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A quantitative method for evaluating ecological risks associated with long-term degradation of deep-sea plastic-containing infrastructure

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ABSTRACT

Presented herein is a newly developed quantitative approach for assessing potential ecological risk resulting from long-term degradation of deep-sea plastic-containing infrastructure. The risk characterisation involves four iterations of modelled 'risk' through forward or backward calculation of a deterministic hazard quotient, mathematically defined as the ratio of estimated exposure to a reference dose (or concentration) for a similar exposure period. The assessment focuses on direct effects of microplastics exposure, wherein exposure concentrations are based on modelled estimates of microplastic mass formation resulting from structure deterioration over time. Predicted no effect concentrations (PNECs) protective of slightly-to-moderately disturbed ecosystems and ecosystems of high conservation value were determined based on a species sensitivity distribution (SSD), in accordance with the current Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Each iteration of risk characterisation is performed irrespective of burial, with varying exposure unit dimensions (i.e. geographically localised and broader regions of microplastic dispersal) and degrees of plastic degradation, designed to conservatively bound the risk characterisation. Additionally, two SSDs derived from different ecotoxicological data sets prioritising either particle shape or marine species are also provided for a sensitivity analysis of the PNEC. Thus, the bounding exercise encompasses all possible outcomes. The risk characterisation approach is reviewed for a case study of two larger plastic-containing flowline assets in an oil production field offshore of Australia. The outcome of the risk assessment is the same for all model iterations: degradation of the subsea plastic-containing flowlines does not pose a risk to the local marine community.

Keywords: degradation, ecological risk, microplastics, NEBA, net environmental benefit analysis based comparative assessment, offshore decommissioning, plastics, risk assessment, toxicity.

Introduction

Australia has in the order of 5000 km of offshore export and inter-field pipelines, 3200 km of infield flowlines and static umbilicals, 57 fixed facilities, and 11 floating facilities currently operating in Commonwealth waters (Advisian 2020). More than half of Australia's offshore petroleum assets are older than 20 years, with some exceeding 50 years and, consequently, are predicted to be approaching the end of the service lifetime (Melbourne-Thomas *et al.* 2021) and require decommissioning soon. The view of the National Offshore Petroleum Safety and Environmental Management Authority is that the designated decommissioning approach must provide equal or better environmental outcomes than default full removal of the infrastructure (considered the 'best case' expectation under current legislation) and meet as low as reasonably practicable (ALARP) levels of risk. Several of the decommissioning options commonly considered in net environmental benefit assessment based comparative assessment (NEBA-CA) involve leaving subsea structure *in situ* (in part or in whole) (Schubel 2020), as the

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structures are known to support diverse and thriving ecological communities (Fowler and Booth 2012; Claisse *et al.* 2014*a*, 2014*b*; Fowler *et al.* 2015; Todd *et al.* 2018; Schubel 2020).

Recently, the ecological risks associated with long-term degradation of plastic-containing infrastructure in offshore oil/gas fields has garnered focus from regulators when evaluating decommissioning strategies for these developments. Microplastics (< 5 mm) are expected to form over time as plastic components deteriorate through physical, chemical, and biological processes. The adverse effects commonly discerned when the impacts of microplastics on the marine environment are discussed include the following: (1) the physical and toxicological effects of microplastic particle exposure, and (2) toxicological effects associated with leached plastic additives and monomers unreacted in the plastic material and hydrophobic organic chemicals (HOCs) from the surrounding environment sorbed to microplastic particles (GESAMP 2015; EPA 2016). To date, risk assessments of microplastics have primarily examined ecological risk associated with exposure to suspended (buoyant) microplastics based on modelled estimates of global plastic input (e.g. Everaert *et al.* 2018), ranges of exposure concentrations currently measured in the global aquatic environment (e.g. Burns and Boxall 2018; Besseling *et al.* 2019), or site-specific measurements (e.g. Jung *et al.* 2021; Pan *et al.* 2021). To the best of the authors' knowledge, an approach for quantitative assessment of the potential ecological risks resulting from long-term degradation of subsea, plastic-containing structures does not exist.

Presented herein is a procedure developed expressly for this purpose, employed at an oil production field offshore of Australia. The risk characterisation approach is reviewed for a 12-inch rigid flowline and piggybacking 2-inch coiled tubing flowline in the field, which represent two of the development's greatest potential sources of microplastics. These structures are each 8+ km in length and situated at greater than 130 m at depth along the seabed. The 12-inch rigid flowline is insulated in four-layer polypropylene (PP), which includes successive layers of fusion-bonded epoxy (FBE) primer, copolymer adhesive, and foamed PP, encased in solid PP with a wall thickness of 3 mm. The 2-inch flowline is insulated in three-layer polyethylene, comprising successive layers of FBE primer and copolymer adhesive, encased in solid high-density polyethylene (HDPE) with a wall thickness of 1.65 mm. The assessment focuses on direct effects of microplastics exposure, wherein exposure concentrations are based on modelled estimates of microplastic mass formation resulting from structure deterioration over time. Ecological risks associated with exposure to leached additives and monomers and HOCs associated with microplastic particles are considered negligible based on a review of the peer-reviewed scientific literature (this topic is beyond the scope of this paper).

Background

Plastic degradation

The mechanisms for plastic degradation can be classified as follows: (1) physical, referring to changes in the bulk structure; (2) chemical, referring to factors which result in changes at the molecular level (e.g. bond cleavage) that weaken and disintegrate the material; or (3) biological, referring to the mineralisation of the material and/or its degradation byproducts by biota (e.g. microbes) in the environment (Chamas *et al.* 2020). The abiotic and biotic processes act in tandem (potentially at vastly different rates), with abiotic degradation leading to more labile products that promote biological degradation (Albertsson and Karlsson 1990; Lee *et al.* 1991; Gautam *et al.* 2007; Wayman and Niemann 2021). Typically, chemical degradation in the natural environment involves either hydrolysis or oxidation, both of which are accelerated by ultraviolet (UV) radiation and heat. Ultimately these processes result in chain scission and depolymerisation, weakening the material and making it susceptible to fracture and deterioration.

The mechanisms responsible for the breakdown of plastics and associated reaction kinetics vary based on the chemical structure of the polymer, though plastic degradation generally proceeds very slowly under natural environmental conditions (Chamas *et al.* 2020). For polyolefins (e.g. PP and HDPE), degradation often is initiated by photooxidation where UV radiation provides the activation energy required to initiate the incorporation of oxygen atoms into the polymer (Ranby and Rabek 1975; Guillet 1980). Specifically, UV radiation electronically excites (and thus makes reactive) certain groups in the polymer (e.g. carbonyl groups; often impurities introduced during the manufacturing process), or dissociate polymer bonds to radicals (photolysis). These photolytic species participate in chain propagation reactions in the presence of oxygen that result in bond cleavage (referred to as chain scission when in the polymer backbone) and depolymerisation, or cross-link through radical recombination when oxygen availability is limited. Photooxidation is restricted to the surface layers of the polymer where the material interacts with light.

In the absence of UV radiation, most plastic polymers are stable for very long periods of time (Grassie and Scott 1988) at ambient temperatures; high temperatures (> 350°C) are typically required for thermally-induced oxidation (Ahmad *et al.* 2014). For example, oxidation of PE does not occur at appreciable rates when exposed to temperatures below 100°C without UV radiation (Gardette *et al.* 2013). For this reason, plastic degradation is expected to be considerably slow at depth in the marine environment.

Plastic degradation rate in the marine environment is expected to decrease with depth due to declining dissolved oxygen availability in the water column. Dissolved oxygen

content typically follows a monotonically decreasing profile along water depth as a consequence of microbial decomposition, lack of atmospheric contact for diffusion, and absence of photosynthesis. Additionally, sediment burial of plastic limits infrastructure interaction with oxygen and is expected to further limit the oxidation of plastics. Oxygen flux across the sediment–water interface and downward transport of oxygen though the sediment column is generally restricted to the top several centimetres (Jørgensen and Revsbech 1985; Glud *et al.* 1994), as oxygen is depleted for microbially-mediated mineralisation of organic matter and re-oxidation of reduced inorganic metabolites.

The rate of polymer degradation is also expected to vary over time with chemical and morphological changes. In crystalline regions where interstitial space is limited, the chain scissions occurring as a result of photooxidation are generally followed by an immediate recombination (cross-linking) caused by the inability of the polymer chains to move and attain different physical conformations and due to slower diffusion of oxygen (Liu *et al.* 2019; Grause *et al.* 2020). Thus, crystalline regions degrade more slowly than amorphous regions and degradation rates are expected to taper as the amorphous regions are eliminated. Additionally, partial polymer degradation can lead to secondary cross-linking and/or crystallisation in amorphous regions adjacent to crystallites, further slowing degradation (Restrepo-Flórez *et al.* 2014). In contrast, morphological changes over time resulting from mechanical degradation produce changes in surface roughness and additional surface area available for oxidation, increasing the degradation rate. Mechanical degradation may be enhanced by vortex-induced vibrations for free spanning flowlines and stresses during deployment of the flowlines. The relatively constant cool temperatures at depth in the sea are expected to reduce thermal movements (i.e. expansions and contractions) of the flowlines and induced stresses that may contribute to mechanical degradation of the plastic.

Microplastic ingestion

Numerous field studies are available demonstrating microplastic particle ingestion in aquatic biota (Van Cauwenberghe and Janssen 2014; Bellas *et al.* 2016; Alomar and Deudero 2017; Courtene-Jones *et al.* 2017; Güven *et al.* 2017; Jabeen *et al.* 2017; Leslie *et al.* 2017; Ory *et al.* 2017; Pazos *et al.* 2017; Silva-Cavalcanti *et al.* 2017). Effect studies with microplastics have explored a range of toxicological endpoints including survival, growth, reproduction, and behavioural and biochemical endpoints. Toxicological response to microplastic exposure varies across test species and is suspected to be affected by particle size and morphology (Gray and Weinstein 2017; Hodson *et al.* 2017; Ziajahromi *et al.* 2017).

The trophic transfer of microplastics and subsequent gradual enrichment (biomagnification) through the food web has been hypothesised, but not demonstrated under environmentally realistic conditions (Burns and Boxall 2018). In fact, several studies propose that microplastic presence in fish and invertebrates is ephemeral and microplastics are readily eliminated from the body (Ugolini *et al.* 2013; Hämer *et al.* 2014; Mazurais *et al.* 2015; Blarer and Burkhardt-Holm 2016; Grigorakis *et al.* 2017; Güven *et al.* 2017). Thus, the available evidence suggests that microplastics do not appreciably accumulate within marine biota, nor biomagnify through the food web.

Methods

Overview

This portion of the paper describes the methodology used to evaluate the potential for adverse ecological effects resulting from plastic degradation and subsequent microplastic formation along the flowlines. The ecological risk characterisation combines the exposure profile with a reference concentration protective of the marine community for a similar exposure period, to produce numerical indices of potential health effect. The risk characterisation is performed in four iterations with varying exposure unit dimensions (i.e. geographically localised and broader regions of microplastic dispersal) and degrees of plastic degradation, designed to provide an examination of model sensitivity to the various inputs and conservatively bound the risk characterisation.

Exposure medium

The PP and HDPE comprising the exposed solid layer of the 12-inch rigid flowline and 2-inch coiled tubing flowline, respectively, are less dense than sea water. For the purposes of the risk characterisation, microplastics forming along the flowlines are assumed either neutrally or positively buoyant and remain in the water column where they may interact with the local ecological community.

Buoyant plastics may develop biofouling and become deposited in sediments. Evaluation of sediments under the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG 2018) involves a tiered approach (Simpson *et al.* 2013). For contaminants without a sediment quality guideline value (as is the case for microplastics), the evaluation involves comparison of site concentrations with background concentrations in reference sediments of comparable grain size from appropriate sites. The information for a proper background assessment of microplastics is scarce and an area for further research. However, microplastics are ubiquitous in the marine environment (Thompson *et al.* 2009) in part due to relatively large land-based inputs and, thus, concentrations of microplastics in oil/gas field sediments are expected to be

predominantly the result of contributions from sources other than plastic structures in the field. For perspective, the volume of plastics entering the oceans per year from land-based sources is estimated between 4.8 and 12.7 Mt (Jambeck *et al.* 2015), an amount that is anticipated to grow with escalating plastics production worldwide. For these reasons, microplastics in sediment have not been further considered in the risk assessment.

Hazard quotient (HQ) calculation

Each iteration of 'risk' characterisation involves forward calculation (Iteration #2–#4) or back-calculation (Iteration #1) of a deterministic hazard quotient (HQ). The HQ metric provides a screening-level evaluation of the *potential* (not probability) for adverse ecological effects by comparing a modelled exposure level over a specified time period to a no-effect level for a similar exposure period:

$$\mathrm{HQ} = \frac{\mathrm{PEC}}{\mathrm{PNEC}};$$

where PEC is the predicted environmental concentration and PNEC is the predicted no-effect concentration. HQ outcomes are typically reported to one significant figure (EPA 2004).

The HQ assumes that there is a level of exposure below which it is unlikely for even sensitive populations to experience adverse health effects. Thus, the exposure level is not considered to pose a risk to animal populations when less than or equal to the PNEC (HQ ≤ 1). Conversely, if the exposure level exceeds the threshold (HQ > 1), there *may* be concern for potential adverse effects, suggesting that further consideration of the potential for effect is warranted. A HQ > 1 does not guarantee that there are ecological receptors bearing a toxicological effect of concern (Tannenbaum 2003) and, furthermore, does not indicate adverse impacts to populations or communities of organisms (Barnthouse 2008). Numerous authors have noted that the results of deterministic models often do not comport with visible evidence of population-level effects at terrestrial sites where these tools have indicated potential for ecological risk (Linzey and Grant 1994; Henning *et al.* 1997, 2003; Boonstra and Bowman 2003; Tannenbaum 2003, 2005; Barnthouse 2008). Thus, the HQ methodology can result in amplified predictions of potential harm to ecological receptors, and provides a very conservative approach to ecological risk characterisation.

The HQ model has been applied in several peer-reviewed ecological risk assessments of microplastics (Burns and Boxall 2018; Everaert *et al.* 2018; Besseling *et al.* 2019; Jung *et al.* 2021).

Rate of plastic degradation

In the present study, degradation and deterioration are used synonymously, to refer to overall mass loss from the initial polymer piece. Loss of microplastics fragments reduces the initial mass, without changing the total amount of plastic present. Thus, the rate of polymer degradation is assumed equal to the rate of microplastic formation.

The polymer degradation rate (r_d) is defined as the differential mass loss per unit time. As previously noted, the mechanisms and kinetics of degradation are contingent upon the intrinsic properties of the plastic and the ambient environmental conditions. Since degradation occurs principally at exposed surfaces, the rate of degradation also varies based on extrinsic properties such as the size and shape of the material, and extent of sediment burial. Thus, the degradation reaction is assumed proportional to the area of the exposed surface, which is expected to decrease as the flowline wall erodes. For annular cylindrical shapes such as the solid plastic casing insulating the flowlines, the following rate law is obtained (adapted from Chamas *et al.*, 2020):

$$r_d = -\frac{dm}{dt} = k_d \rho\, \mathrm{SA} = k_d \frac{m}{V} \mathrm{SA} = k_d m \left(\frac{2r_2}{r_2^2 - r_1^2} \right)$$

where:

- k_d is the linear rate (m/year) representing the perpendicular depth of plastic degraded per unit time (referred to as the 'specific surface degradation rate' [SSDR]), a variable that is contingent on the intrinsic properties of the plastic and varies based on environmental conditions;
- ρ is the density of the plastic (g/m^3);
- SA and r_2 are the surface area (m^2) and corresponding radius (m), respectively, of the outer wall of the solid plastic casing exposed to environmental factors that precipitate degradation;
- r_1 is the inner radius of the outer plastic casing (m); and
- m is the mass of the plastic casing thickness (g).

Assuming the density of the plastic and the length of the structure remain constant, integration and algebraic rearrangement yield solutions for the undegraded plastic mass as a function of time and time for complete degradation (derivation is included as supplementary information):

$$m_t = \rho \pi l \left[\left\{ \left(\frac{m_i}{\rho \pi l} + r_1^2 \right)^{\frac{1}{2}} - k_d t \right\}^2 - r_1^2 \right]$$

$$t_d = \frac{1}{k_d} \left\{ \left(\frac{m_i}{\rho \pi l} + r_1^2 \right)^{\frac{1}{2}} - r_1 \right\}$$

where:

- m_i and m_t are the initial mass of plastic material and mass at time t, respectively (g);
- l is the length of the cylindrical structure (m); and
- t_d is the time for complete degradation (year).

Chamas *et al.* (2020) derived SSDRs for several commonly studied plastics under various environmental conditions based on a detailed review of experimental studies of plastic degradation available in the peer-reviewed scientific literature, including PP and PE when immersed in shallow water and exposed to sunlight, and with/without rapidly degrading fillers and/or laboratory-based pre-treatments that accelerate degradation. Several of the water-based studies reviewed by Chamas *et al.* (2020) were performed in the field and, thus, SSDRs derived from these studies inherently reflect the effects of naturally occurring mechanisms that sustain or accelerate plastic degradation in shallow water environments (e.g. UV radiation, mechanical degradation by currents, and microbial action). In the present study, structure deterioration and sequential microplastic formation is calculated twice for each model iteration: once based on an SSDR reported by Chamas *et al.* (2020) for a shallow water environment with exposure to sunlight, and a second time incorporating degradation accelerants. Both calculations can be expected to overestimate degradation and provide a conservative range of lifetime estimates for the solid plastic layer insulating each flowline, as the deep marine setting lacks the necessary UV radiation and/or thermal energy to provide the activation energy required to initiate and/or sustain oxidative degradation. Furthermore, laboratory pre-treatments that accelerate degradation are not available in the natural environment.

Plastics are expected to undergo chemical and morphological changes as degradation proceeds and, thus, the SSDR is expected to vary with time. However, degradation is an irregular and inconsistent process, and this effect cannot be quantified or modelled at this time. Accordingly, plastic degradation is conservatively estimated using the high-end range SSDR reported by Chamas *et al.* (2020) for each plastic type and degradation condition (Table 1).

The lifetime of the 3 mm solid PP insulation encasing the 12-inch rigid flowline is estimated between 200 and 400 years, while the lifetime of the 1.65 mm solid HDPE layer encasing the smaller 2-inch flowline is estimated between 75 and 150 years. The rate of mass loss resulting from microplastic formation over the lifetime of the structure is depicted in Fig. 1. The cumulative theoretical annual contribution of plastic mass from the solid casings of both flowlines is <15 g/m (<100 kg across their full length), assuming each flowline is fully exposed and degradation accelerants (e.g. laboratory pre-treatments) are not present.

Table 1. High-end specific surface degradation rates (k_d) reported by Chamas *et al.* (2020) for polypropylene and high-density polyethylene, employed in the degradation rate model.

Plastic	Shallow marine k_d (μm/year)	Shallow marine k_d with degradation accelerant (μm/year)
Polypropylene	7.5	15[A]
High-density polyethylene	11	22

[A]k_d reported by Chamas *et al.* (2020) for accelerated degradation of PP (4.6 μm/year) is lower than its non-accelerated counterpart, which is not expected. Accordingly, the k_d was conservatively assumed to be 15 μm/year based on the 2:1 ratio reported for HDPE degradation with/without laboratory pre-treatments.

Effects assessment

Species sensitivity distribution (SSD)

The present risk characterisation approach examines the consequences of microplastic exposure to the local ecology at the community level based on SSD of chronic toxicity data, in accordance with the current Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG 2018) ('the Guidelines'). A SSD is a model of the variation in sensitivity of species to a particular stressor (Posthuma *et al.* 2002). SSDs are derived by fitting a selected statistical distribution to toxicological endpoint data obtained from single-substance bioassays representing a range of taxa. Toxicological endpoints of concern are those that drive population persistence, growth, or decline, including growth suppression, decline in reproductive function, and/or mortality (Connors *et al.* 2017). The fitted distribution is used to infer a concentration that will be protective of a desired proportion of species in a hypothetical aquatic community. For non-bioaccumulative contaminants, the Guidelines recommend use of the 5% hazard concentration (HC_5; protective of 95% of species in an ecosystem) for protection of slightly-to-moderately disturbed ecosystems and the 1% hazard concentration (HC_1; 99% protection) for protection of ecosystems with high conservation value (Warne *et al.* 2018). The risk assessment utilises both the HC_5 and more conservative HC_1 to examine model outcomes for different categories of ecosystem condition.

Ecotoxicological data pertaining to microplastics

A total of 51 freshwater and marine animal ecotoxicity studies examining the toxicity (i.e. suppressed growth, decreased reproductive performance, and/or mortality) of microplastic exposure (e.g. ingestion) in the water column were identified through a detailed search of the peer-reviewed scientific literature. A variety of experimental designs have been used to evaluate the impacts of microplastics on freshwater and marine organisms. The most common test material is polystyrene, followed by PP. Studies frequently focus on single-size, spherical particles, with mixtures of irregular shapes (e.g. fibres and weathered fragments) of various sizes tested less frequently despite their prevalence in environmental samples (Phuong *et al.* 2016).

The ecotoxicity studies were screened for use in the SSD following procedures set forth in the Guidelines and based on environmental relevance. Accordingly, two SSDs were

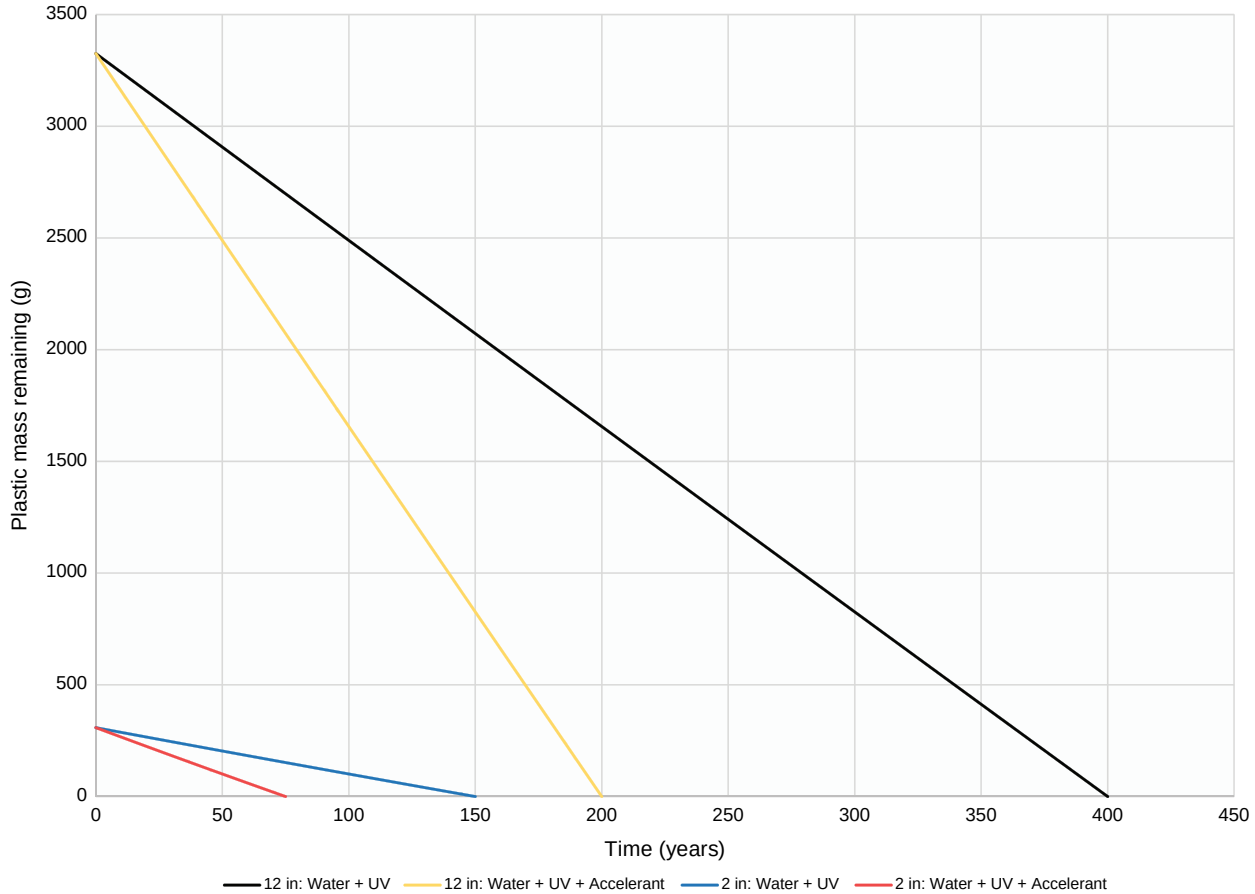

Fig. 1. Modelled mass loss along an exposed 1 m section of the 2-inch coiled tubing flowline and 12-inch rigid flowline based on shallow water degradation conditions (e.g. exposure to UV radiation) and also including degradation accelerants.

generated based on different data sets prioritising either particle shape or marine species, to examine the sensitivity of the PNEC (HC_5 and HC_1) outcome.

1. SSD #1 reflects the variation in species sensitivity to microplastic mixtures containing irregular shapes of various sizes. The SSD includes chronic EC10 (i.e. effect concentration at which 10% effect is observed) and no observed effect concentration (NOEC) data for 10 marine and freshwater species of various life stages (e.g. adult, juvenile, and larval) representing six taxonomic groups (Table 2). Combining marine and freshwater data was necessary to achieve the minimal data requirements and statistical power under the Guidelines; this is a common practice in risk assessment of microplastics (Burns and Boxall 2018; Besseling *et al.* 2019; Jung *et al.* 2021).
2. SSD #2 reflects the variation in marine species sensitivity to microplastics. The SSD includes chronic NOEC data for 15 marine species of various life stages representing eight taxonomic groups, based on studies employing either mixtures of irregular shapes of various sizes or single-size spherules (Table 3).

The data used in the SSDs are almost exclusively unbounded NOEC, meaning that a statistically significant difference between the test group exposed to the highest concentration of microplastics and the control group was not detected. Thus, true NOECs may be higher than the reported values and the HC_5/HC_1 values employed in the risk assessment may be lower than the true HC_5/HC_1.

Predicted no effect concentration (PNEC) for microplastics

The data in Table 2 and Table 3 were separately entered into Burrlioz 2.0 software (Barry and Henderson 2014), in accordance with the Guidelines. The Burrlioz software automatically selects the type of distribution that is fitted to the toxicity data based on data count. Toxicants that have toxicity data for ≥8 species that belong to at least four taxonomic groups are fitted to a three-parameter Burr Type-III distribution, for reasons described by Batley *et al.* (2018). The SSD outputs are shown in Fig. 2*a*, *b* (SSD #1) and Fig. 3*a*, *b* (SSD #2). The Burrlioz software computes a HC_5 of 0.066 mg/L and HC_1 of 0.032 mg/L for SSD #1, and a HC_5

Table 2. Ecotoxicological data inputs to SSD#1. Studies examine effects of exposure to irregularly-shaped microplastics of various sizes in marine or freshwater species.

Source	Test subject: Species, Phylum, Biome	Life stage	Polymer type	Particle shape/size	Concentrations tested	Exposure duration	Effect endpoint	NOEC or effect-level
Green (2016)	*Ostrea edulis* (Mollusca) Marine macroinvertebrate	Adults	PE	Irregular/0.48–316 µm	0.8, 80 µg/L	60 days Chronic	Growth	NOEC: 80 µg/L (0.08 mg/L)
Imhof *et al.* (2017)	*Daphnia magna* (Arthropoda) Freshwater macroinvertebrate	Adult	Two mixtures, assorted, acrylic	Irregular/<100, 29.5 µm central tendency	580 part./mL	21 days Chronic	Growth	NOEC: 8.41 mg/L[A]
Jung *et al.* (2021)	*Cyprindon variegatus* (Chordata) Marine fish	Adult	Mixed	Mixed/12–704 µm	5 mg/L	28 days Chronic	Mortality	NOEC: 5 mg/L
Karami *et al.* (2017)	*Danio rerio* (Chordata) Freshwater fish	Larvae	PE	Irregular/<17.6 µm	5, 50, 500 µg/L	20 days Chronic	Growth	NOEC: 500 µg/L (0.5 mg/L)
Qiao *et al.* (2019)	*Danio rerio* (Chordata) Freshwater fish	Adult	PS	Irregular/<250 µm	200 µg/L	21 days Chronic	Growth	NOEC: 200 µg/L (0.2 mg/L)
Reichert *et al.* (2019)	*Acropora muricata* (Cnidaria) Marine macroinvertebrate	Adult	PE	Irregular/65–410 µm	0.25 mg/L = 203 part./L	168 days Chronic	Growth	NOEC: 0.25 mg/L
Reichert *et al.* (2019)	*Pocillopora verrucosa* (Cnidaria) Marine macroinvertebrate	Adult	PE	Irregular/65–410 µm	0.25 mg/L = 203 part./L	168 days Chronic	Growth	NOEC: 0.25 mg/L
Reichert *et al.* (2019)	*Porites lutea* (Cnidaria) Marine macroinvertebrate	Adult	PE	Irregular/65–410 µm	0.25 mg/L = 203 part./L	168 days Chronic	Growth	NOEC: 0.25 mg/L
Yokota *et al.* (2017)	*Microcystis aeruginosa* (Cyanobacteria) Freshwater microorganism	–	Assorted	Irregular/<200 µm	66.7 mg/L	21 days Chronic	Growth	NOEC: 66.7 mg/L
Yokota (2017)	*Dolichospermum flosaquae* (Cyanobacteria) Freshwater microorganism	–	Assorted	Irregular/<200 µm	66.7 mg/L	21 days Chronic	Growth	NOEC: 66.7 mg/L
Ziajahromi *et al.* (2017)	*Ceriodaphnia dubia* (Arthropoda) Freshwater macroinvertebrate	Juvenile	Polyester	Fibres/100–400 µm	31.25–1000 µg/L	9 days Chronic	Reproductive performance	EC10 = 208 µg/L (0.208 mg/L)

[A]Based on spherical shape and central tendency diameter.

Table 3. Ecotoxicological data inputs to SSD#2. Studies examine effects of exposure to irregularly-shaped and spherical microplastics of various sizes in marine species.

Source	Test subject: Species, Phylum, Biome	Life stage	Polymer type	Particle shape/ size tested	Concentrations tested	Exposure duration	Effect endpoint	NOEC
Beiras *et al.* (2018)	*Oryzias melastigma* (Rotifera) Marine microinvertebrate	Adult	PE	Sphere/4–6 µm	0, 1, 10 mg/L	12 days Chronic	Mortality	10 mg/L
Beiras *et al.* (2018)	*Acartia clausi* (Arthropoda) Marine macroinvertebrate	Larvae	PE	Sphere/4–6 µm	0, 1, 3, 10, 30 mg/L	2 days Chronic	Mortality	30 mg/L
Cole and Galloway (2015)	*Crassostrea gigas* (Mollusca) Marine macroinvertebrate	Larvae	PS	Sphere/1 µm	1000 part./mL = 1.25×10^{-3} mg/L	8 days Chronic	Growth	1.25×10^{-3} mg/L
Cole and Galloway (2015)	*Crassostrea gigas* (Mollusca) Marine macroinvertebrate	Larvae	PS	Sphere/10 µm	1000 part./mL = 1.95 mg/L	8 days Chronic	Growth	1.95 mg/L
Davarpanah and Guilhermino (2015)	*Tetraselmis chuii* (Chlorophyta) Marine microalgae	–	PE	Sphere/1–5 µm	0.046–1.472 mg/L	4 days Chronic	Growth	1.472 mg/L
Gambardella *et al.* (2017)	*Artemia franciscana* (Arthropoda) Marine macroinvertebrate	Larvae	PS	Sphere/0.1 µm	0, 0.001, 0.01, 0.1, 1, 10 mg/L	2 days Chronic	Mortality	10 mg/L
Green (2016)	*Ostrea edulis* (Mollusca) Marine macroinvertebrate	Adults	PE	Irregular/ 0.48–316 µm	0.8, 80 µg/L	60 days Chronic	Growth	80 µg/L (0.08 mg/L)
Jung *et al.* (2021)	*Cyprindon variegatus* (Chordata) Adult fish	Adult	Mixed	Mixed/ 12–704 µm	5 mg/L	28 days Chronic	Mortality	5 mg/L
Lo and Chan (2018)	*Crepidula onyx* (Mollusca) Marine macroinvertebrate	Larvae	PP	Sphere/2–2.4 µm	6×10^4, 1.4×10^5 part./mL	95 days Chronic	Growth, mortality	0.7 mg/L[A]
Reichert *et al.* (2019)	*Acropora muricata* (Cnidaria) Marine macroinvertebrate	Adult	PE	Irregular/ 65–410 µm	0.25 mg/L = 203 part./L	168 days Chronic	Growth	0.25 mg/L
Reichert *et al.* (2019)	*Pocillopora verrucosa* (Cnidaria) Marine macroinvertebrate	Adult	PE	Irregular/ 65–410 µm	0.25 mg/L = 203 part./L	168 days Chronic	Growth	0.25 mg/L
Reichert *et al.* (2019)	*Porites lutea* (Cnidaria) Marine macroinvertebrate	Adult	PE	Irregular/ 65–410 µm	0.25 mg/L = 203 part./L	168 days Chronic	Growth	0.25 mg/L
Ribeiro *et al.* (2017)	*Scrobicularia plana* (Mollusca) Marine macroinvertebrate	Adult	PS	Sphere/18.4 µm	1 mg/L	14 days Chronic	Growth	1 mg/L

(*Continued on next page*)

Table 3. (*Continued*)

Source	Test subject: Species, Phylum, Biome	Life stage	Polymer type	Particle shape/ size tested	Concentrations tested	Exposure duration	Effect endpoint	NOEC
Seoane *et al.* (2019)	*Chaetoceros neogracile* (Ochrophyta)	–	PS	Sphere/0.5 µm	2.5 mg/L	3 days	Growth	2.5 mg/L
	Marine microalgae					Chronic		
Sjollema *et al.* (2016)	*Dunaliella tertiolecta* (Chlorophyta)	–	PS	Sphere/0.5 µm	25, 250 mg/L	3 days	Growth	250 mg/L
	Marine macroalgae					Chronic		
Sjollema *et al.* (2016)	*Dunaliella tertiolecta* (Chlorophyta)	–	PS	Sphere/6 µm	25, 250 mg/L	3 days	Growth	250 mg/L
	Marine macroalgae					Chronic		
Sjollema *et al.* (2016)	*Dunaliella tertiolecta* (Chlorophyta)	–	PS	Sphere/0.05 µm	25, 250 mg/L	3 days	Growth	25 mg/L
	Marine macroalgae					Chronic		
Wang *et al.* (2019)	*Artemia parthenogenetica* (Arthropoda)	Juvenile	PS	Sphere/10 µm	0.55–550 µg/L	14 days	Mortality	550 µg/L (0.550 mg/L)
	Marine macroinvertebrate					Chronic		

[A]Based on spherical shape and central tendency diameter.

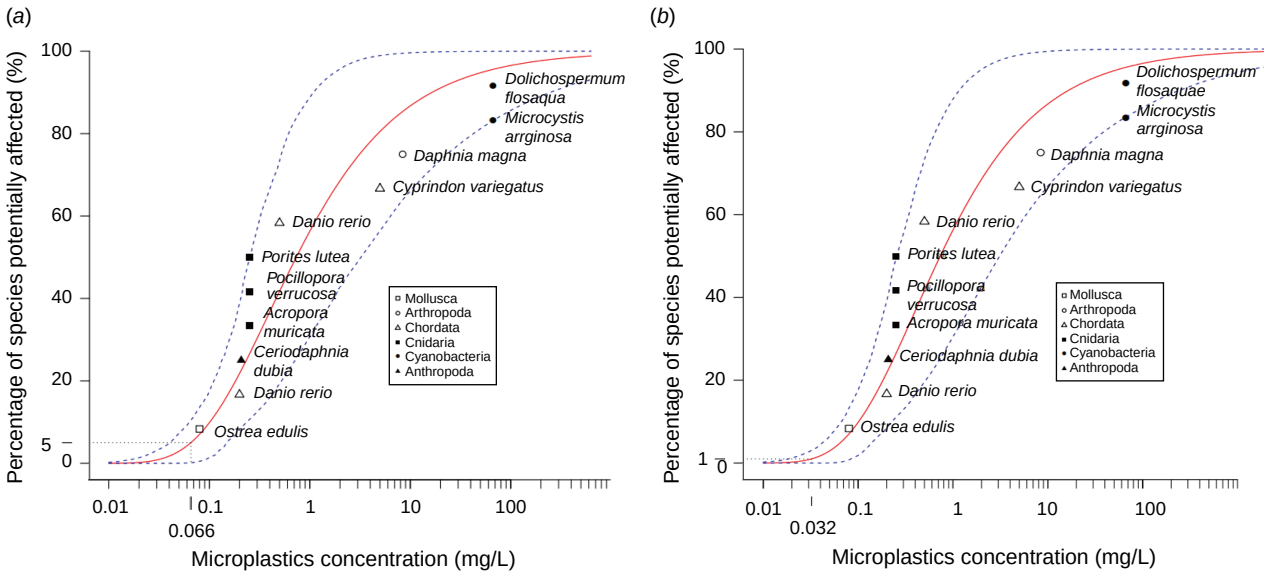

Fig. 2. SSD#1, which reflects the variation in marine and freshwater species sensitivity to microplastics mixtures containing irregular shapes of various sizes, predicting (*a*) 0.066 mg/L microplastics as protective of 95% of the marine community and (*b*) 0.032 mg/L microplastics as protective of 99% of the marine community.

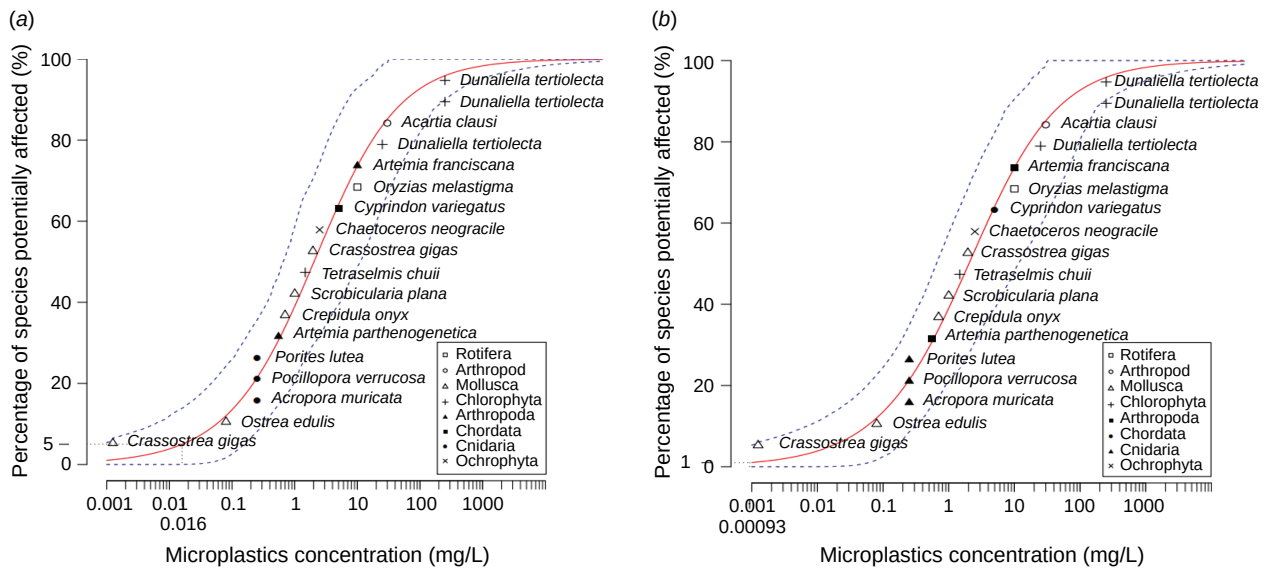

Fig. 3. SSD#2, which reflects the variation in marine species sensitivity to microplastics (irregular shapes and spherules, predicting (*a*) 0.016 mg/L microplastics as protective of 95% of the marine community and (*b*) 0.00093 mg/L microplastics as protective of 99% of the marine community.

of 0.016 mg/L and HC_1 of 0.00093 mg/L for SSD #2. The HC_5 of 0.016 mg/L and HC_1 of 0.00093 mg/L were conservatively selected as the PNEC for protection of slightly-to-moderately disturbed systems and systems of high conservation value, respectively.

Risk model iterations

The four iterations of risk characterisation are described herein. In Iterations #1–#3, microplastic mass formation is modelled over a 21-day period characteristic of a chronic exposure scenario for adult fish (Warne *et al.* 2018). Shorter durations are considered chronic exposure for other organisms and life stages; however, the longer chronic exposure period was selected as a conservative measure to allow for more plastic degradation, maximising theoretical exposure in the risk characterisation. The rate of plastic mass loss resulting from microplastic formation is assumed constant over the lifetime of the structure at the median degradation rate, mathematically derived as the quotient of the total plastic mass in the outer

casing and the time of complete degradation. Each successive iteration is representative of a more realistic exposure profile under ambient conditions and, collectively, provide an examination of model sensitivity to the exposure unit dimensions. In Iteration #4, complete degradation of the flowlines and microplastic exposure are assumed to occur over a very brief time frame, bounding modelled estimates of plastic degradation. Iterations #1–#4 are simulated several times commingling different estimations of microplastic contribution from the flowlines (based on different SSDRs representing shallow marine degradation conditions and also incorporating degradation accelerants) and PNECs (HC_5 and HC_1), as appropriate.

Each iteration of risk characterisation is performed for a 1 m segment of the flowlines irrespective of burial ($l = 1$). Since the extent of microplastic formation and the dimensions of the exposure unit are equally proportional to the length of the flowline, the PEC is considered constant along each 1 m length of flowline. Thus, the results of the risk characterisation are applicable along the full length of the flowlines. As such, the analysis is conservative should sections of the flowline be buried, which would further reduce degradation times and exposures.

Iteration #1

Iteration #1 provides an estimate of the dimensions of the exposure unit where microplastics must concentrate and remain for 21 days to result in concentration greater than the PNEC (HQ > 1) that may represent a potential hazard to marine organisms. A conceptual illustration is shown in Fig. 4.

This iteration conservatively assumes microplastic dispersion (e.g. advection by currents) is negligible and that marine organisms wholly reside in the space of the microplastic mass cluster during the time frame evaluated. Thus, the purpose of the first iteration is to determine if unrealistic hydrologic conditions and ecological behaviour (i.e. fauna home range and movement) would be necessary to pose a potential risk to the marine community.

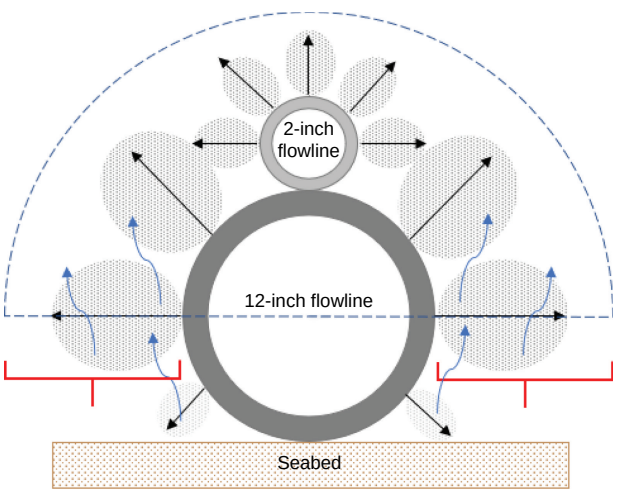

Fig. 4. Conceptual illustration of the exposure profile in Iteration #1 and Iteration #2, where the red bracket indicates the distance of microplastic dispersion from the plastic casing insulating the flowlines. In Iteration #1, this distance is calculated through back-calculation of the HQ. Iteration #2 assumes this distance is 30.5 m.

Iteration #1 involves back-calculation of the HQ based on a PEC that is 1.5 times the PNEC (HQ = 2, accounting for significant figures), with the final step of the derivation process as follows:

$$d = \left(\frac{2m_{21}}{\pi l \times 1.5 \times \text{PNEC}} + r_2^2\right)^{\frac{1}{2}};$$

where:

- d is the distance from the outer wall of the flowline (in m) (in the case of piggy-backing flowlines, the calculation is performed with respect to the larger flowline);
- r_2 is the radius (in m) of the outer wall of the flowline;
- m_{21} is the cumulative microplastic mass contribution across l m length of both flowlines over 21-day period of degradation (in µg); and
- PNEC is the predicted no-effect concentration (i.e. HC_5 or HC_1) (in µg/m^3).

Iteration #2

Iteration #2 provides an evaluation of the potential for ecological impairment for a localised exposure scenario where newly formed microplastics are assumed to concentrate and remain within an arbitrary distance of 100 ft (30.5 m) of the flowlines. This second iteration is considered to be more realistic than the first iteration as it provides more realistic representation of microplastic dispersion, although it can be expected to be conservative in that it is highly likely that microplastic material that separates from the surface of the flowlines would be dispersed more widely.

Iteration #2 involves forward calculation of the HQ as follows:

$$\text{HQ} = \left(\frac{\text{PEC}}{\text{PNEC}}\right) = \left(\frac{2m_{21}}{\pi l\,(30.5^2 - r_2^2)}\right)\left(\frac{1}{\text{PNEC}}\right)$$

Iteration #3

Iteration #3 provides an evaluation of the potential for adverse ecological effects where the dimensions of the exposure unit are estimated via a three-dimensional 'box model' of microplastic fate and transport (conceptually illustrated in Fig. 5). The box model defines the spatial extent of newly formed microplastic mass based on advective flux due to transfer by water current velocity (X, Y) and density-driven rise velocity (Z). The box model assumes that (1) the vector of ocean current velocity and orientation of plastic structures are coplanar on the XY plane, thus the current does not contribute to the vertical migration of the microplastics and

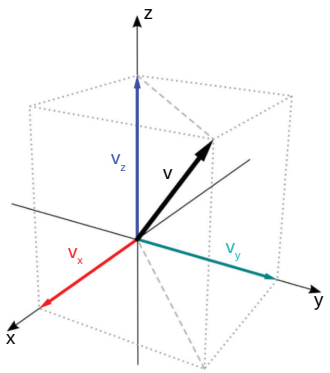

Fig. 5. Conceptual illustration of the 'box model' used to define the region of microplastic dispersion and exposure unit in Iteration #3 and Iteration #4. V_x, V_y, and V_z are the velocity vectors for microplastics separated from the flowlines in the X, Y, and Z plane, respectively, defined by the advective flux driven by ocean current and density-driven rise.

(2) the vector of ocean current velocity and course of the flowlines are oriented at a 45° angle. This is considered to be the most realistic scenario for organisms that may be exposed to microplastics that migrate from the plastic infrastructure.

In Iteration #3, microplastic mass formation over a 21-day period is condensed into a 1-h window during which the microplastics are carried away from the flowlines and interact with the local ecology (i.e. the box model is based on 1 h of dispersion). The 1-h exposure period represents chronic exposure for microinvertebrate gametes and macroalgae during early life stages and is the shortest time period representative of chronic exposure among aquatic ecological assessment endpoints (Warne *et al.* 2018).

Iteration #3 is expected to overestimate microplastic mass formation during the 1-h exposure period and assumes this extent of exposure is sufficient to cause adverse effects at the community level. Thus, this iteration results in amplified estimates of risk.

The calculation for Iteration #3 is as follows:

$$\mathrm{HQ} = \left(\frac{\mathrm{PEC}}{\mathrm{PNEC}}\right) = \left(\frac{m_{21}}{d_x \times d_y \times d_z}\right)\left(\frac{1}{\mathrm{PNEC}}\right);$$

$$d_x = |v \cos 45^\circ| t,\ d_y = |v \sin 45^\circ| t,\ \text{and}\ d_z = r_v t;$$

where:

- d_x, d_y, and d_z are the distance of microplastic dispersion in the X, Y, and Z directions (m);
- v is the velocity of the ocean current (m/h);
- r_v is the density-driven rise velocity of the microplastics (m/h); and
- t is the duration of microplastic dispersion (1 h).

Iteration #4

Iteration #4 provides an overly conservative evaluation of the potential for adverse ecological effects under a worst-case exposure scenario, in which the full mass of plastic in the outer layer of the flowlines is assumed to concentrate, immediately, as microplastics in the exposure unit defined by the box model in Iteration #3. The calculation for Iteration #4 is the same as for Iteration #3, except m_{21} is replaced by the totality of the plastic mass in the solid layer insulating the flowlines.

Results

Overview

A summary of the degradation modelling and risk assessment results are provided in Table 4 and Table 5, respectively. The risk assessment results are reviewed herein.

Iteration #1

The computation predictably produced arbitrarily small measurements of distance from the flowline wall (from 4 to 24 m), meaning the cumulative mass of microplastics displaced from the flowlines are limited to an extremely small region of the overall field, and area use by marine fauna is restricted to the same region. The implication that microplastic distribution is restricted in this locality and thus may present a risk to the marine community is unrealistic based on natural ambient conditions and behaviour of marine fauna.

1. Plastic degradation is extremely slow under deep marine conditions (and likely to be significantly less than modelled), and microplastics are far more likely to be advected

Table 4. Degradation modelling.

Component	Plastic	Shallow marine environment		Shallow marine environment with degradation accelerant	
		t_d (years)	m_{21} (µg)	t_d (years)	m_{21} (µg)
12 inch rigid flowline	PP	400	478 296	200	956 592
2 inch coiled tubing	HDPE	150	118 275	75	236 550
Both flowlines	–	–	596 571	–	1 193 143

t_d, time for complete degradation; m_{21}, mass loss/21 days, no burial, l = 1 m.

Table 5. Risk assessment results.

(a) Risk characterisation Iteration #1

	Shallow marine environment				Shallow marine environment with degradation accelerant			
Component	PNEC (µg/L)	Volume around 1 m exposed section (L)	Volume around 1 m exposed section (m^3)	Distance from flowline (m)	PNEC (µg/L)	Volume around 1 m exposed section (L)	Volume around 1 m exposed section (m^3)	Distance from flowline (m)
Both flowlines	16.0	24 857	24.9	4.0	16.0	49 714	49.7	5.6
Both flowlines	0.93	427 650	427.6	16.5	0.93	855 299	855.3	23.3

(b) Risk characterisation Iteration #2

	Volume around 1 m exposed section		Shallow marine environment			Shallow marine environment with degradation accelerant		
Component	(m^3)	(L)	Concentration (µg/L)	HQ PNEC = 16 µg/L	HQ PNEC = 0.93 µg/L	Concentration (µg/L)	HQ PNEC = 16 µg/L	HQ PNEC = 0.93 µg/L
Both flowlines	1460.4	1 460 434	0.41	3×10^{-2}	4×10^{-1}	0.82	5×10^{-2}	9×10^{-1}

(c) Risk characterisation Iteration #3

	Volume of dispersion from exposed 1 m section		Shallow marine environment			Shallow marine environment with degradation accelerant		
Component	(m^3)	(L)	Concentration (µg/L)	HQ PNEC = 16 µg/L	HQ PNEC = 0.93 µg/L	Concentration (µg/L)	HQ PNEC = 16 µg/L	HQ PNEC = 0.93 µg/L
Both flowlines – 'Low Range' rise velocity	4.67×10^6	4.67×10^9	1.3×10^{-4}	8×10^{-6}	1×10^{-4}	2.6×10^{-4}	2×10^{-5}	3×10^{-4}
Both flowlines – 'High Range' rise velocity	1.77×10^7	1.77×10^{10}	3.4×10^{-5}	2×10^{-6}	4×10^{-5}	6.7×10^{-5}	4×10^{-6}	7×10^{-5}

(d) Risk characterisation Iteration #4

	Volume of dispersion from exposed 1 m section		All plastic mass in exposed 1 m section		
Component	(m^3)	(L)	Concentration (µg/L)	HQ PNEC = 16 µg/L	HQ PNEC = 0.93 µg/L
Both flowlines – 'Low Range' rise velocity	4.67×10^6	4.67×10^9	7.8×10^{-1}	5×10^{-2}	8×10^{-1}
Both flowlines – 'High Range' rise velocity	1.77×10^7	1.77×10^{10}	2.0×10^{-1}	1×10^{-2}	2×10^{-1}

and diluted by the known water currents in the vicinity of the flowlines and density-driven rise through the water column. Thus, concentrations in the vicinity of the flowlines are expected to be lower than those that are associated with chronic (0.016 and 0.00093 mg/L) or acute (≫0.016 and ≫0.00093 mg/L) effects resulting from long-term and short-term exposure, respectively.

2. Animals move about their home range in pursuit of resources, shelter, and reproduction, and experience exposure in various portions of their home range based on time spent and behaviour engaged in each area. The home range size of marine fish and mammals, including threatened and endangered species, far exceeds the dimensions of the exposure unit characterised by Iteration #1, sharply decreasing the probability that these receptors will be exposed to a localised concentrated condition emanating from the flowlines (an unrealistic scenario). Additionally, the available evidence in the peer-reviewed scientific literature indicates that microplastics do not appreciably accumulate within marine biota, nor biomagnify through the food web and, thus, the unrealistic condition modelled by Iteration #1 would not be expected to propagate throughout the marine community by way of the food chain.

Accordingly, Iteration #1 concludes that unrealistic hydrologic conditions and ecological behaviour would be necessary for microplastic formation from the flowlines to pose a potential risk to the local marine community.

Iteration #2

Iteration #2 calculates the PEC in the order of 10^{-2} to <1 times the PNEC (HQs < 1) should dispersion of the degraded plastic mass be localised to within 30.5 m of the flowlines. Thus, Iteration #2 concludes microplastic formation from the flowlines does not pose a risk to the local marine community.

Iteration #3

Unpublished technical documents from the field indicate that the ocean current velocity above the seabed is approximately 0.2 m/s. Microplastic rise velocities reported in the scientific literature vary for different shapes and sizes with values ranging from 0.005 to 0.019 m/s (Kukulka *et al.* 2012; Kooi *et al.* 2016). Based on these inputs, microplastics are modelled to concentrate within a cuboid-shaped region with dimensions of 378 m (X) × 613 m (Y) × 18–68 m (Z) above the seabed (Table 6). Iteration #3 is simulated a total of eight times, commingling different PNECs, SSDRs, and exposure unit dimensions based on the low/high rise velocity.

Iteration #3 calculates the PEC in the order of 10^{-6} to 10^{-4} times the PNEC (HQs ≪ 1) when the cumulative degraded mass from both flowlines is dispersed throughout the region defined by the box model. Thus, Iteration #3 concludes microplastic formation from the flowlines does not pose a risk to the local marine community.

Table 6. Box model of microplastic dispersion from the flowlines (1 h).

Ocean current velocity	V_x [A] (m/s above seabed)	V_y [B] (m/s above seabed)	V_z (m/s above seabed)
Low-end estimate	0.141	0.141	0.005
High-end estimate	0.141	0.141	0.019
Microplastic dispersion (1 h)	**d_x (m)**	**d_y (m)**	**d_z (m)**
Low-end estimate	509	509	18.0
High-end estimate	509	509	68.4

[A] V_x = | (0.2 m/s) cos(45°) |. [B] V_y = | (0.2 m/s) sin(45°) |.

Iteration #4

Iteration #4 indicates that the PEC is in the order of 10^{-2} to <1 times the PNEC (HQs < 1) when the totality of the mass from both flowlines is dispersed through the region defined by the box model in Iteration #3. Thus, the fourth iteration of risk characterisation concludes that microplastic formation from the flowlines does not pose a risk to the local marine community.

Conclusion

This approach developed for quantitative assessment of the risks associated with long-term degradation of deep-sea plastic – containing infrastructure involves four iterations of modelled 'risk' based on forward or backward calculation of a deterministic HQ, using an SSD derived in accordance with Australian water quality guidelines. The multiple iterations provide for an examination of model sensitivity to the various model inputs describing the PEC. The sensitivity of the PNEC was assessed through derivation of two SSDs from different data sets, prioritising either particle shape or marine species. Thus, the approach serves to bound the risk characterisation, encompassing all possible outcomes. In developing the model estimates, conservative assumptions have been made that can be expected to significantly overestimate the level of risk, reflecting the desire to protect sensitive populations.

In the case study of two plastic-containing flowlines in an oil production field offshore of Australia, the outcome of the risk assessment is the same for each model iteration: degradation of the flowlines does not pose a risk to the local marine community.

Supplementary material

Supplementary material is available online.

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Data availability. The site-specific data used in the risk characterisation model are not publicly available at this time.

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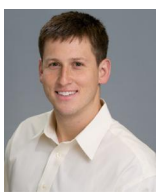

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