbon Economy Leadership Fund ($1.5B in funding for provinces and territories that have adopted the PCF) and the Low Carbon Economy Challenge ($500M in funding for innovative private sector projects that reduce emissions and generate clean growth). | High – will encourage development and uptake of low carbon technologies, fuels and infrastructure in transportation sector | Link 1 |

| Department | Program | Years In Effect | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|
| **Government of Saskatchewan (GOS)** | Implementation of the GOS climate change strategy | 2017 - ongoing | In 2017, the GOS released its overarching Prairie Resilience climate change strategy. This strategy recognizes the importance of the transportation sector to our province and also identifies the need to work towards greater efficiency and lower emissions. Specifically, the strategy commits to continue to support industry in expanding the size and usage of the short haul (short line) rail systems. | Low – The GOS is currently developing a baseline utilization of the short line railway industry in SK. The GOS is also working to identify ways to either directly or indirectly encourage modal shift from truck to rail in situations where the shift will result in reduction in overall GHG emissions. | Link 1 |
| **Natural Resources Canada / US EPA, freight carriers, logistics companies, truck carriers, multimodal carriers** | SmartWay Transport Partnership | Canada / 2012- ongoing | A voluntary, North America-wide program to enhance the environmental performance and reduce GHG emissions and fuel use of freight supply chains.<br>Offshoot programs in Canada include SmartDriver and the Green Freight Assessment Program, which are also led by NRCan.<br>SmartWay has proven to be an effective voluntary program for monitoring performance, sharing best practices, and assessing low-carbon technology and fuels. | High – SmartWay provides a wide variety of resources and opportunities for engagement within all four rail decarbonization areas | Link 1 |
| **Ontario Ministry of Transportation** | Metrolinx: GO Rail Expansion and Network Electrification Project | 2017 - ongoing | Through the GO Rail Expansion program, the Province is transforming the GO Transit rail network into a comprehensive, all-day rapid transit network with two-way, all-day service every 15 minutes over core segments of the GO Transit rail network. The Province will look to the private sector to propose innovative approaches to meet future GO Transit rail service levels, including opportunities for technology that could be used to electrify core segments of the GO Transit rail network. | Medium – focus is on expanding GO passenger rail network and exploring options for electrification, and GHG reductions can be realized both through modal shift from cars to passenger diesel rail and through rail propulsion switching from diesel/combustion to electric | Link 1<br><br>Link 2 |

| Department | Program | Years In Effect | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|
| Québec Department of Energy and Natural Resources | Marine, Air and Rail Transportation Efficiency Assistance Program (PETMAF) | ongoing | Aims to reduce or avoid GHG emissions by improving the energy efficiency of organizations and companies that operate marine, air and rail transportation services, including the use of more efficient transportation equipment and equipment and the use of at energies emitting less GHGs. Businesses must have an establishment in Québec. It offers funding in two areas: infrastructure and equipment, and studies and pilot projects. | High - will encourage development and uptake of low carbon technologies, fuels and infrastructure in transportation sector | Link 1 |
| | Greenhouse Gas Emissions Reduction / Avoidance Program through Intermodal Transportation Development (PREGTI) | ongoing | Reduce or avoid GHG emissions from the transportation of goods and people through the establishment of intermodal projects and the promotion of maritime and rail services. Businesses must have an establishment in Québec. | High - will encourage development and uptake of low carbon technologies, fuels and infrastructure in transportation sector | Link 1 |
| | Technoclimat | ongoing | Financial support to encourage the development, in Québec, of technological innovations in energy efficiency, renewable energies, bioenergy and reduction of GHG emissions (up to 50% of eligible expenses to a maximum of $3 million). It is financed through the Electrification and Climate Change Fund. | Medium - not rail specific, although rail GHG reduction initiatives are within scope | Link 1 |
| Québec Ministry of the Environment and the Fight Against Climate Change | Electrification and Climate Change Fund (previously the Green Fund) | 2006 - ongoing | A roughly $1.3 billion fund that is financed through Québec's participation in the North American carbon market, and the province's 4% gas tax. It will support public and private programs that contribute to the electrification of the transportation and industrial sectors, as well as infrastructure. | High – will encourage development and uptake of low carbon technologies, fuels and infrastructure in transportation sector | Link 1<br>Link 2 |
| Sustainable Development Technology Canada (SDTC) | Sustainable Development Tech Fund | Canada / ongoing | The SD Tech Fund supports projects that are pre-commercial and have the potential to demonstrate significant and quantifiable environmental and economic benefits in one or more of the following areas: climate change, clean air, clean water and clean soil. Covers (on average) 33% of eligible project costs, with an average contribution of $3M.<br>Funding must be applied for by the technology provider (e.g. OEM). SDTC generally focuses on SMEs. The Fund is not specific to rail, but broad enough to encompass it. | High – the Fund supports pre-commercial RD&D that has clear linkages to clean air and climate change | Link 1 |

| Department | Program | Years In Effect | Description | Relevance (High, Medium, Low) | Links |
|---|---|---|---|---|---|
| **Transport Canada** | National Trade Corridors Fund | Canada / 2017 - 2028 | Provides funding support to trade-related infrastructure projects in Canada.<br>The $2-billion National Trade Corridors Fund (NTCF) helps fund infrastructure projects in Canada. Infrastructure projects could include work to airports, ports, rail yards, transportation facilities and access roads. Funding for infrastructure projects that increase the flow of people and goods, or help adapt transportation systems to climate change and new technology. | High – NTCF provides funding for a various types of low-carbon rail infrastructure | Link 1 |

# Appendix D – Federal, Provincial and Territorial RD&D Activities

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **CUTRIC and Transport Canada / TC Innovation Centre** | National Rail Innovation Initiative | Canada / 2018 - 2020 | A technology scan to support the department's efforts to facilitate a more integrated and sustainable transportation system through the deployment of new technologies in close collaboration with private industry, academia, rail operators, and other government agencies.<br>• Scan rail technologies that reduce GHG emissions & improve mobility services<br>• Identify the "Top 10" feasible technology theme areas that will build Canada's future passenger and freight mobility systems | Focus Areas:<br>Alternative Propulsion (catenary-free rail electrification; hydrail switching yards as step towards hydrail freight; hydrail for passenger and light-rail); Energy Efficiency (data collection, information management and AI; simulation tool for propulsion systems; CNG and H2; commuter rail electrification); Operational Optimization (enhancing Union Station's flow an capacity); Alternative Materials (build a rail car demonstrator to assess different materials (includes designing, manufacturing, validation, operability, and joining techniques); 3D printing of rail parts; hybrid lightweight structures; sandwiched sheet polymers)<br><br>Catenary-free electrified rail facilities was most popular idea at 5 cross-Canada workshops, followed by passenger hydrail. Efforts to coordinate and streamline academic research are needed. | Link 1 |
| **Metrolinx** | Hydrail Feasibility Study | Ontario / 2017 - 2018 | The study investigated the technical and economic feasibility of the Hydrail System using hydrogen fuel cells to electrify the GO rail service (by 2025). It's an exploration of how hydrogen fuel cell technology, as an alternative to traditional overhead wires, can support the operation of a large scale commuter rail network. | Findings:<br>1. It should be technically feasible to build and operate the GO Transit network using hydrogen fuel cell powered rail vehicles<br>2. The overall lifetime costs of building and operating the Hydrail System are equivalent to that of a conventional overhead electrification system<br>3. The implementation of the Hydrail System of this scale and complexity is innovative and presents a different set of risks as well as benefits, as compared to conventional electrification.<br><br>Key challenges include designing and building a fleet of H2 trains, and securing enough electricity to produce H2 via electrolysis (would require 1% of Ontario's daily generated supply). | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Metrolinx** | Throttle Control Program | Ontario / 2018 - ongoing | Reduce fuel consumption by managing throttle while running locomotives on the line. | Minimize the need to speed up between stations which reduces fuel consumption. | Link 1 |
| | Excess Idle Reduction | Ontario / 2014 - ongoing | Reduce fuel consumed by reducing the time the locomotive is on idle. | GO introduced the IntelliTrain reporting technology to minimize excess idling. It resulted in a 66% reduction in excess idling. | Link 1 |
| | Tier 4AC locomotives | Ontario / 2018 - ongoing | Reducing fuel consumption by introducing a more fuel efficient technology. | Invest to introduce a more fuel efficient locomotive into the active fleet. | Link 1<br>Link 2 |
| **Natural Resources Canada** | Evaluation of Liquefied Natural Gas Infrastructure Build-up for Supplying Fuel to the Rail Market in Canada | Canada / 2017 - 2018 | This study is intended to look for opportunities where an initial deployment of LNG-fuelled locomotives in Canada could make economic and operational sense.<br>The results from this study will be used to inform research and development activities aimed at encouraging the use of natural gas as a locomotive fuel in Canada. | This study mapped 80% of diesel used by Class 1 freight rail in Canada to mainlines between major hubs. NG pipelines were found to service all major hubs where locomotives are refuelled.<br>The report identified an immediate opportunity to leverage existing LNG capacity in Vancouver and Montreal by converting approximately 120 CP locomotives deployed in coal hauling service from the Elk Valley to Vancouver and 60 CN locomotives deployed in the Montreal - Toronto corridor. A focused expansion program providing LNG at just four new key locations in the rail network would enable the conversion of a further 800 locomotives. All freight moving in the Vancouver-Montreal corridor would therefore be converted to LNG displacing a cumulative total of up to 1 million litres of diesel annually and producing savings of up to $500 million annually. The investment in liquefaction facilities and refuelling infrastructure is calculated to be $680 million. The investment in locomotive conversion kits and LNG tenders, estimated at just over $1 billion, has a healthy return on investment so long as diesel prices remain above 80c/L and locomotive conversion technologies maintain substitution above 75%. | Report shared by Transport Canada |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Natural Resources Canada** | Economic and Environmental Benefits of Natural Gas Fuel for the Rail Sector in Canada | Canada / 2016 - 2017 | The objective is to estimate the economic benefits of using natural gas as a fuel for locomotives in Canada, to evaluate where gaps exist in current research on environmental benefits and make recommendations for addressing the gaps that are found. | If the cost of rail diesel remains over 80c/L, then 40-50% savings in fuel cost can be achieved by converting existing locomotives to run on a blend of natural gas fuel and diesel using OEM supplied conversion kits, leading to a favourable business case. To achieve these savings, large fleets of locomotives need to be converted and refuelled at centralized refuelling depots with new co-located liquefaction plants. Carbon pricing is unlikely to affect the business case decision, even at $50/tonne. A major investment is required in liquefaction and locomotive equipment, estimated at $2.8 billion for the highest potential mainline freight locomotive population of 1,420 locomotives identified in the report. 30% average reduction in NOx and 88% reduction in PM are predicted if all locomotives were converted from their current tier status as of 2014 to the best available OEM-provided natural gas conversion kit. Initial dual fuel technologies being deployed do not reduce GHGs, however advances in engine technology currently under development by both major OEMs do have the potential to reduce GHG emissions. Further R&D is required. | Link 1 |
| **Northwest Territories Department of Infrastructure** | GHG Grant Program for Buildings and Industry | Northwest Territories / 2019 - 2022 | Funding program to support GHG reduction projects and initiatives (up to 25% funding) | The Government of the Northwest Territories' (GNWT) GHG Grant Program for Buildings and Industry is an application based non-repayable grant program designed to support greenhouse gas (GHG) emissions reduction projects and initiatives for NWT businesses, industry and non-profit organizations.Funding and resources for this program have been provided jointly by the GNWT in support of the GNWT's 2030 Energy Strategy, and by Environment and Climate Change Canada (ECCC) under the Low Carbon Economy Leadership Fund (LCELF) in support of the Pan-Canadian Framework on Clean Growth and Climate Change.Examples of potential projects include: fuel switching to lower GHG emissions, and energy retrofits to mobile equipment. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Ontario Ministry of Transportation (MTO)** | High Speed Rail in Ontario: Report by the Special Advisor for High Speed Rail | Ontario / 2015 - 2016 | Identify economic development opportunities, assess international experience in HSR, explore potential financing and delivery models, and provide advice on a preliminary business case for HSR in the Windsor, London, Kitchener-Waterloo, Toronto corridor. | The study assessed two scenarios: A) HSR on a dedicated right-of-way with top speeds up to 300 km/h, and B) HSR operating on a combination of conventional and dedicated tracks with top speeds up to 250 km/h. Scenario A was found to be cost prohibitive due to extensive tunnelling requirements, and scenario B was found only to be cost-effective in the London-Toronto portion of the corridor (due to higher anticipated ridership levels). The London-to-Toronto scenario B option was estimated to cost $7.5B, while the full Windsor-Toronto route would cost $21B. | Link 1 |
| **Sustainable Development Technology Canada (SDTC) / Unit Electrical Engineering Ltd.** | Energy Efficient Transit Propulsion Pilot Program Project | British Columbia / 2012 - ongoing | Pilot electric propulsion systems for mass transit rail vehicles | Unit Electrical Engineering and its consortium are building more powerful, lighter and more efficient propulsion systems. One function of the propulsion system in particular, the Linear Induction Motor or LIM, is being redesigned and optimized to reduce the weight and improve efficiency of the current air-cooled unit by 10%. The optimization of the LIM translates into lowered capital costs, reduced lifecycle costs, and contributes to a reduction in electricity costs to the end user. The optimized LIM will also address the differing needs for transit systems in emerging markets by introducing a more effective cooling approach that protects from sand and temperature extremes, making this product ideal for export opportunities, specifically in the Middle East where rail is an emerging sector. | Link 1 |
| **TransPod** | TransPod Hyperloop | 2018 - ongoing | Create a high speed rail corridor between Calgary and Edmonton to reduce rail-based travel time from 3 hours to 30 minutes. | TransPod, a private consortium, has said the $6B project will be funded by the private sector. However it has requested support from the Government of Alberta before securing $100M in private funds to build a 10 km test track. The bullet-type high speed train could reach speeds of up to 1,000 km/h, and would be fully electrified. It would operate within a low-pressure vacuum tube to minimize friction. Pending government approval, construction of the 10 km test track would take place between 2021 and 2023. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Transport Canada / Alberta Innovates - Technology Futures (AITF), AAR, CP, CN, University of Alberta, NSERC** | Canadian Rail Research Laboratory (CaRRL) | Alberta / 2019-2024 | To conduct research related to ground hazards and winter service reliability (including winter-resilient fuels and materials) | Focus is not on fuel efficiency or emissions, but the lab can help to verify that emerging fuels, technologies and components are robust enough to perform well in Canadian winters. Themes of CaRRL's current phase of research are: assessing infrastructure performance, ballast quality and degradation, cold weather performance of rail and rolling stock, quantitative risk management of railways and optimizing rail operations and control systems. | Link 1 |
| **Transport Canada** | Electrification of the Freight Rail Sector in Canada: Review of the Feasibility, Costs and Benefits | Canada / 2017 - 2018 | To foster a better understanding among government and industry rail stakeholders on the costs, challenges, technology and environmental benefits of the electrification of the freight rail sector in Canada as a means to further reduce emissions from their operations. To assess the feasibility of an electrified freight rail network in Canada. | Heavy-haul rail freight electrification technology is mature and rapidly gaining a dominant share of traffic in countries around the world. Up to 2.3 Mt of $CO_2e$ emissions could be eliminated (43% of freight rail emissions) and annual cost savings of $520 million generated through electrification of all transcontinental Canadian Class I freight traffic between Vancouver and Montreal. To do this, an investment of over $10 billion would be required over a multi-year period. This would enable the conversion of ~60% of all freight traffic to electric traction and wold electrify 25% of Canada's entire freight network. A more economically attractive scenario of partial electrification from Vancouver as far as Regina or Winnipeg would cost between $2.3 and $5.2 billion with returns on this investment of between 6% and 25%, depending on the price of diesel and the cost per kilometer of electrification. GHG reductions in this scenario are only 1.3 million tonnes $CO_2e$ because a large portion of the electrification takes place in provinces that currently have a high electricity carbon intensity. | Report shared by Transport Canada |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Transport Canada** | Transport Canada Innovation Centre | Canada / 2018 - ongoing | The Innovation Centre is a transportation innovation research, development and deployment (RD&D) organization that supports emerging transportation technologies to help ensure Canadians can benefit from a safe, secure, clean, and integrated transportation system. | A key focus of the Centre is to help prepare Canada for disruptive transportation technologies. It does this by partnering with industry, academia and government to help advance emerging transportation technologies through RD&D. The Centre's rail-related RD&D initiatives focus on six themes which mirror the priorities of RRAB: grade crossings and trespassing, service efficiency and capacity, energy, environment and climate change, infrastructure and rolling stock, human factors, and ground hazards research and cold weather operations. | Link 1 |
| | Rail Research Advisory Board of Canada (RRAB) | Canada / 1989 - ongoing | 1. Optimize collaboration and create synergy in the railway R&D programs of the three performing sectors of industry, government and academia;<br>2. Help mobilize resources and programs to address problem areas and issues of particular relevance in Canada; and<br>3. Facilitate participation by industry and academia in the formulation and implementation of railway-oriented R&D programs by the federal government. | RRAB is Transport Canada's primary mechanism for enhancing rail-related research collaboration between research bodies, industry and federal departments. A major focus is on advancing the safety of Canada's rail industry. RRAB has six themes for rail R&D which are the same as those of TC Innovation Centre for rail RD&D. Between 2009-2015, RRAB has undertaken 74 R&D projects, of which 8 were completed under Energy and Environment theme. | Link 1<br><br>Link 2 |
| | Clean Rail Academic Grant Program | Canada / 2011 - 2018 | To provide federal funds to academic researchers developing technologies and practices to reduce rail emissions | The Program awarded $250,000 in grants per year ($25k for 10 academic projects). Projects in scope included those on: locomotive systems, data management, infrastructure, railcars (lightweighting and friction reduction), fuels, optimizing rail operations. | Link 1<br><br>Link 2 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
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| **Transport Canada** | Clean Transportation System - Research and Development Program (CTS-RD) | Canada / 2019 - 2022 | CTS-RD supports needed research in the aviation, marine, and rail sectors. It gathers the technical information needed to develop evidence-based regulations and measures to reduce emissions, and achieve Canada's goal of reducing GHG emissions. | CTS-RD has a budget of $1.5M over three years. Maximum grant per project is $100,000. Eligible projects in rail space must focus on reducing or measuring air pollutants, and be focused on one of the following topics: locomotive systems, network management, infrastructure, rail cars, fuels, optimizing rail operations. | Link 1 |
| **Transport Canada / CRB Innovations, ECCC, NRCan** | Higher-Concentration-Blend Lignin-Derived Diesel Fuels for Rail Applications | Canada / 2014 - ongoing | This project, in partnership with NRCan's CanmetENERGY Ottawa, aims at developing the highest-concentration-blend possible of lignin-derived diesel fuel. It is also meant to assess the feasibility of using lignin-derived diesel fuels as drop-in biofuels in order to reduce emissions from the rail sector. | Current renewable diesels are produced from edible sources, including canola oil, soya oil and other lipids, which can have a negative impact on food prices. However, lignin is a highly abundant source that has no effect on food prices, and is readily available to be mixed with fuels to reduce transportation emissions of non-biogenic greenhouse gases and criteria air contaminants. The lignin-derived feedstock can be processed to create a diesel fuel fraction which meets all but one CGSB 3.18 locomotive fuel specification, but will have research and testing conducted to improve the quality of the fuel in order to meet that specification.<br><br>CRB Innovations built a satellite plant that produced 5.5M L of fuel per year. Cost of fuel (with no profits) was $0.61/L. Payback period for plant was 11 years. Cost of raw biomass had large impact on costs and payback period.<br><br>Emissions tests were conducted with drop-in fuel by ECCC in a Tier 4 engine. CO and HC measurements exceeded limits, but ECCC stated that with proper aftertreatment systems installed (DPF or DOC) emissions would easily meet standards. Further tests are needed, but it's expected that drop-in fuel will show reductions vs diesel. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
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| **Transport Canada and Environment and Climate Change Canada (ECCC)** | Hydrail Switcher Locomotive Project | Canada / ongoing | | TC and ECCC are working together to explore the design and deployment requirements for a hydrogen fuel cell switcher locomotive operating within a trainyard with supporting fuel infrastructure. The Phase I feasibility study examined the facets of deploying a hydrogen switcher locomotive in a long-term demonstration project.<br><br>• Assessment of hydrogen fuel cell technology for rail primer movers<br>• Identification of a representative and typically used switcher locomotive<br>• High level design and deployment characteristics as well as estimated cost of conversion<br>• Overview of refueling infrastructure options<br>• Estimated air quality, GHG and economic impacts<br>• Overview of codes, standards and regulations<br>• Impact on railway operations<br>• Recommendations to advance project<br><br>The study concluded that hydrogen fuel cell technology is ready for deployment in a demonstration program. The recommendations direct parties to consider a demonstration project that gradually integrates hydrail use into railway operations over time. | |
| **Transport Canada and Natural Resources Canada** | Two-in-One Catalytic Converter for Simultaneous Removal of NOx and PM from Locomotives | Canada / ongoing | | TC's Innovation Centre has partnered with NRCan CanmetENERGY Ottawa to design a two-in-one catalytic converter for locomotives that would control emissions of NOx and PM without a penalty to engine power. The improved catalytic converter would improve emissions reduction performance. | |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
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| **Transport Canada and Environment and Climate Change Canada (ECCC)** | Locomotive Emission Testing During Revenue Operation | Canada / ongoing | | TC and ECCC have partnered to evaluate locomotive exhaust emissions during revenue operations in summer and winter conditions. This project will expand knowledge of locomotive emissions which are currently tested in static conditions only. Revenue-service operation emission data will provide a baseline against which future experiments using emission-reducing technologies can be compared. The data collected in this project could be used to evaluate the effectiveness of GHG reducing technologies. | |

# Appendix E – Canadian Rail Industry Activities

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| Canadian National Railway (CN) | Fleet Renewal | ongoing | Overarching objective: this initiative will contribute to is to reduce GHG emissions intensity of fleet by 29% by 2030 (from 2015 levels) | CN is investing in and upgrading its fleet, making significant investments in Tier 4 locomotives, new-generation railcars, hybrid and electric vehicles.<br><br>The new Tier 4 locomotives are designed to meet regulatory standards producing less criteria air contaminants.<br><br>Cleaner, more fuel-efficient rail and non-rail equipment will be important to continue to decouple growth from GHG emissions. | Link 1 |
| | Innovative Technology | ongoing | Overarching objective this initiative will contribute to is to reduce GHG emissions intensity of fleet by 29% by 2030 (from 2015 levels) | CN explore and invest in innovative technologies, from telemetry systems, to distributed power, to energy managements systems.<br><br>Trip Optimizer – Regulates the speed of a train by controlling the locomotive throttle and dynamic brake, and computes the most efficient manner to handle the train.<br><br>Distributed Power (DP) – Allows for remote control of the locomotives and improves braking performance, train handling and fuel efficiency. It improves safety from reduced sticking brakes and damaged wheels. | Link 1 |
| | Use of Big Data | ongoing | Overarching objective this initiative will contribute to is to reduce GHG emissions intensity of fleet by 29% by 2030 (from 2015 levels) | Optimizing horsepower to gain fuel efficiency, through locomotive telemetry systems.<br><br>CN is collecting data to improve performance and fuel conservation, using the Horsepower Tonnage Analyzer (HPTA). The HPTA tool was built in house and gives crews instructions and real-time monitoring to ensure they only use the power needed during a trip, by optimizing a locomotive's horsepower-to-tonnage ratio. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| Canadian National Railway (CN) | Enhancing Operating Practices | ongoing | Overarching objective this initiative will contribute to is to reduce GHG emissions intensity of fleet by 29% by 2030 (from 2015 levels) | CN operating model, Precision Scheduled Railroading, allow us to use fewer railcars and locomotives to ship more freight in a tight, reliable and efficient operation.<br><br>Routing protocols and collaborations with ports and terminal operators are helping to minimize dwell times and further drive fuel efficiency.<br><br>Through on-the-job training, CN is working closely with train crews and rail traffic controllers on best practices for fuel conservation – from locomotive shutdowns in yards to streamlined railcar handling, train pacing, coasting and braking strategies.<br><br>CN's locomotive engineers receive real-time information on train characteristics, performance and terrain, through an Energy Management System (EMS), which helps to compute the most efficient train settings and regulate speed. Providing information to track performance in real time to enable fuel conservation through notch limiting, idling reduction and horsepower optimization. | Link 1 |
| | Expanding use of Cleaner Fuel | ongoing | Overarching objective this initiative will contribute to is to reduce GHG emissions intensity of fleet by 29% by 2030 (from 2015 levels) | Driven by regulatory requirements, the growth of the renewable fuel market presents an opportunity for CN to further reduce emissions by using biodiesel blends in its locomotive fleet.<br><br>In the coming years, CN look forward to working with suppliers to explore the greater use of renewable fuels. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Canadian National Railway (CN)** | LNG Demonstration | 2013 | Explore feasibility of LNG locomotives in terms of environmental performance and costs | In 2013, CN retrofitted two 3,000-horsepower locomotives with engines that run on a fuel mix of 90% liquefied natural gas and 10% diesel. They operated in AB. The project was abandoned for a few key reasons:<br>• the fuel tank technology was not there yet for CN use (there were also regulatory hurdles as the high fuel volume required a rail tender)<br>• the reductions in GHG emissions were fairly low considering high CAPEX costs (20 – 30% GHG reductions by using LNG instead of diesel)<br>• the business case was not good (the initial driver was the cost of natural gas vs diesel but markets fluctuated) | Link 1<br>Link 2<br>Link 3 |
| | Providing low carbon transportation and logistics solutions to customers | ongoing | Provide customers with suite of intermodal solutions to optimize environmental, social and economic benefits of different modes | In addition to providing a fuel-efficient transportation service, CN believes that rail can be an integral part of the climate change solution offering both environmental and economic advantages. Compared to other transportation modes, rail is the most fuel efficient method of moving freight over land – on average, trains are approximately four times more fuel efficient than trucks. To leverage these benefits, CN offers its customers intermodal freight shipping, which combines the resources of different transportation modes, such as trucking and rail – allowing each mode to be used for the portion of the trip to which it is best suited. As a result, intermodal shipping helps lower transportation costs, reduce emissions, traffic congestion, accidents, and the burden of an overstressed public road transportation infrastructure. | |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Canadian National Railway (CN)** | Investments in rail infrastructure, network fluidity and efficiency | ongoing | Deploy targeted investments to various elements of Canada's rail network to enhance efficiency, sustainability and capacity | In 2018, CN implemented its largest-ever capital expansion program to add new infrastructure needed to boost the railway's capacity, fluidity and resiliency. This included investment in double track and yard capacity to allow CN to better manage through and recover from unplanned network disruptions, as well as the addition of new, more efficient locomotives. Specifically, in 2018, CN invested a record C$3.5 billion in its capital program, with C$1.6 billion invested to maintain the safety, integrity and fluidity of the network, C$1.0 billion on strategic initiatives to increase capacity, enable growth and improve network resiliency, including line capacity upgrades and information technology initiatives, and C$500 million on equipment capital expenditures including 65 new high-horsepower locomotives. | |
| **Canadian Pacific Railway (CP)** | Trip Optimizer – Technology added to line haul locomotive fleet | 2009 - ongoing | To date CP has installed TO technology on its active line haul locomotives with a goal to apply this technology to 85% of this part of our road fleet (approximately 600 locomotives) by 2025. | Since 2009, CP has actively installed Trip Optimizer (TO) technology on high-horsepower road haul locomotives. Effectively a sophisticated locomotive cruise control optimized for fuel economy, FTO equipped locomotives enable trip planning to significantly reduce fuel and energy consumption. TO takes into account factors such as train length, weight, and track grade to determine the optimal speed profile for a given portion of track. TO systems have been demonstrated to effectively reduce locomotive fuel consumption and corresponding GHG emissions by an average 5%.<br><br>As of 2019 through the use of TO technology CP estimates an annual GHG emissions savings of 72,333 Tonnes CO2e. | |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Canadian Pacific Railway (CP)** | Locomotive Modernization Program – upgrades to locomotive fleet | 2017 - 2024 | Starting in 2017, and continuing through 2024, CP plans to significantly upgrade and retrofit up to 321 six-axle, high-horsepower locomotives.<br><br>A conservative estimate of emissions reductions associated with this project have been calculated based on a fuel efficiency guarantee of 2.7% as provided by our equipment vendor. It is anticipated that the combined effect of locomotive upgrades coupled with installed fuel saving technology will result in a realized fuel savings beyond 2.7%. | The Locomotive Modernization program is a multi-year fleet renewal program at CP. Starting in 2017 through 2019 CP upgraded and retrofitted 171 locomotives to meet our operational needs. Locomotive modernization includes technology upgrades, advanced EPA Tier 1+ diesel engines, enhanced cooling and improved traction systems. All units will be equipped with GE Trip Optimizer and Distributed Power which are both EPA certified fuel/emissions reduction technologies.<br><br>By the end of 2019, these improvements will be reflected in 25% of our active line haul fleet, having a direct and positive impact on CP's fuel efficiency and corresponding GHG and air pollutant emissions<br><br>Modernized locomotives currently in service through 2019 were estimated to have reduced GHG emissions by 11,475 Tonnes CO2e. | Link 1<br><br>Link 2<br><br>Link 3 |
| | Precision Scheduled Railroading – efficiency focused railway operating model | 2012 - ongoing | There are no specific GHG goals associated with PSR. | Since 2012, CP has successfully operated its precision scheduled railroading (PSR) approach. One of the first Class 1 freight railways to successfully implement this approach, PSR involves constant monitoring and optimization of all railway assets and processes to maximize operational efficiency, improve outcomes for CP's stakeholders and increase safety for employees and communities. To support PSR objectives, CP has made significant upgrades to its rail network, locomotive and car fleet, improved operational practices and invested in technological improvements. Notable factors contributing to improved fuel economy include enhanced productivity from running longer trains and proficient operating plan efficiency.<br><br>Implementation of process improvements and changes to operating practices associated with CP's precision scheduled railroading operations model has driven a 3% improvement in fuel efficiency between 2017 and 2018 – saving an estimated 74,663 Tonnes CO2e annually. | Link 1<br><br>Link 2 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
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| **Canadian Pacific Railway (CP)** | CP Fast Pass Technology – deployment of technology to improve operations and support CP value chain and low carbon transportation solutions. | 2018 - ongoing | To better align with customer objectives for low-carbon transportation solutions, CP continues to optimize intermodal operations to provide opportunities for the proficient transfer of goods and materials from truck to train transport. | A significant portion of CP's business involves the long haul transport of consumer goods and materials that have typically shipped using heavy highway truck transport. By shifting an increasing portion of these materials from highway transport to freight rail services, CP is able to have a material impact on GHG emissions for customers and the North American transportation sector.<br><br>In 2018, CP implemented Fast Pass technology at all 10 intermodal locations, expediting truck traffic through facilities improving efficiency and dramatically reducing truck wait times and associated idling.<br><br>Collectively CP's improvements in intermodal operations have resulted in heightened demand for freight traffic through these facilities. In 2018 intermodal traffic increased by nearly 10% with an additional 2.4 billion revenue-ton miles transported for customers. CP estimates this project to have resulted in approximately 139,945 Tonnes CO2e less GHG emissions our customers avoided shipping these materials by train versus highway truck transport. | Link 1<br><br>Link 2<br><br>Link 3 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Canadian Pacific Railway (CP)** | 8,500-foot High Efficiency Product Trains (HEP) – improving supply chain capacity and efficiency | | Once fully implemented, HEP Trains will allow for more grain to be hauled within a shorter train, thus increasing shipping capacity and improving fuel consumption by reducing the number of trains required to haul the same volume of grain. Significant GHG savings are expected due to improved fuel efficiency. | CP is driving towards a supply chain model capable of loading, transporting and unloading 8,500-foot long, power-on, unit trains with a minimum of 134 hopper cars of export grain in Canada.To support the HEP train model, CP is purchasing 5,900 new grain hopper cars as a part of a $500 million dollar multiyear investment to upgrade its grain car fleet. New grain cars will replace an aging fleet, allowing CP to provide better and more efficient service to farmers and the North American economy.<br><br>By 2022, CP has committed to purchase a total of 5,900 new hopper cars enabling a complete removal of all low-capacity hoppers from the fleet. The 8,500-foot train model will carry a minimum of 134 grain hopper cars based on industry-average car lengths, which is 20% more grain than traditional 112 car grain trains. As CP and the industry move towards shorter, higher-capacity cars, CP will be able to fit more cars and more grain on each 8,500-foot train. The end result is more grain transported to market more efficiently than ever before.<br><br>CP is also investing in longer sidings and upgrades to its track network enable these longer trains to move seamlessly between elevators and ports. Elevators capable of receiving an intact 8,500-foot train to have the empty hoppers placed clear of our main track operations, loaded and made ready for pick up once again all clear for CP's main track, providing the best use of capacity to move more grain. CP's investment in longer sidings and upgrades to its track network enable these longer trains to move seamlessly between elevators and port. Through infrastructure investment and collaboration with grain companies and port operators, this enhanced train model allows railways, elevators, and ports to increase throughput and better utilize resources. Grain elevator and port terminal infrastructure is being built and expanded to load and unload 8,500-foot trains clear of the mainline track. | Link 1<br><br>Link 2 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
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| **Southern Railway of British Columbia (SRY)** | Alternative fuel research | ongoing | Initiative is designed to gain insight into the feasibility of alternative fuel locomotives in the context of switching activities. | SRY is supporting research being conducted at the University of British Columbia Okanagan into alternative fuels for switching locomotives: 1) hydrogen-fueled and 2) battery hybrid. | |
| **VIA Rail** | Fleet Renewal | 2022 - ongoing | Increase the adoption of fuel efficient, low-carbon locomotive engines and fleet vehicles | Starting in 2022, the delivery of VIA's brand new fleet of trains will include more fuel efficient Tier 4 Diesel engines with the option to operate on electrified rail infrastructure as it becomes available.<br><br>VIA established a green strategy for the replacement of its fleet of vehicles: 100% of administrative vehicles purchased in 2018 were zero emission vehicles. | Link 1<br><br>(see pages 9, 42, 43) |
| | Intermodal Partnership | ongoing | Reduce GHG from use of cars through addressing the first and last mile | Door-to-door type travel through partnering with a diverse range of intermodal carriers including commuter trains, motor coaches, car-sharing companies, airport shuttle bus services, and airlines. | Link 1<br><br>(see page 14) |
| | Fleet Refurbishment (renovation of older fleet) | 2009 - 2013 | Achieve GHG reductions of 15-20% through engine upgrades | Complete renovation of older F-40 locomotives (from 2009-2013) with separate HEP engines. | Link 1 (see page 39) |
| | Renovation of Cars | 2018 - 2020 | Achieve GHG reductions of 3-5% through electrical system upgrades | Multiple initiatives to replace older electrical systems with more energy efficient systems. | Link 1 (see page 39) |
| | Locomotive Engineer Scorecard | 2019 - ongoing | Achieve GHG reductions of 3-5% through operating practices that reduce fuel consumption | Mapping optimal operating parameters for the train on specific segments (Safety & Environment (fuel consumption)) and coach others.<br>Performance Measurement: VIA's innovative telemetry Wi-Tronix system is enabling it to compile data and monitor progress through the LE Energy Management Scorecards. Through on-the-job instruction, VIA coaches its LEs on ways to reduce train idling and improve fuel efficiency.<br>Simulator Training: In support of VIA's fuel efficiency improvement objectives, last year it enhanced its apprentice LE simulator training program. By adding a new feature to the simulator, VIA can now train its apprentice LEs on how to better operate locomotives for lower fuel consumption. | Link 1<br><br>(see page 40) |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| VIA Rail | Optimization of Cycling | ongoing | Achieve GHG reductions of 3-5% through increasing capacity and efficiency of trains on high demand routes | Deployment of most fuel efficient engines on most demanding routes.<br>Train Cycling: VIA significantly increased capacity on its trains where it experienced the highest demand on its network. By better cycling our trains, we increased the number of available seat miles by 5.3% when compared to 2016. | |
| | Idling Policy | ongoing | Achieve GHG reductions of 3-5% through the minimization of idling in railyards | Minimize idling in the yards. Use of telemetry Wi-tronix system to reduce train idling. The need for idling will be minimized in part through the use of Hotstart engine heating systems, and reduced layover times. | Link 1<br><br>(see page 41) |
| | Promote use of trains vs cars or other modes | ongoing | Promote the use of passenger travel by rail as a low-carbon option to cars | To promote the environmental benefits of passenger rail, VIA is taking important steps to educate and create awareness amongst Canadians. Provide an accessible and affordable alternative to cars and support the necessary shift we need to make to reduce the transportation sector's contribution to climate change and smog. | |
| | Use of Biodiesel (B5) | ongoing | Reduce net GHG emissions through the use of low-carbon biodiesel | Biodiesel blends up to 5% are used on a voluntary basis (not a requirement). Suppliers are not able to confirm precise amounts of biodiesel in batches purchased. | |
| | Test of B20 on one engine | | Explore the impacts of higher biodiesel blends on existing engines | Test report completed. Technical issues are frequent on current engine with use of B5+, including less power generation. Requires higher consumption to develop the same power as regular diesel. | |
| | Adjustments to software | ongoing | Reduce GHG emissions by using software to limit available power at low speeds | To limit the available power at very low speeds as this can't be converted to traction power (thereby reducing fuel consumption). | |
| | Future initiatives: Dedicated tracks Electrification of Rail | | Expand VIA's dedicated track network and explore options for electrification | More dedicated tracks will reduce idling due to delays caused by host railways.<br><br>VIA network electrification has the potential to reduce GHG emissions by 13.9 million t CO2e. | |
| | Fuel Efficiency | ongoing | Enhance fuel efficiency to reduce GHG and CAC emissions | VIA has improved its fuel conservation and train idling practices, achieving a 21% reduction in GHG emissions since 2009. Related measures have also led to a reduction in CAC emissions by 20% based on 2009 levels. | |

# Appendix F – International Activities

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
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| **Association of American Railroads (AAR)** | Transportation Technology Center (TTCI) | USA | Primary focus is rail safety. | Past TTCI research has indirectly contributed to GHG reductions through research into double-stack container cars, higher capacity freight cars, AC electric locomotives, wheel-rail lubrication, etc.<br><br>No ongoing projects explicitly related to GHG emissions reductions. It hosts an Annual Research Review, and partners with external groups on strategic research initiatives. | Link 1 |
| **BNSF Railway Company / Wabtec** | Battery electric high-horsepower freight locomotive prototype | USA / 2019 – ongoing | Reduce GHG and air pollutant emissions from yard and road operations. | EV locomotive prototype designed to work when coupled to diesel locomotive(s), which would result in a hybrid train. When completed, the locomotive will be tested on 350 mile stretch between Stockton and Barstow in California. The hybrid consist will use the electric locomotive in rail yards and will use it to supplement power while in transit. Uses Li-ion batteries on a large scale (~20,000 cells). Future versions might use up to 50,000 cells. Before testing begins, a charger will be installed at BNSF's Mormon Yard in Stockton. Locomotive will also charge via regenerative braking. Will use energy management software to analyze route ahead and optimize use of electric power to minimize fuel use. Testing should begin in late 2020.<br><br>Pilot funded in part by $23M (USD) grant from CARB (through cap-and-trade revenues). | Link 1<br><br>Link 2<br><br>Link 3 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
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| **Federal Railroad Administration (FRA)** | | USA | FRA's focus is on improving transportation safety and fostering the development of high-speed rail (HSR) and other intercity passenger service (no explicit focus or programming on GHGs) | FRA's National Rail Plan Progress Report (2010) states that high-performance freight rail and high-speed rail can contribute to DOT's goal of reducing emissions from freight transportation by improving the fuel efficiency of freight vehicles, and by reducing transportation's petroleum consumption.<br><br>In 2009, FRA funded a study that provided a comparative evaluation of rail and truck fuel efficiency on corridors and services in which both modes compete. The study examined 23 freight movements and took multiple distances and commodities into consideration. This evaluation concluded that rail was more fuel efficient than truck on all 23 movements and that fuel savings from using rail can be significant.<br><br>Under The National Environmental Policy Act (NEPA), FRA must examine potential impacts to the social and natural environment when considering approval of proposed transportation projects. | Link 1<br><br>Link 2<br><br>Link 3<br><br>Link 4<br><br>Link 5<br><br>Link 6 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
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| Norfolk Southern Corporation (NS) / Federal Railroad Administration, Penn State University | NS 999 battery electric switcher locomotive | USA / 2008 - 2017 | Assess feasibility of battery electric switcher locomotives for energy savings and reduced emissions | The project to develop the NS 999 battery electric switcher locomotive started in 2008. The test locomotive was produced by converting an older locomotive. A problem with battery power management with the original lead-acid batteries led to the project being abandoned in 2011. A second round of testing using more capable lithium-ion batteries in both simulated and operating conditions was more successful. This testing showed that the lithium-ion pack provided significant energy storage and would be suitable for more rigorous switching duties, local service, and road use.<br><br>Penn State University created a hybrid locomotive model that shared the load between a diesel engine and the lithium-ion pack. The model allowed for dynamic braking energy to recharge the battery pack. NS then used this model to conduct an experiment that showed a hybrid locomotive could reduce emissions of pollutants. However the experiment showed the resulting fuel savings did not offset the capital cost of the battery pack, so each hybrid locomotive would need a large annual subsidy. Battery charging and charging infrastructure was another challenge identified by the experiment. | Link 1 |
| Port of Los Angeles | Zero-Emission Track-Miles Locomotive Project | USA / 2018 - 2020 | Demonstrate a battery electric switcher locomotive in commercial service | The Port of Los Angeles has partnered with VeRail Technologies, Inc. to build and demonstrate a zero-emission switcher locomotive in the San Pedro Bay Ports. This project began as a concept for a near zero-emission natural gas system, and has now evolved into a fully zero-emission, battery electric platform. The 2,100 horsepower six-axle switcher locomotive will operate throughout the network of in-harbor rail lines that service the Ports of Los Angeles and Long Beach, and is anticipated to be capable of working a full 12-hour shift before needing to charge. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
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| **Argonne National Laboratory and Federal Railroad Administration (FRA)** | Railroad Energy Intensity and Criteria Air Pollutant Emissions | USA / 2014 - 2018 | 1. FRA tasked Argonne with updating GREET model to include rail module<br><br>2. Updated model to include life-cycle costs and impacts of CNG, LNG, and dimethyl ether (DME) usage in the rail sector | The project report assumes that natural gas locomotives meet Tier 4 standards for all pollutant emissions. This assumption was made due to inability of Argonne to gather data from rail operators and manufacturers. It found that if the cost differential between diesel and natural gas increases, there "may still be a pathway for natural gas".<br><br>On DME, the report states it's a good alternative to diesel due to high cetane number (helps auto-ignition at low temperatures) and high oxygen content (leads to less soot/PM formation than diesel). DME can be derived from natural gas or methanol cost-effectively, though it can also be produced from a wide range of biomass and organic waste. It is not a GHG and is believed to have lower NOx emissions than diesel. It only has 67% the energy density of diesel. GREET model data for DME were based on on-road trucking test results, as no DME locomotive data was available. DME is believed to result in significant reductions of NOx (27%), PM (94%), HC (74%) and CO (95%) relative to diesel. A challenge is that DME has only half the energy density of diesel, requiring its fuel tanks to be twice as large. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
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| **California Air Resources Board (CARB)** | Technology Assessment: Freight Locomotives | USA / 2016 (with demon-stration and commer-cialization timelines out to 2029) | • To help inform and support CARB planning, regulatory and voluntary incentive efforts in support of CARB's objective to transition on-road and off-road mobile sources to zero tailpipe emissions everywhere possible, and near-zero emissions with clean, low-carbon renewable fuels everywhere else, to meet air quality and climate goals<br>• Scope: considers potential advanced locomotive technologies that could operate on the existing rail network with emissions well below the current national Tier 4 emission levels | The CARB report recommends dual paths for locomotive technology development:<br><br>1) Seek criteria and toxic pollutant reductions beyond Tier 4 in the near to mid-term with aftertreatment augmented by on-board batteries.<br>2) Develop the zero-emission track-mile or zero-emission locomotive technologies needed in the mid to long-term (2025-2050).<br><br>It identifies potential actions such as:<br><br>a) Ask U.S. EPA to define the next generation of national emissions targets for locomotive engine manufacturers ("Tier 5" standards)<br>b) Support development and demonstrations of on-board battery technologies that could also provide zero-emission track-mile capabilities in and around railyards<br>c) For the longer term, advocate for the development of the solid oxide fuel cell gas turbine (SOFC-GT) concept, a potential technology to power an interstate line haul locomotive across the North American freight rail system.<br>d) Support RD&D program to commercialize zero-emission track-mile and zero-emission locomotive technologies.<br>e) Highest priorities for R&D: improvements in battery and fuel cell energy density and costs to support introduction of:<br>– Battery and/or fuel cell powered switch locomotives.<br>– Battery and/or fuel cell tenders for regional operation of medium horsepower and freight interstate line haul locomotives.<br>– Fuel cell (SOFC/GT) freight interstate line haul locomotives. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Alstom / LNVG, Linde Group, Hydrogenics** | Coradia iLint Hydrogen Powered Train | Germany / 2016 – ongoing | Replace diesel passenger trains with non-emitting alternatives on non-electrified stretches of rail | Alstom's hydrogen powered Coradia iLint passenger train entered into active service (two units) in 2018. Design elements include clean energy conversion, flexible energy storage in batteries, and smart management of traction power and available energy. Designed for operation on non-electrified lines. Alstom and LNVG signed contract for delivery of 14 H2 trains in 2017. This contract includes 30 years of maintenance and fuel supply. iLint range on full tank is 1,000 km. Top speed is 140 km/h. Trains were manufactured in Lower Saxony, the same region they will be used in. The Linde Group built H2 fueling station, with plans to eventually use wind-powered electrolysis to produce the fuel. Trains are GHG emissions free and quiet (60% noise reduction), and are direct substitutes for existing diesel trains.<br><br>In Oct 2019, Dutch rail infrastructure provider ProRail, local rail operator Arriva and energy company Engie announced pilot to test the iLint in Province of Groningen, Netherlands. Two week trial scheduled to begin in Q1 2020. | Link 1<br><br>Link 2<br><br>Link 3<br><br>Link 4 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Germany Federal Ministry of Transport and Digital Infrastructure (BMVI)** | Rail Freight Masterplan | Germany / 2017 | The masterplan commits to significantly increase rail's market share of all freight traffic in Germany by 2030.<br><br>A related goal (not included in Masterplan) is to electrify 70% of Germany's freight rail network by 2025. | The Masterplan is a comprehensive package of measures to permanently boost the rail freight sector and offer shippers higher-quality rail freight services at more competitive prices. The focus is on: ensuring high-capacity infrastructure; making extensive use of the potential for innovation; and improving the transport policy framework. It identifies rail freight as a key element of a sustainable mobility and transport strategy, and contains specific milestones, actions, and timeframes.<br>Relevant milestones and actions:<br>• Increase the level of automation in railway operations: technical systems in mainline operations, such as driver assistance systems, will support the train driver in adopting an energy-efficient driving style, thereby reducing $CO_2$ emissions and energy consumption.<br>• Expedite technological innovations for rolling stock: deploy more hybrid locomotives (with electric propulsion for the "last mile" without overhead lines); streamline sets of regulations governing rolling stock with regard to the hampering of innovations by national and/or European provisions; create funding options for the deployment of low-emission and hybrid locomotives and innovative freight cars<br>• Boost multimodality (e.g., granting regulatory privileges to electrically powered and low-emission road vehicles operating initial and terminal hauls in intermodal transport)<br>• Expand electric haulage on and with the railways (e.g., launch a special programme for further electrification of the rail network; develop standardized solutions for the electrification of lines and service facilities with simple requirements; develop and fund electrically powered solutions for initial and terminal hauls to and from the railways; implement a double track approach)<br>• Limit the burden imposed by levies and taxes | Link 1<br><br>Link 2 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **UK Department for Transport** | Decarbonising Transport: Setting the Challenge | UK / 2020 | Identify the current challenges and steps to be taken when developing the transport decarbonization plan (which should result in net zero carbon by 2050).<br><br>Report states that the main way to achieve rail freight decarbonisation is through direct government intervention to roll out further electrification. Network Rail is currently developing the Traction Decarbonisation Network Strategy (TDNS), which will inform decisions about the scale and pace of decarbonization between now and 2050. | Notes on current government aims and targets: UK recognises the environmental benefits of rail freight, and invested £235M in the Strategic Freight Network between 2014 and 2019 to improve the capacity and capability for freight. In 2018, challenged the rail industry to produce a vision for removal of all diesel-only trains from the network by 2040. An industry taskforce assessed the decarbonization options available, and made recommendations for the rail sector.<br><br>Current policies to deliver on the net zero by 2050 targets include: Investment in new technology (e.g., recently funded competitions that provided over £4M for projects to drive decarbonization across passenger and freight); and Providing freight grant schemes to support the carriage of freight by rail and water on routes where road haulage has a financial advantage. These policies helped remove around 900,000 freight truck journeys a year from Britain's roads.<br><br>Planned future work: working with industry on the Traction Decarbonisation Network Strategy, which will consider both passenger travel and rail freight and will inform the deployment of electrification and novel technologies across the railway. Aims to build on Network Rail's internal Decarbonisation Programme to reduce carbon emissions from railway operations, including depots. Plans to publish a new multi-modal freight strategy which will build on the work of the National Infrastructure Commission and will include the Government's final response to the recommendations in the Commission's report Better Delivery: The challenge for freight. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **National Rail / Eurostar, South Eastern Railway, East Midlands Railway, Thameslink, DB Cargo UK** | High Seed 1 (HS1) | UK / 2003 - ongoing | Provide rapid, low-emissions (90% less than flying) travel between London, Kent, and several destinations in mainland Europe. Also allows for rapid, sustainable freight transport. | All-electric trains connect London to the Channel Tunnel (109 km stretch). Trains travel up to 230 km/h in the UK, and up to 300 km/h in mainland Europe. Existing connections to Paris (2 hours), Amsterdam (4 hours), Brussels (2 hours), and Ashford, UK (37 mins)). This route has been in operation since 2007. The route is supplied by electricity using overhead catenary lines with 25 kV AC railway electrification. Some stations on the route include third-rail 750 V DC power to avoid the need to switch power supplies.<br><br>Locomotive rolling stock on HS1 includes: Eurostar e300 (Class 373), Javelin (Class 395), and Eurostar e320 (Class 374) for passenger rail, and Class 92 for passenger and freight. HS1 launched freight services in 2011 to deliver time-sensitive freight from the UK to mainland Europe in a sustainable manner. It includes a connection to the DB Schenker freight depot at Dollands Moor (near the Chunnel). | Link 1<br><br>Link 2 |
| **National Rail / UK Department for Transport, municipal governments, airports** | High Speed 2 | UK / 2029 - 2040 | 1. Create better linkages between cities and regions to enhance economic development in struggling areas.<br>2. Add capacity by taking inter-city trains off of existing network to free up space for commuter and freight rail. This will address over-crowding and reduce the number of on-road freight trucks.<br>3. Reduce transport carbon footprint by providing low-carbon option for long-distance travel. HS2 will be 17 times less carbon intensive than flying and 7 times less than driving.<br>3 objectives summarized as: connectivity, capacity, carbon. | New high speed rail network under development, offering service to 8 of UK's 10 biggest cities. HS2 Phase One (215 km stretch between London and Birmingham) will open between 2029 and 2033. Phase Two will add over 330 km of track and connect to Manchester and Leeds. HS2 will serve more than 25 stations and help to connect over 30 million people. It is projected to create over 500,000 jobs and lead to the construction of 90,000 homes close to HS2 stations.<br><br>Operating speeds will be 330-360 km/h. It will be a double track network using standard gauge rail, powered by 25 kV AC overhead catenary lines.<br><br>Over 60 years, the network is expected to provide over 92B pounds in net benefits and 44B in revenue, for a total benefit-cost ratio of roughly 1.4. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Autonomous Parisian Transportation Administration (RATP)** | Battery-overhead electric locomotives | France / 2017 - 2019 | | In 2017 RATP ordered 12 battery-overhead electric locomotives to be used to tow trains that provide maintenance for France's regional railway network infrastructure. These units will be able to charge their nickel-cadmium batteries via overhead catenary wires when wires are available, and will also be able to operate on non-electrified portions of track. The locomotives, produced by Spanish rail firm CAF, will have a maximum power output of 1MW. | Link 1<br><br>Link 2 |
| **French National Railway Company (SNCF)** | Railway electrification and SNCF LGV (high-speed rail) | France / 1920 - 2020 | Provide travellers with high speed ground-based transport powered by domestically-sourced energy | Railway electrification in France began in the 1920s, with mainlines originating in Paris and the Pyrenees electrified using overhead catenaries at 1,500 V DC. Today France is home to more than 15,000 km of electrified rail lines (55% of total rail network). Following WWII, 25 kV AC overhead lines became the norm. Today, all electrical equipment ordered by France's national railway company SNCF can use both 25 kV AC and 1,500 V DC.<br><br>Exploratory work on high speed rail began in 1966, originally with the intent of being powered by gas turbines. The 1973 oil crisis caused the focus of efforts to shift to electric propulsion. The first line came into service in 1981, connecting Paris and Lyon, powered by 25 kV AC overhead catenary lines. As of 2017, the system has over 2,600 km of high speed rail, with top speeds ranging from 250 to 320 km/h. High speed trains run on conventional tracks and can utilize conventional rail and station infrastructure, unlike high speed lines in Spain and Japan. The high speed lines are typically reserved for use by high speed trains to avoid congestion. All lines are fenced to prevent people and wildlife from crossing tracks, and level crossings are not permitted. | Link 1<br><br>Link 2 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Union Of European Railway Industries (UNIFE)** | RAILENERGY | EU / 2006 - 2010 | Reduce energy consumption in railway systems thus reducing life cycle costs of operation and CO2 emissions. The project target is to achieve a 6% reduction in the specific energy consumption of the rail system by 2020. | The special feature of RailEnergy is the holistic approach to energy efficiency. A global model was developed to simulate energy losses in railway systems. The model was based on a 'plug & play' principle for each component or operational pattern. RailEnergy serves as a platform for an integrated development of new methodologies, techniques and technologies. The final outcome of the RailEnergy project confirms that an average relative energy saving of more than 7% can be reached. Recommendations included predicting the energy performance of new rolling stock using the new technical standard TecRec 100_001, greater use of eco driving (saving potential between 5% and 15%), energy efficient traffic management (saving potential between 10% and 20%), and parked train management (saving potential of 4% to 8%). | Link 1<br><br>Link 2 |
| **International Union of Railways (UIC) and European Commission (EC)** | CAPACITY4RAIL | EU / 2013 - 2017 | Develop guidance, standards and technology demonstrations to help rail stakeholders optimize infrastructure, operations and vehicle performance, leading to a global freight rail sector that is adaptable, resilient and competitive. | CAPACITY4RAIL aims to pave the way for future railway systems, delivering coherent, demonstrated, innovative and sustainable solutions for track design, freight systems, operation and advanced monitoring. With a comprehensive system vision, it will contribute to the development of guidance documents identifying further actions to be undertaken and future technologies and systems to be developed. It will contribute to the development and definition of European Standards (e.g., EC white paper), future regulations, and European research policy and will facilitate a high degree of interoperability. Adaptable systems that offer high operational capacity and high reliability and resilience to hazards are needed. Such changes will only be achieved through global efforts that combine optimization of infrastructure, operations and vehicle performance. | Link 1<br><br>Link 2 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **European Commission (EC)** | Shift2Rail | EU / 2009 - ongoing | 1. Enhance rail service quality by improving reliability and punctuality by up to 50%<br><br>2. Reduce congestion and CO2 emissions by doubling European rail capacity<br><br>3. Reduce costs of infrastructure and rolling stock by 50%<br><br>4. Retain Europe's leadership in global rail market | Shift2Rail seeks focused research and innovation (R&I) and market-driven solutions by accelerating the integration of new and advanced technologies into innovative rail product solutions. Shift2Rail promotes the competitiveness of the European rail industry and meets changing EU transport needs. R&I carried out under this Horizon 2020 initiative develops the necessary technology to complete the Single European Railway Area (SERA). The program's scope applies to both passenger and freight transport.<br><br>In 2019, Shift2Rail released "Study on the use of fuel cells and hydrogen in the railway environment". Study shows potential for use of H2 and fuel cells in multiple units, shunter locomotives and freight locomotives. | Link 1<br><br>Link 2<br><br>Link 3<br><br>Link 4<br><br>Link 5 |
| **The European Rail Research Advisory Council (ERRAC)** | Rail 2050 Vision | EU / 2017 - ongoing | Roadmap sets out the future capabilities needed from railways to meet the future needs of Europe and provides a route to utilizing new technologies to achieve these capabilities. | A high-level freight and passenger rail roadmap for Europe out to 2050. Highlights critical roles that rail plays in Europe's economy and sustainability efforts, as well as challenges (e.g., climate change) and opportunities (e.g., the creation of a Single European Rail Area (SERA), smart transport technologies, low-carbon transport).<br><br>The 2017 Vision document was preceded by a more detailed 2012 document called Rail Route 2050: The sustainable backbone of the Single European Transport Area (Link 2). It also draws from a high-level 2011 EC paper titled: White Paper: Roadmap to a Single European Transport Area - Towards a competitive and resource efficient transport system (white paper is profiled as a separate item).<br><br>The Vision is heavily technology-driven and foresees digitalization, automation and smart infrastructure significantly enhancing the efficiency of rail operations while eliminating carbon emissions. | Link 1<br><br>Link 2 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **European Commission (EC)** | White Paper: Roadmap to a Single European Transport Area - Towards a competitive and resource efficient transport system | EU / 2011 - 2012 | To ensure continued economic and environmental viability of the rail sector in Europe. | Reducing emissions and oil use is a core objective, to be addressed by new technology and infrastructure. Reducing congestion is a theme too, as are safety, security and noise reduction. Emissions reduction goal for entire transport sector in Europe was 60% of 1990 levels by 2050. Rail was seen as a major contributor towards achieving that goal. | Link 1 |
| **The Community of European Railway and Infrastructure Companies (CER) and International Union of Railways (UIC)** | Moving Towards Sustainable Mobility: A Strategy for 2030 and Beyond for the European Railway Sector | EU / 2010 | To provide a medium and long term plan for the European rail sector that aligns with broader environmental and public policy objectives. | A voluntary rail industry strategy to highlight rail's low-carbon performance and ensure it plays a central role in global transportation decarbonization strategies. Targeted $CO_2$ reductions from 1990 levels are: 30% by 2020, 50% by 2030, and 100% by 2050. It calls for $NO_x$ and $PM_{10}$ reductions of 40% by 2030 and 100% by 2050 (from 2005 levels). It also calls for a 50% improvement in energy efficiency by 2050.<br><br>It lays out three broad ways to reduce $CO_2$: improved energy efficiency, the electrification of lines, and the decarbonization of electricity generation.<br><br>Several CER and UIC programs continue to aim for these targets. | Link 1 |
| **Australasian Railway Association (ARA)** | Code of Practice for Management of Locomotive Exhaust Emissions | Australia and New Zealand | The Code of Practice describes recommended practices for the management and improvement of exhaust emissions of diesel freight locomotives in the Australian railway industry | The Code 1) sets a maximum particulate emission standard for new locomotives and 2) requires that locomotives receiving a major overhaul will, where current emissions of particulate matter are greater than 0.3g/kWh, be fitted with an upgrade kit to bring emissions to or below that level.<br><br>ARA publishes data provided to it by freight rail operators to meet the operators' obligations under the Code of Practice. The Code focuses on two key priorities: GHGs and diesel particulates.<br><br>Companies are required to report annually on their implementation of the Code. | Link 1<br><br>Link 2 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Japan Ministry of Infrastructure Transport and Tourism (MLIT)** | Eco Rail Line Project | Japan / 2016 - ongoing | Promote reductions in energy usage by rail lines and the adoption of carbon emission reduction technologies | The MLIT implements the Eco Rail Line Project, supporting rail operators who are implementing systematic efforts to reduce power consumption and carbon emissions for entire rail lines by using renewable energy in train stations and train control centers and installing energy saving facilities for efficient energy usage. | Link 1 |
| **Japan Ministry of Infrastructure Transport and Tourism (MLIT) / Railway Freight Association (RFA)** | Eco Rail Mark | Japan / 2005 - ongoing | The "Eco Rail Mark" program was established in 2005 to certify companies and products which are proactively working to achieve low environmental impact rail freight transport. It is intended to support and promote a modal shift to freight rail transport. | Supporting companies and products certified with the "Eco Rail Mark" raises awareness of environmental impact reduction efforts among freight companies, manufacturers, and consumers. The MLIT and Railway Freight Association (RFA) are striving to further promote and expand the "Eco Rail Mark". As of September 2017 there are 188 "Eco Rail Mark" product certifications for 213 products, 85 certified companies, and 31 certified supporting companies. | Link 1 |
| **The Association of Japanese Private Railways** | Commitment to a Low Carbon Society | Japan / 2020 and 2030 targets | Commitment targets:<br>• Reduce power usage by 5.7% by 2020 in comparison to 2010;<br>• Reduce power usage by over 5.7% by 2030 in comparison to 2010 | The Commitment includes the following primary measures as means to reduce energy usage from the rail sector:<br>• Introduction of energy saving cars: Energy saving trains such as VVVF controlled trains, which consume less power, and regenerative brake-equipped trains, which efficiently use power, are being introduced.<br>• Usage of energy saving systems (regenerative brakes, regenerated power storage devices). Trains with regenerative brakes, which generate power when braking and return the power to the overhead power lines for reuse, are being actively introduced.<br>• Noise and vibration reduction: Efforts to reduce noise and vibration include the introduction of reduced noise trains, the use of long rails to reduce the number of rail joints, the use of synthetic crossties, and the installation of elastic material such as rubber between concrete railbeds and crossties.<br>• Use of natural energy: Solar panels and wind power generators are being installed on the roofs of stations, using sunlight and wind to generate the power used by the stations. | Link 1 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **Russian Railways** | Electrification program: Trans-Siberian Railway (TSR) | Russia / 2002 (TSR electrification completed) – 2030 (upgrades and modernization) | Electrify the entire Trans-Siberian Railway network (~9,300 km) and additional rail corridors | Russia has accelerated electrification programs since 2000 with the stated goal of electrifying all major corridors in its rail network. Trans-Siberian railway electrification, which technically began 1929, was completed in 2002. This important freight corridor has seen increasing traffic – particularly international container freight – despite the challenging winter weather conditions. Traction voltage used is 3 kV DC, with 25 kV AC in some sections. The entire railway is a double track network.<br>As part of a 3.2 trillion Rouble ($64B CAD) modernization and upgrading program, state-owned Russian Railways will electrify more than 7,400 km of track over the period 2008 to 2030, financed in part through public-private partnerships. | Link 1<br><br>Link 2 |
| **Indian Railways** | Broad gauge rail network electrification | India / 2014 - 2023 | Completely electrify India's broad gauge rail network by 2023 | The Government of India has prioritized rail electrification to reduce costs, improve capacity and reduce dependence on foreign oil. 5,186 route kilometers were electrified between 2014 and 2017 alone. 58%, or 37,500 km, of the national broad gauge network is electrified as of early 2020. The remaining 27,000 km will be electrified between 2020 and 2023. Approximately two thirds of freight is currently moved by electric traction in India. | Link 1<br><br>Link 2<br><br>Link 3 |
| **Central Government of the People's Republic of China** | Rail and transport hub electrification | China | | China has seen rapid growth in railway electrification due to an extensive railway infrastructure investment program under the last three successive Five-Year Plans. Investment in railway fixed assets has exceeded 800 billion Yuan ($190B CAD) in each of the past five years. Nearly half (47%) of the Chinese railroad track was electrified by 2010. Close to 70% of China's 127,000 km rail transport network was electrified by the end of 2017. This includes over 22,000 km of high speed rail – the longest such national network in the world. In 2016, 75% of all rail activity (passenger and freight combined) was on electric trains, up from 21% in 1995. China is also pursuing the electrification of all major shipping ports and harbours on the Yangtze River and at other intermodal shipping hubs. | Link 1<br><br>Link 2 |

| Owner/ Stakeholders | Title | Location / Timeline | Objectives | Description | Links |
|---|---|---|---|---|---|
| **International Union of Railways (UIC)** | Energy Efficiency Best Practice Workshops | Global / 2015 - 2017 | To facilitate the exchange of experiences to provide practical advice to practitioners attempting to achieve energy efficiency improvements, and thereby reduce costs and emissions. | In recent years many projects have delivered promising results for energy efficiency (Railenergy, CleanER-D, Energy consumption, Parked Trains, etc.) and other important ones are ongoing (MERLIN, Shift2Rail). What is in many cases lacking is knowledge related to practical implementation and optimization in the field. This project develops and communicates best practices for rail energy efficiency through a series of workshops. These workshops will bring together experts to share experiences, find solutions and overcome obstacles.<br><br>This series was followed-up by a European-specific workshop series from 2018-2020. | Link 1<br><br>Link 2 |
| **International Union of Railways (UIC)** | Door to Door | Global / 2018 - ongoing | To support sustainable mobility strategies in countries and regions globally, by providing expertise on last mile mobility best practice in the passenger rail sector. | Passenger rail use could be increased through the provision of enhanced last mile mobility options for travellers. A modal shift to rail will lead to a wide range of environment, social and economic benefits, such as reduced congestion, improved air quality, reduced CO2 emissions, and equitable access to transportation services. This project assesses various options from multiple perspectives to provide guidance on last mile mobility solutions. | Link 1 |

# Railway Association of Canada Proposal:
## *Rail Pathways Initiative*
## *Phase 2: Rail Decarbonization Roadmap for Canada*

**Septmber 18, 2020**

**Attn:** Peter Bedrossian, Director, Regulatory Affairs, Railway Association of Canada

Contents

# 1 Introduction

Dear Peter,

On behalf of Pollution Probe and the Delphi Group, I'm pleased to submit the enclosed proposal for the development of Rail Pathways Initiative Phase 2: Rail Decarbonization Roadmap for Canada. This government and industry backed initiative represents an important step forward in helping to maintain and grow the rail sector's role as a leading low-carbon provider of freight and passenger transport throughout Canada. Efforts to further reduce the carbon intensity of the sector will help to future proof it in light of decarbonization efforts beginning to take place within other transportation modes, and regulatory efforts to reduce economy-wide greenhose gas emissions. Phase 2 will culminate in the development of a Roadmap which leverages best practices, emerging technologies and Canadian capacity to chart a course towards a sustainable and achievable future for the sector.

We have assembled a high-quality team with extensive experience to carry out this project and look forward to working with you. Should you require any further information on our proposal, please do not hesitate to contact me.

Sincerely,

Steve McCauley
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## 1.1 High-Level Scope of Work

We understand that the Railway Association of Canada (RAC) and Transport Canada (TC) are looking to identify and assess GHG reduction opportunities in Canada's rail sector to inform decarbonization priorities in the years and decades ahead. Building off of the successes achieved through Phase 1 of the Rail Pathways Initiative and the MOU between TC and the RAC, this initiative will deliver on the following objectives:

1. Develop an analytical framework for assessing GHG reduction opportunities in Canada's rail sector.
2. Identify and assess potential GHG reduction measures.
3. Create a multi-stakeholder work plan for GHG reduction actions.
4. Develop and initiate a Roadmap implementation strategy.

Objectives 1 to 4 will be addressed in a Rail Decarbonization Roadmap document, which will include stakeholder-informed decarbonization targets for Canada's rail sector. The assessment of actions will include economic as well as environmental considerations, and will be firmly situated in a Canadian context. The development of the Roadmap and its recommendations will be stakeholder-informed, leveraging and building upon the public-private Steering Committee established during Phase 1 of the Pathways Initiative.

Means of attaining the information required to assess options and develop a decarbonization roadmap will include the engagement of subject matter experts and a review of literature and data pertinent to rail decarbonization in a Canadian context. Rather than taking a prescriptive or top-down approach, the Roadmap will be a collaborative effort, with recommendations and targets informed by input received from key stakeholder groups. Likewise, the implementation of the Roadmap will be collaborative in nature, with contributions envisioned from public, private and civil society entities. The project team will catalogue and share with the Project Authority all data sources and any analytical frameworks used over the course of the project, in addition to other resources referenced.

Phase 2 will take approximately six months to complete. The primary deliverables will be the Rail Decarbonization Roadmap and the Implementation Strategy. With the conclusion of Phase 2, key rail sector stakeolders should have a clear picture with regard to which actions can contribute the most to decarbonization efforts, which are feasible in the short, medium and long term, and what the optimal roles of different stakeholders are in the implementation of decarbonization actions.

## 1.2 Proponent Overview

**About Pollution Probe**

Pollution Probe is a national, not-for-profit; charitable organization that exists to improve the health and well-being of Canadians by advancing policy that achieves positive, tangible environmental change. Pollution Probe has a proven track record of working in successful partnership with industry and government to develop practical solutions for shared environmental challenges.

In recent years, Pollution Probe's work has been focused around three broad areas: transportation, energy, and human health. The increasing synergies between these three areas, and the increasing attention that each are receiving on the national and global stage has greatly enhanced both the relevance and importance of Pollution Probe's work. Contributing to the decarbonization of Canada's transportation sector is a core organizational objective, and its experience and expertise enable it to take a holistic approach to meeting this objective, developing solutions that account for the most salient factors and impacts in a pragmatic manner.

**About The Delphi Group**

The Delphi Group is a sustainability and environmental risk management consultancy delivering new and innovative ways to approach climate change and sustainability. We work together with organizations to identify, track, manage and mitigate non-financial risks to drive environmental and economic performance. We develop and effectively deploy environmentally responsible organizational strategies that focus on the bottom line and enhance long term performance. We help clients turn environmental initiatives into strategic imperatives that focus on reducing waste, minimizing energy consumption, and optimizing raw material use. Our extensive experience and our highly skilled team provide you with the comprehensive skills set needed to accomplish your project needs. We currently work with 32 of Canada's Top 100 companies and a wide range of public sector clients. They trust our proven capabilities to deliver the quality service they require. We are a pioneer in the sustainability field and clients from around the globe hire us for our expertise, insights, wisdom, and the practical know-how we have accumulated over the course of doing business for 30 years.

# 2 Approach

The statement of work for the development of the Rail Decarbonization Roadmap consists of the following central elements:

**1. Develop analytical framework for assessing GHG reduction opportunities in the rail sector**

- To include engagement of Steering Committee established during Phase 1, increased engagement with provinces and territories, as well as additional subject matter experts on technology readiness, business case for technologies, modal shift potential, and GHG reduction potential of specific mechanisms.
- Will include identification of barriers to the implementation of decarbonization measures, as well as gaps in public and private engagement on rail decarbonization in Canada.

- Project team will engage experts in federal and provincial/territorial governments, industry, and civil society through one-on-one interviews and consultations.

2. **Identify and assess potential GHG reduction measures**

- Identify GHG reduction measures and assess their potential with regard to the deep decarbonization of the rail sector using the assessment framework developed in step 1.
- Will include short, medium and long term measures which would achieve deep decarbonization in the rail sector and which would facilitate modal shift.
- Measures to be examined will go beyond emerging technology adoption to include potential financing mechanisms, RD&D required in short and medium terms to achieve long term targets, mechanisms to address regulatory barriers, and an exploration of how policy can address barriers.

3. **Create multi-stakeholder work plan for GHG reduction actions**

- For each decarbonization mechanism, appropriate roles for key stakeholder groups (e.g., federal and provincial governments, industry, academia, etc.) will be identified, as will key steps required to implement decarbonization mechanisms.
- Certain elements of the strategy and work plan will be informed by global roadmapping and RD&D initiatives through which the effectiveness of certain measures has been validated.
- GHG reduction targets that are achievable based on barriers, gaps and opportunities identified will be proposed.

**4. Develop and initiate Roadmap implementation strategy**

- Develop Roadmap implementation strategy prioritizing short, medium and long term actions and indicating which stakeholder groups should have leadership roles.
- Communications and outreach on Roadmap to inform stakeholders and the public on rail decarbonization initiatives and progress.
- Establishment of monitoring and measurement mechanism to ensure the implementation and effectiveness of Roadmap recommendations.

Each of the core elements above will be addressed in a separate section of the Roadmap document. The Roadmap will have a structure and style comparable to that of the Landscape Document completed during Phase 1 of the Pathways Initiative.

The Railway Association of Canada (RAC) and Steering Committee member organizations will be given the opportunity to review and comment on two draft versions of the Roadmap prior to finalization. Socialization of the Roadmap will be at the discretion and direction of the RAC.

# 3 Project Budget

| Rail Pathways Phase 2: Budget | | | | | | |
|---|---|---|---|---|---|---|
| Project Team Member / Role | Steve McCauley / Senior Project Advisor | Matt Beck / Senior Technical Advisor | Cara LaRochelle / Senior Consultant | Derek May / Consultant | Mariana Eret / Research Analyst | Total Hours |
| Hourly Rates | $200 | $241 | $215 | $160 | $130 | |
| Project Tasks | | | | | | |
| **1. Develop analytical framework for assessing GHG reduction opportunities in the rail sector** | | | | | | |
| Engagement of Steering Committee | 7.5 | | 7.5 | 7.5 | 3.5 | 26.0 |
| Identification of barriers to the implementation of decarbonization measures, as well as gaps in public and private engagement on rail decarbonization in Canada | 3.5 | 2.0 | 15.0 | 7.5 | | 28.0 |
| Engagement with provincial/territorial governments | 7.5 | | 12.0 | 12.0 | 5.0 | 36.5 |
| Engagement with experts: one-on-one interviews and consultations with industry, government and subject matter experts subject matter experts on technology readiness, business case for technologies, modal shift potential, GHG reduction potential of specific mechanisms. | 3.5 | | 22.5 | 22.5 | | 48.5 |
| **2. Identify and assess potential GHG reduction measures** | | | | | | |
| Identify GHG reduction measures and assess using the assessment framework developed in step 1. (This will include short, medium and long term measures which would achieve deep decarbonization in the sector and which would facilitate modal shift). Measures to be examined will include potential financing mechanisms, RD&D required in short and medium term to achieve long term targets, mechanisms to address regulatory barriers, and an exploration of how policy can address barriers. | 3.5 | 3.5 | 35.0 | 22.5 | 7.5 | 72.0 |
| **3. Create work plan for GHG reduction actions** | | | | | | |
| For each decarbonization mechanism, identify appropriate roles for key stakeholder groups (e.g., federal and provincial governments, industry, academia, etc.) and key steps required to implement decarbonization mechanisms | 7.5 | 3.5 | 30.0 | 15.0 | | 56.0 |
| Review of global roadmapping and RD&D initiatives in which the effectiveness of measures has been validated to inform the strategy and work plan | | | | 20.0 | 15.0 | 35.0 |
| Propose GHG reduction targets that are achievable based on barriers, gaps and opportunities identified | 3.5 | 2.0 | 7.5 | 7.5 | | 20.5 |
| **4. Develop and implement Roadmap implementation strategy** | | | | | | |
| Develop implementation strategy | 7.5 | 2.0 | 22.5 | 22.5 | | 54.5 |
| Communications and outreach on Roadmap to inform stakeholders and the public on rail decarbonization initiatives and progress | | | 2.0 | 3.0 | | 5.0 |
| Monitoring and measurement mechanism to ensure the implementation and effectiveness of Roadmap recommendations | 5.0 | | 5.0 | 5.0 | | 15.0 |
| | | | | | | |
| *TOTAL HOURS* | 49.0 | 13.0 | 159.0 | 145.0 | 31.0 | 397.0 |
| *Subtotal Fees* | *$9,800* | *$3,133* | *$34,185* | *$23,200* | *$4,030* | *$74,348* |

The project team will deliver the final Rail Decarbonization Roadmap for Canada in April of 2021. An initial step in the project will be to develop a detailed work plan, including timelines and key milestones, in consultation with the RAC.

# 4 Project Team Members

**Steve McCauley, Senior Director, Policy, Pollution Probe**

Steve is a leading expert in the environmental field, having spent more than twenty years managing the development and implementation of policies and programs in numerous areas, including climate change, air pollution, and the advancement of renewable energy and clean technologies. He led the Government of Canada's federal and international regulatory actions on greenhouse gas emissions, air contaminant emissions and chemicals management in transportation, energy and other industrial sectors. Since joining Pollution Probe, Steve has driven the implementation of the Pathways Initiative and the organization's participation in the development of the *Pan-Canadian Framework on Clean Growth and Climate Change*. He recently led the delivery of the project *Accelerating the Deployment of Plug-In Electric Vehicles in Canada and Ontario* in partnership with Bruce Power, and the NRCan-funded project *The Electrification of Transportation in Canada: State of Play, Barriers and Regional Opportunities*, which focussed on ZEV deployment in Atlantic Canada and the Prairies. He most recently led a project with Toyota Canada that culminated with the publication of a policy framework report entitled *Decarbonizing Transportation in Canada: Building a Foundation for* Success. Steve also worked closely with the Government of Ontario to deliver its Electric and Hydrogen Vehicle Advancement Partnership and currently sits on the Government of Canada's ZEV Advisory Committee.

**Matt Beck, Director, Climate, Sustainability and Transportation Services, The Delphi Group**

With over 15 years of experience in professional sustainability roles, Matt is a string advocate for best practices that can drive improved social, environmental and economic outcomes. Matt is particularly interested in the nexus of energy, technology and environment that is rapidly evolving how businesses can achieve success. Prior to joining Delphi, Matt was Director of Sustainability for Imaginea Energy, where his focus on enabling hydrocarbon production that fits within an evolving sustainable energy system drove development and integration of new benchmarking and reporting techniques for clean energy production. He also held roles at Husky Energy, green building and construction companies, as well as a consultant working in the agriculture and tourism sectors. He has also served as CTO of an online tourism operations company where he oversaw the implementation of a web platform and e-commerce technologies.

**Derek May, Senior Project Manager, Pollution Probe**

Since joining Pollution Probe as a Project Manager in 2013, Derek has led a number of files focused on reducing emissions from transportation sectors including light-duty, heavy-duty, and off-road vehicles, as well as rail. Derek has developed strategies and tools to facilitate the deployment of low-carbon vehicles, such as EVs, as well as low-carbon fuels. He has led projects in partnership with all three levels of government in Canada, as well as industry partners. He also represents Pollution Probe in several expert transportation committees and working groups.

**Cara Larochelle, Senior Consultant, The Delphi Group**

In her nearly two decades as a consultant, Cara has completed numerous technical research and policy development projects, which have incorporated widespread government, private sector and industry consultations. Cara's areas of expertise include energy efficiency, both in the transportation and built environment sectors; as well as policy development and program support. She has worked with a range of clients including federal and provincial government departments, municipalities, educational institutions and industry associations.

**Mariana Eret, Policy Analyst, Pollution Probe**

Mariana's work at Pollution Probe has included conducting research on low-carbon transportation fuels, technologies and practices to support the implementation of Pollution Probe's Pathways Initiative. Prior to her current role, Mariana interned at Canada Green Building Council, assisting staff with special Chapter initiatives that included research on innovative green building, and energy and transportation policies implemented by Ontario municipalities. Mariana holds a Master of Environmental Studies degree from York University and a Master of Arts (Honours) degree in Environmental Sustainability from Dundee University, Scotland.

## 4.1 Project Delivery Experience

Pollution Probe and the Delphi Group have successfully partnered on numerous transportation initiatives. Our project team's work in the transportation sector has spanned well over two decades, and as our experience and expertise has grown, so too has the scope of our transportation-related activities. Our body of work on Canada's transportation sector in recent years includes the following projects:

- ***Modal Optimization as a Contributor to Reducing GHG Emissions in Canada***: The objective of this project was to identify modal optimization solutions that can be implemented by a variety of stakeholders and that will contribute to Canada's climate targets, including net zero by 2050, by improving energy efficiency across the transportation sector and determining the optimal way to move people, goods and services. The study examines trends, GHG reduction potential, economic impacts, key stakeholder groups, technology readiness, and optimal locations in Canada related to mode shifting.
- ***Guide to Electric Vehicle Charging in Multi-Unit Residential Buildings:*** This guide is aimed at providing a more detailed understanding of the specific process and unique considerations associated with the installation of EV charging infrastructure in MURBs. The guide outlines the types of installation considerations that may vary by building (e.g., electrical capacity, configuration of building's electrical system, parking locations, etc.), provides "step-by-step" guidance to walk relevant stakeholders through the necessary steps for EV charging station installation, including roles and responsibilities, and identifies potential barriers and strategies for streamlining the installation process.
- ***Opportunities for Low-Carbon Mobility Actions in Canadian Municipalities: Best Practices and Guidance***: This comprehensive study explored a wide variety of collaborative actions that Canadian cities can lead to encourage the use of ZEVs across all modes of transport to address GHG and air

pollutant emissions and alleviate congestion. The project report serves as a public-facing best practice guide for municipalities to inform the establishment of low-carbon mobility actions. Four broad areas for municipal action to encourage the use of ZEVs were identified: low-emission zones, restricted road access, parking space removal, and congestion pricing. These broad areas for action were supplemented with 32 complementary areas on topics ranging from infrastructure and public transit, to financial levers, regulatory instruments, partnerships, and urban planning and design.

- ***Workshop on Ultrafine Particle Emissions from Transportation***: While CO2 emissions have been steadily declining in most vehicle classes, there are concerns that the introduction of certain fuel-saving technologies, such as gasoline direct injection (GDI) engines and turbochargers, has inadvertently caused a major spike in emissions of ultrafine particle (UFP) emissions, which can lead to adverse impacts on air quality and human health. This workshop showcased findings from top researchers in a variety of fields at the nexus of transportation and human health. The event also provided opportunities to pose questions to the experts and discuss concerns related to the emerging issue of UFPs and transportation. It was attended by a wide range of experts from industry, government, academia and the not for profit sector.

- ***Zero Emission Vehicle Charging in Multi-Unit Residential Buildings and for Garage Orphans***: This project culminated in the development of an actionable guidance document focused on the deployment of EVSE in MURBs and garage orphan households, targeted at key stakeholder groups such as all levels of government, EVSE providers, residential property developers and managers, and utilities. The project report outlines the unique barriers to ZEV charging faced by these residents and identifies best practices and potential solutions for addressing them. The research methodology for the study combined an in-depth literature review with a series of interviews with key subject matter experts and stakeholders.

- ***Framework for Municipal Zero Emission Vehicle Deployment***: This project's objective was to provide support to Canadian municipalities seeking to engage in activities to increase ZEV uptake in their communities. The Framework report presents a comprehensive suite of actions that any municipality can customize to meet its needs in reducing emissions from all modes of motorized transport through the increased deployment of ZEVs. These actions are divided into two distinct matrices (one for privately-owned passenger vehicles, and the other for fleets and medium- and heavy-duty vehicles) and fall under several prominent categories, each of which necessitates the expertise of unique stakeholder groups that operate within different subsets of the transportation system. Common challenges and barriers associated with different types of actions are described, along with a series of potential solutions. Report contents and findings were informed by secondary research, a series of interviews with subject matter experts from across Canada, two expert workshops, and follow-up consultations.

- ***Locomotive Emissions Monitoring Program***: For more than seven years, Pollution Probe has participated in the MOU Management and Technical Review Committees of the Locomotive Emissions Monitoring (LEM) Program led by the Railway Association of Canada and Transport Canada. The Delphi Group recently assumed responsibility as the lead developer of annual LEM reports.

- ***International Case Studies on Goods Movement Strategies:*** Countries such as United States, United Kingdom, Germany, Sweden, and China have implemented sustainable, low-carbon goods movement strategies and measures to reduce GHG emissions, improve local air quality, decrease road congestion, and increase productivity through the efficient movement of goods. This project looked at international sustainable and low-carbon goods movement strategies for on-road freight in six case study countries. Policies reviewed include measures to improve fuel consumption, emission standards, alternative fuels and technologies, green freight programs, operational practices, and others.

- ***City of Toronto Electric Mobility Strategy Assessment Phase***: Toronto's Electric Mobility Strategy is a core component of its long-term climate action plan, TransformTO. This project laid the groundwork for the Strategy's development. It involved extensive primary and secondary research including a literature review, case study analysis, data gathering and analytics, a stakeholder workshop, and expert interviews. An advisory group consisting of key local and regional stakeholders was assembled, and a comprehensive report was developed based on our research and stakeholder consultation. The report provides general background information on electric mobility and charging options, and explores current EV adoption levels and the supportive programs in place in Toronto. It then looks at municipal best practices in facilitating electric mobility, and provides advice and resources on how to address key issues through the Strategy.

- ***Metro Vancouver Light Duty Vehicle Emission Reduction Policy and Program Options:*** Development of a multi-criteria analysis of various policy and program options for reducing air pollutant and GHG emissions from light duty vehicles in Metro Vancouver. The results of two stakeholder workshops and policy/program option research were summarized in a final report.

- ***Decarbonizing Transportation in Canada: Building a Foundation for Success:*** This project involved a review of current policies that influence transportation energy use and the development of a comprehensive policy option framework intended to support the reduction of energy demand and GHG emissions from transportation in Canada. The project report articulates and explains the key factors that influence energy demand in Canada's transportation sector, and highlights how these factors interact at a system-wide level.

- ***The Electrification of Transportation in Canada: State of Play, Barriers and Regional Opportunities:*** Undertaken with support from Natural Resources Canada and automotive and electricity system stakeholders, this project focussed on research and extensive stakeholder consultation to develop solutions for enhancing the deployment of zero emission vehicles (ZEVs) in Atlantic Canada and the Prairie provinces.

- ***Electric and Hydrogen Vehicle Advancement Partnership***: The Probe-Delphi team serves as the independent evaluator of the EHVAP, which is tied to the Government of Ontario's Electric Vehicle Incentive Program (EVIP) and is intended to build the foundation for the achievement of Ontario's goal of 5% ZEV sales by 2020. All members are required to submit ZEV advancement plans, which are evaluated as sufficient/insufficient to contribute towards meeting the 5% goal, and the evaluators also work with automakers and other partners to refine and strengthen their plans.

- ***Canada's Zero Emission Vehicles Strategy***: Extensive participation as a member of the Advisory Committee of this national initiative, as well as in the infrastructure, public awareness and total cost of ownership working groups.
- ***Low Carbon Vehicle Technology***: A study for NRCan outlining vehicle technology and infrastructure technology for alternative fuels. The study assessed vehicle applicability for the fuels from LDVs to class 8 trucks given the current and near-term state of technology.
- ***Alternative Fuels Report***: A study prepared for the Ontario Trucking Association, Canadian Natural Gas Vehicle Alliance, Enbridge and Union Gas to assess the use of alternative fuels in the truck freight transportation sector in Ontario and explore the GHG and cost impacts of a transition from diesel.
- ***EV-Grid Gap Analysis Study***: Identified barriers and opportunities related to EVs from the perspective of Canadian electricity providers. Interviews, research and analysis were used to gain insights into state of utility preparedness regarding EVs and key findings were shared with NRCan.
- ***The Pathways Initiative***: A stakeholder-driven, expert-informed program designed to identify and assess best practices, approaches and technologies which can lead to deep decarbonization in the Canadian transportation sector. A kick-off workshop for the Initiative was held in March, 2016, bringing together over 80 global transportation experts and stakeholders.
- ***Roundtable Series on Emerging Transportation Technologies***: Brought together global leaders from industry, academia and government for intensive technical dialogue on the future of transportation in Canada in areas such as light-duty vehicles, heavy-duty vehicles, off-road vehicles and biofuels.
- ***HDV and LDV Standards***: Ongoing technical and policy support to Environment and Climate Change Canada, focussing efforts on stringency, barriers and opportunities associated with implementation.
- ***Report on Cost and Adoption Rates for Fuel Saving Technologies for Trailers in the Canadian Freight Sector***: A stakeholder consultation and expert analysis process to assess costs, performance, and uptake levels of fuel-saving technologies for heavy duty trailers, completed in collaboration with the International Council on Clean Transportation.
- ***Business Case for Electric Vehicle Use in Service Vehicle Fleets (Project EVAN)***: A business case and environmental justification for fleet managers in both the public and private sectors to incorporate EV technology into their procurement and vehicle replacement plans.
- ***City of Calgary Electric Vehicle Strategy***: Development of a comprehensive recommendations report as well as tools for municipality-led actions that promote and support the adoption of EVs.
- ***Electric Mobility Adoption and Prediction (EMAP) Project***: A multi-year partnership with five major Canadian utilities to develop EV deployment strategies and assess local grid readiness for high levels of EV uptake.
- ***Ontario Electric Vehicle Power Providers' Working Group***: Pollution Probe is the only national ENGO represented in this group, which was created by Ontario's Ministry of Energy to inform the

province’s updated Long-Term Energy Plan, and is comprised of technical experts from utilities and the Government of Ontario who are focused on the development of progressive EV-related policies and regulations.

- ***Workshop on Low Carbon Fuel Standards***: A multi-stakeholder process that developed guidance for British Columbia’s Renewable and Low Carbon Fuels Regulation.