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7 **Title:**

8 A catchment-scale perspective of plastic pollution

9 **Running Head:**

10 Catchment-scale plastic pollution

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30 **Abstract**

31 Plastic pollution is distributed widely across the globe, but compared with marine
32 environments, there is only rudimentary understanding of the distribution and effects of plastics
33 in other ecosystems. Here, we review the transport and effects of plastics across terrestrial,
34 freshwater and marine environments. We focus on hydrological catchments as well-defined
35 landscape units that provide an integrating scale at which plastic pollution can be investigated.
36 Diverse processes are responsible for the observed ubiquity of plastic pollution, but sources,
37 sinks and fluxes in river catchments are poorly quantified. Nevertheless, early indications are
38 that rivers are hotspots of plastic pollution, supporting some of the highest recorded
39 concentrations. River systems are also likely pivotal conduits for plastic transport among the
40 terrestrial, floodplain, riparian, benthic and transitional ecosystems with which they connect.
41 Although ecological effects of micro- and nano-plastics plastics might arise from a variety of
42 physical and chemical mechanisms, understanding of their nature, severity and scale is
43 restricted and lacks consensus in comparison to macro-plastic research. Furthermore, whilst
44 individual-level effects are often graphically represented in public media, knowledge of the
45 extent and severity of the impacts of plastic at population, community and ecosystem levels is
46 limited. Given the potential social, ecological and economic consequences, we call for more
47 comprehensive investigations of plastic pollution in ecosystems to guide effective management
48 action and risk assessment. This is reliant on (i) expanding research to quantify sources, sinks,

49 fluxes and fates of plastics; (ii) improving environmentally relevant dose-response
50 relationships for different organisms and effect pathways, (iii) scaling up from studies on
51 individual organisms to populations and ecosystems, where individual effects are shown to
52 cause harm; and (iv) improving biomonitoring through developing ecologically relevant
53 metrics based on contemporary plastic research.

54 **1. Introduction**

55 Plastic waste production across the globe has reached approximately 6300 million metric tons
56 (MT), most (79%) of which has been disposed of to land-fills and more widely into the
57 surrounding environment (Geyer et al., 2017). The annual flow of plastic pollution to the
58 world's oceans is estimated to be 4.8–12.7 MT, a large proportion of which comes from sources
59 on land and is transported by rivers or wind (Jambeck et al., 2015). Plastic pollution is
60 comprised of a variety of different organic polymers and is invariably categorised based on
61 particle size: nano (<1 μm), micro (0.01–5 mm), meso (5–25 mm) and macro (>25 mm). Once
62 *in situ* within ecosystems, degradation and fragmentation processes make the identification and
63 removal of these plastic particles difficult. Recent reviews and theoretical models have,
64 however, indicated a large number of potential sources, fluxes and sinks of plastics across the
65 wider environment (Alimi et al., 2018; Browne et al., 2011; de Souza Machado et al., 2018a;
66 Horton et al. 2017a; Wagner et al., 2014). A more detailed understanding of the sources, fluxes
67 and effects of these anthropogenic pollutants, and a more comprehensive quantification of their
68 fate, is now required urgently to determine the risks to people and ecosystems across the globe
69 (de Souza Machado et al., 2018a; Horton & Dixon, 2017; Nizzetto et al., 2016a).

70 Large production volumes, long-term environmental persistence and potential ecological
71 effects have meant that plastic pollution has received increasing attention (Thompson et al.,

72 2009). The variety of plastic sizes (microns to metres) and characteristics (e.g. shape, physical
73 and chemical properties) make this group of pollutants particularly diverse (Rochman, 2015).
74 In turn, the diversity and ubiquity of plastic particles within natural systems, mean there is a
75 wide variety of ways in organisms can interact with, become entangled in, or ingest plastic
76 pollution (e.g. Cole *et al.*, 2013; Foekema *et al.*, 2013; Lusher *et al.*, 2013, 2015a; Hall *et al.*,
77 2015). Although existing information indicates the potential for effects across biological
78 communities and human populations (Halden, 2010), our understanding of the effects of plastic
79 pollution on people and ecosystems remains constrained. Furthermore, despite widely
80 identified interactions between organisms and plastics, a comprehensive mechanistic
81 understanding of effect pathways remains limited, with a few notable exceptions (e.g. ingestion
82 and energy reserve depletion: Wright *et al.*, 2013a). Further to this, existing dose-response
83 relationships for effect pathways are relative restricted and are often limited in either their
84 taxonomic breadth or utility (e.g. unrealistic concentrations and/or plastic characteristics:
85 Phuong *et al.*, 2016). Notable exceptions are presented by recent studies, where existing
86 predicted no effect concentrations for microplastics have been collated – covering a number of
87 plastic types and size categories, as well as incorporating a range of aquatic organisms (Burns
88 & Boxall, 2018; Everaert *et al.*, 2018).

89 In this review, we critically evaluate existing evidence for the fluxes and effects of plastic
90 pollution from a catchment-scale perspective. We focus particularly on freshwater systems as
91 highly connected networks through which plastics are transported from sources in terrestrial
92 environments to marine ecosystems. Throughout the manuscript we aim to: (i) synthesise
93 existing knowledge regarding the fluxes and effects of plastic pollution across hydrological
94 catchments; (ii) highlight emerging areas that require further research; and (iii) identify
95 improvements to aid the development and integration of catchment-scale research.

96 **2. Fluxes of plastics through hydrological catchments**

97 Hydrologically defined river catchments offer valuable units in which to consider the sources,
98 fluxes and fates of plastic pollution (Fig 1). This is because the transport of plastics often
99 follows hydrologically pathways, and hydrological pathways are determined clearly by
100 topography, surface morphology and drainage patterns (Bracken et al., 2013).

101 Once released into the environment, plastics reach across all ecosystems and ecotypes across
102 the globe (Geyer et al., 2017). Plastic particles are widespread, even in areas considered to have
103 little to no human influence, such as the deep sea, Arctic sea ice and remote uninhabited islands
104 (Lavers & Bond, 2017; Peeken et al., 2018; Van Cauwenberghe et al., 2013). Along their
105 movement from source to sink, plastics interact with their physical, chemical and biological
106 environment in ways that depend on the characteristics of the plastic (size, shape, polymer type,
107 etc.) so that it is not practical to consider ‘plastics’ as a singular form of pollution. Nevertheless,
108 for the purposes of this discussion, we highlight existing theoretical and empirical evaluations
109 of the flux and effects of a broad group of ‘plastics’ across ecosystems.

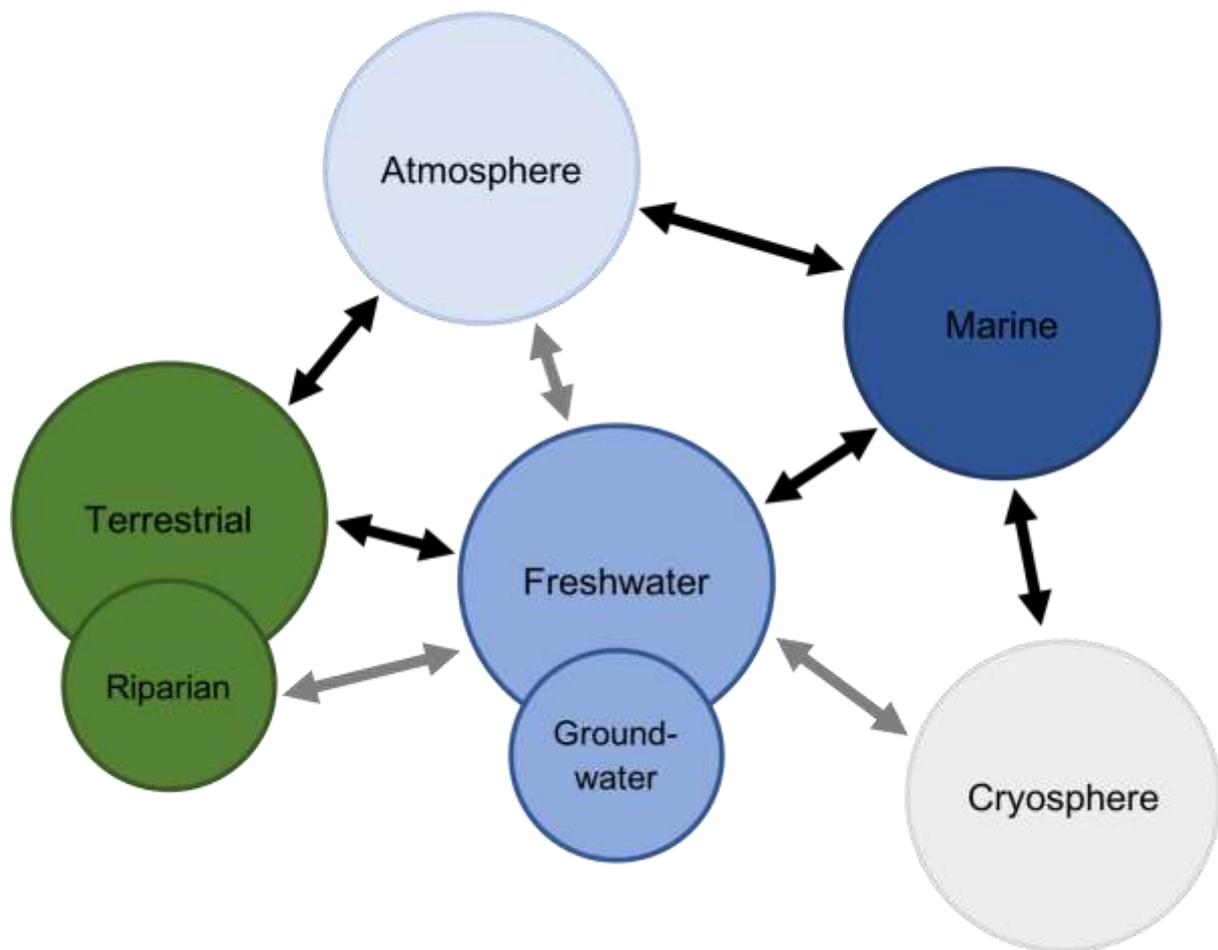


Fig. 1. Conceptual diagram of plastic fluxes across the compartments of hydrological catchments. Specific pathways, indicated by black arrows, are further discussed within the main body of text. Grey arrows represent theoretical fluxes that have yet to be investigated in detail (see *Underrepresented ecosystems*).

110 The movement of plastic across and between compartments of river catchments is analogous
 111 to other catchment-scale processes involving fluxes, transformations and storage (Horton &
 112 Dixon, 2017). It has been theoretically suggested that microplastic particles behave in a similar
 113 manner to other particulate matter with similar characteristics (e.g. density, size and shape),
 114 such that movement of these particles resembles the flux of others (e.g. sediment/soil particles,
 115 fine and coarse organic matter (Nizzetto et al., 2016a). In reality, however, it is likely that the
 116 unique diversity of shape, density, size, or surface complexity of plastic particles, limits the
 117 accuracy and utility of existing models to predict plastic movement across and within
 118 ecosystems. Furthermore, the behaviour of larger particles of plastic (meso to macro) within

119 ecosystems remains poorly understood. The processes responsible for transporting these larger
120 particles are likely similar to those transporting microplastics, yet operate at larger scales,
121 involve more energy and occur less frequently. As a result of these complications, there
122 remains insufficient data to accurately parameterise and validate empirical transport models
123 for plastic pollution.

124 While movement of plastic between atmospheric, terrestrial and freshwater systems appears to
125 multidirectional, marine systems are generally perceived to act as sinks for plastics, with
126 limited outfluxes (Browne et al., 2011). However, a significant amount of plastic is transported
127 through river catchments (Lebreton et al., 2017). While this is likely to be the main source of
128 marine plastics (Nizzetto et al., 2016a), little is known about the residence time of plastics in
129 freshwaters, which could also trap significant amounts of material. Quantification of all the
130 pathways from land to sea remains limited (but see Clark *et al.*, 2016; Galloway *et al.*, 2017),
131 yet is key to supporting the estimation of ecological risk across systems.

132 The characteristics of hydrological catchments are like to maintain important implications for
133 the flux of plastic pollution across the landscape. Features such as topography, hydrology and
134 land use, are likely responsible for altering the mass balance of plastics within catchments –
135 influencing both the diversity and volumes of plastic emitted from sources, the nature and
136 magnitude of transport processes, as well as the likelihood of temporary storage across
137 ecosystems within the wider hydrological catchment. Limited information exists at the
138 catchment-scale, however, existing studies investigating plastic pollution across terrestrial,
139 freshwater, atmospheric and marine systems provide a basis for understanding catchment-scale
140 transport of plastic pollution.

141 ***2.1. Terrestrial systems***

142 Several sources of plastic pollution are associated with human activities across the terrestrial
143 environments present within hydrological catchments (de Souza Machado et al., 2018a; Hurley

144 & Nizzetto, 2018). Plastic pollution stems from a wide array of activities, creating a patchwork
145 of point and diffuse sources across catchments, with both rural and urban soils are considered
146 to be contaminated by plastic particles (Nizzetto et al., 2016b). Intensive agricultural practices
147 distribute plastics across rural regions through the degradation of machinery, diffuse littering,
148 application of sewage sludge as a fertiliser (Zubris & Richards, 2005) and plastic mulching
149 (Steinmetz et al., 2016). The redistribution of sewage sludge is particularly interesting,
150 transporting plastics associated within urban activities across some rural landscapes (Horton et
151 al., 2017a; Zubris & Richards, 2005). The flux of plastics from this activity is potentially
152 important considering that 80–90% of plastics entering sewage treatment are stored in sludge
153 (Talvitie et al., 2017), and a large amount of MPs (4196–15385 MP kg⁻¹ dry mass) remain
154 post-treatment of biosolids (Mahon et al., 2017). Within Europe, Nizzetto *et al.* (2016b)
155 estimated that 125–180 t of microplastics per million inhabitants are added to agricultural soils
156 as a result of sewage sludge application. Urban land use and associated activities also provide
157 several different sources of plastic pollution (Ballent et al., 2016; Nizzetto et al., 2016b). In
158 particular, loss during waste disposal, industrial spillage and release from landfills provide
159 significant inputs of plastic (Lechner & Ramler, 2015; Sadri & Thompson, 2014). The large
160 production of plastics in terrestrial systems, limited land area and range of distribution
161 processes may result in a greater environmental concentration within these ecosystems,
162 compared to marine environments (Horton et al., 2017a).

163 The flux and storage of plastic within terrestrial systems have been catalogued theoretically,
164 but there are few field data. Once in terrestrial ecosystems, plastics accumulate in soils and can
165 be ingested by soil-dwelling organisms (Rillig, 2012; Rillig et al. 2017a). Existing empirical
166 data indicate that plastics are incorporated into earthworm casts (Huerta Lwanga et al., 2017),
167 and also that polyethylene microbeads (0.71–2.8 mm) reach down into the subsurface through
168 earthworm burrows (Rillig et al., 2017b). Concentration of plastic in soils varies: river

169 floodplains across Switzerland revealed relatively low concentrations of microplastics (0–
170 55.5 mg kg⁻¹, Scheurer & Bigalke, 2018), but more heavily contaminated industrial soils (300–
171 67500 mg kg⁻¹) have been observed from samples collected in Australia (Fuller & Gautam,
172 2016). The lightweight nature of plastic material, means that in terrestrial systems, particles are
173 more easily transported by wind and weather events (Zylstra, 2013), diffusing their distribution
174 across catchments.

175 Plastics stored in terrestrial systems may subsequently be re-mobilised and subsequently
176 transported within or across catchments (Dris et al., 2015a; Duis & Coors, 2016; Wagner et al.,
177 2014). Although empirical assessments are absent from the literature, soil erosion during heavy
178 rainfall is likely to increase the flux of plastic particles from soils to river systems (Bläsing &
179 Amelung, 2018). In particular, landfills in low lying areas prone to flooding, present a
180 significant source of plastics into freshwater ecosystems (Brand et al 2018). In some cases, as
181 during flood events, plastics may even return to land, however the flow of plastics out of
182 terrestrial systems appears dominant and drives the global plastic cycle (see de Souza Machado
183 *et al.*, 2018a).

184 **2.2. Atmospheric systems**

185 Plastic, as a result of its lightweight characteristics, can be suspended and transported within
186 the atmosphere at both the catchment and regional scale (Dris et al., 2016; Prata, 2018). Plastics
187 enter the atmospheric system through a variety of pathways across catchments, including
188 combustion of waste plastic, wind erosion of various media, urban dust (including tyre wear
189 particles, paint particles and synthetic fibres) (Lee et al., 2016; Unice et al. 2012) and diffuse
190 litter (Dris et al., 2016). The majority of plastic observed in atmospheric systems falls into the
191 micro- and nano- size classes, nevertheless, larger particles may be suspended in the
192 atmosphere if they support a suitable set of characteristics (e.g. disposable plastic bags and
193 balloons). Significant concentrations of plastic are observed within the lower atmosphere (0.3–

194 1.5 MPs m⁻³), yet compared to indoor air these values are relatively low (1–60 MPs m⁻³) (Dris
195 et al., 2017). Polyurethane, polypropylene and polystyrene microplastic particles were
196 identified in atmospheric fallout, at concentrations between 175 to 313 MP m⁻² day⁻¹ in
197 Dongguan city (Cai et al., 2017). Similar concentrations of microplastic were also observed
198 using passive samplers in Paris; 2–355 MPs m⁻² day⁻¹ (Dris et al., 2016). The fallout of these
199 particles is, in turn, responsible for the accumulation of particles in ‘street dust’. For example,
200 ‘street dust’ collected from sites across Tehran exhibited 88–605 microplastics per 30 g of dust
201 (Dehghani et al. 2017). The atmosphere therefore appears to store and transport plastic, and
202 while there is limited evidence of long-range atmospheric flows of plastic, microplastic
203 pollution occurs in remote environments such as alpine lakes (Free et al., 2014). The storage
204 and transportation of plastics in the atmosphere is likely temporally variable; influenced by the
205 prevailing meteorological conditions at different time scales. Thus, it is unlikely that the
206 atmosphere provides a long-term store of plastics, instead acting as a temporary store, as well
207 as a potential short- and long-distance transport pathway.

208 **2.3. Freshwater systems**

209 Freshwater ecosystems include a diverse assemblage of running, standing, surface and
210 underground waterbodies. Running waters act as conduits connecting terrestrial and marine
211 systems, providing an important long-range transport mechanism, as well as storage
212 opportunities in some benthic or riparian habitats (Horton & Dixon, 2017). Standing waters,
213 including lakes and ponds, may also act as accumulators and stores of plastic (Vaughan et al.,
214 2017). The role of freshwaters in the transport of plastics across catchments is thus highly
215 dependent upon the characteristics of the waterbody.

216 The sources of plastic entering freshwater ecosystems are varied and spatially heterogeneous,
217 ranging from diffuse inputs stemming from run-off to point sources such as Wastewater
218 Treatment Works (WwTWs) and Combined Sewer Overflows (CSOs) (Horton et al., 2017a).

219 Domestic sewage collects a variety of plastic types, including synthetic wet wipes, microbeads
220 (Duis & Coors, 2016) and polymer fibres from the laundering of synthetic textiles (Napper &
221 Thompson, 2016). WwTWs effectively remove the vast majority of both large and small
222 plastics from raw influent (95–99%), yet these point sources remain an important contributor
223 of smaller microplastic particles to freshwater ecosystems (Murphy et al. 2016; Talvitie et al.,
224 2017). These contributions from treated effluent, however, are spatially variable in response to
225 variable removal efficiencies across WwTWs (Siegfried et al., 2017). Microplastics removed
226 during treatment are also not completely disconnected from entering the environment, with the
227 retention of plastics in sludge (Mahon et al., 2017) and the potential for subsequent re-
228 application across catchments. Further sources of micro- and macro-plastic identified within
229 existing literature include, diffuse urban pollution, stormwater drains (Horton et al., 2017b),
230 combined sewage overflows and litter (Horton et al. 2017a). The combined effects of urban
231 pollution sources have been shown to generate enhanced concentrations of plastics within
232 freshwater systems, for example the highly populated Lake Erie maintains far greater
233 concentrations of microplastic particles (43,000 MP km⁻²) in comparison to lakes in proximity
234 to less populated regions, e.g. Lake Huron (6,541 MP km⁻²) and Lake Superior (12,645
235 MP km⁻²) (Eriksen et al., 2013). As a result of the ubiquity of point and diffuse sources of
236 plastic pollution within freshwaters, it is not surprising that plastic has been widely identified
237 within a range of freshwater habitats (Free et al., 2014; Horton et al., 2017b). Data from
238 freshwater systems, thus far, indicate that these systems are important hotspots of plastic
239 pollution, holding some of the highest concentrations of (micro)plastics recorded in either
240 water and sediments across the globe (Hurley et al., 2018; Mani et al., 2015).

241 River systems act as conduits, connecting terrestrial, riparian, floodplain and transitional
242 ecosystems within their catchments. Theoretical and modelling assessments support the
243 notions of particle transfer across habitats, but also under certain conditions significant storage

244 (see Nizzetto *et al.*, 2016a). The retention and transport of plastics are a product of particle
245 characteristics (density and dimensions) and environmental characteristics (flow regime)
246 (Nizzetto *et al.*, 2016a). Within river systems plastics may pool in benthic sediments
247 (Castañeda *et al.*, 2014) or be transferred along an altitudinal gradient towards marine
248 ecosystems (Lebreton *et al.*, 2017; Mani *et al.*, 2015). This transport may occur throughout the
249 water column, with significant transport observed both on the surface (Dris *et al.*, 2015b; Aaron
250 Lechner *et al.*, 2014) and subsurface (Morritt, Stefanoudis, Pearce, Crimmen, & Clark, 2014)
251 of river systems.

252 The interaction between storage and flux processes is highlighted in a recent study by Hurley
253 *et al.* (2018), which indicates the significant mobilisation and removal of sedimentary
254 microplastics in response to high flow events. In this example, 0.85 ± 0.27 tonnes of plastic
255 was removed from a single catchment during an individual flood event (Hurley *et al.*, 2018).
256 Similar flood events may also be responsible for distributing plastics onto floodplains. The net
257 or total flux of plastics from terrestrial sources, through hydrological networks to marine
258 systems however remains poorly understood. It is, however, estimated that global river
259 networks are responsible for transferring 1.15–2.41 MT of plastic pollution to marine
260 environments (Lebreton *et al.*, 2017). This estimate, however, is based solely upon surface
261 transport and does not account for suspended and bedload transport. As a result, the mass of
262 plastic transported through river systems are likely to be underestimated, with the combination
263 of surface and subsurface transport more likely accounting for a greater proportion of the total
264 4.8–12.7 MT estimated entering marine environments per year (Jambeck *et al.*, 2015).

265 **2.4. Marine systems**

266 Oceans are often considered the end-point of plastic fluxes from hydrological catchments
267 (Horton & Dixon, 2017). As highlighted previously, it is estimated that fluxes of plastics from
268 rivers provide a major input of macro- and micro-plastics into marine environments across the

269 globe (Lebreton et al., 2017; UNEP, 2016). With 50% of the global population residing within
270 31 km of the coast (Small & Cohen, 2004), direct inputs of plastics are also likely to be
271 significant. Finally, industrial activity, such as commercial fishing, contributes to the total
272 plastic burden within marine ecosystems (Lusher et al., 2015b). In most cases these activities
273 release macro-plastics, such as netting and plastic sheeting, which then degrades to form
274 microplastic particles when exposed to physical, chemical or biological processes (e.g.
275 Davidson, 2012). The potential variety of plastic sources generates a widespread distribution
276 of plastics in the marine environment, yet heterogeneity exists with accumulation zones and
277 plastic hotspots (Lusher, 2015). Plastic transport processes are widespread and heterogeneous
278 within the marine environment (Browne et al., 2011). Ocean and wind circulation currents,
279 ranging from small-scale vertical mixing to large-scale oceanic gyres, appear responsible for
280 the observed patchiness of plastic distribution within marine systems (Kukulka et al., 2012;
281 van Sebille et al., 2015). In coastal regions, local hotspots may also be generated by the influx
282 of plastics from river systems (Frias et al., 2014).

283 Although not commonly appreciated, plastics are also transported out of marine and coastal
284 ecosystems to terrestrial and atmospheric environments through wind and wave action (e.g.
285 storm surges) (Horton et al., 2017a). These transport pathways redeposit plastic to
286 coastal/terrestrial systems. For example, a large proportion of plastic litter present across
287 coastal regions is derived from marine environments, transported and deposited through wave
288 action (Browne et al., 2011). The suspension of plastic by aeolian processes is responsible for
289 transferring particles from marine to atmospheric systems, with microplastics potentially
290 aerosolised alongside the sea surface microlayer (Wright & Kelly, 2017). Plastic particles will
291 also settle through the water column and become incorporated in marine sediments (Van
292 Cauwenberghe et al., 2013). The rate at which this process occurs is influenced by
293 amalgamation within faecal pellets (Cole et al., 2016) or incorporation into algal structures

294 (Long et al., 2015). The accumulation of plastic in benthic sediments provides a temporary
295 store which may be remobilised by physical and biological processes, although there is limited
296 research on the mechanisms of plastic transport in marine systems (Martin et al., 2017).

297 **2.5. Underrepresented ecosystems**

298 There are several ecosystems where the occurrence of plastics remains largely unexplored. In
299 particular, groundwater and cryosphere ecosystems, as well as riparian ecotones have received
300 relatively limited attention. Yet the potential for these ecosystems to significantly influence the
301 storage and flux of plastics is not negligible.

302 Within the cryosphere, the remobilisation of plastics resulting from increasing melt-rates, may
303 provide a significant source of plastics to other ecosystems. Existing research demonstrates
304 high concentrations of plastic debris (40–250 MP L⁻¹ melted ice) stored in Arctic sea-ice
305 (Obbard et al., 2014; Peeken et al., 2018). The release of plastic from sea ice is likely an
306 important contributor to the flux of plastic within marine systems. As an example, the net
307 melting of sea ice between 2011 and 2016 is estimated to have released 7.2–8.7 x 10²⁰ MP in
308 the size range of 0.011–5 mm (Peeken et al., 2018). Within glacierised hydrological
309 catchments, patterns of continuing deglaciation may lead to a significant release of plastic,
310 however, little is known about the distribution of plastic contamination across these
311 compartments of the cryosphere.

312 Groundwater systems provide important stores and transfer pathways of pollutants, e.g.
313 pesticides (Toccalino et al., 2014), so it is likely that these systems would store and transport
314 micro- and nano-plastics (Rochman, 2018). While interstitial pore space within rock strata,
315 hydrologic connectivity and subsurface flow paths, limit particle sizes, it is likely that some
316 systems like karsts may also transport or store larger particle sizes. The relative contribution of
317 groundwater to the total flux of plastic pollution, is likely relatively restricted due to pore size
318 restrictions.

319 Riparian ecotones, as the main interface between terrestrial and freshwater systems, are also
320 obvious points for transfer and storage. Recent studies have used citizen science techniques to
321 quantify the levels of macroplastic litter along riverbanks and riparian zones, observing an
322 average of 0.54 ± 1.2 litter items m^{-2} across Germany (Kiessling et al., 2019). Riparian zones
323 likely provide temporally variable effects on the storage and transfer of plastic pollution. For
324 example, during floods plastics are prone deposition above the bank, namely if the riparian
325 vegetation increases retention. River level (water height), velocity, vegetation type, coverage
326 and roughness, are here key regulating factors in the storage, release or transport of plastics in
327 riparian ecosystems.

328 **3. Biological retention and cycling of plastics across catchments**

329 Plastics are transported, ingested, cycled and sometimes retained by biota. Biological
330 interactions such as ingestion also alter the physical and chemical properties of these plastics,
331 which in turn influences the movement (flux and storage) of plastic between ecosystems. As
332 an example, as plastics are incorporated into faecal pellets, phytoplankton aggregates or biofilm
333 matrices, the otherwise buoyant plastic particles gain a propensity to sink, leading to increased
334 deposition in sediments (Cole et al., 2016; Long et al., 2015; Rummel et al., 2017). The
335 aggregation of particles as a result of egestion may subsequently alter the distribution of
336 plastics whilst also increasing their bioavailability to organisms feeding on faecal material
337 (Ward & Kach, 2009). Once in food webs, plastic particles may be retained through cycling
338 between trophic levels, moving upwards through the food web as a consequence of predation
339 (e.g. Nelms *et al.*, 2018) and re-entering the basal resources through egestion. The residence
340 time of plastic particles within the biological component of food webs is unknown. Higher
341 plants may also retain plastic, with significant aerial accumulation, in the branches and foliage
342 of plants in both terrestrial and riparian systems, as well as entangled in subterranean and
343 subaquatic plant material. The storage of plastics in the biotic components of ecosystems,

344 ultimately however, is restricted with the majority of plastic particles likely to return to the
345 environments from which they were sequestered, through a series of processes including
346 egestion and decomposition (Wright et al., 2013b).

347 Organisms may also facilitate the transport of plastics across habitats and ecosystems. For
348 example, the dispersal of some organisms across the landscape may act to redistribute plastics
349 at a range of spatial scales, from microhabitats to continents. Across short distances, organisms
350 such as worms and collembolans may transport plastics via ingestion, attachment and active
351 transport (Maaß et al., 2017). Recent studies have also indicated the ability of mosquitos (*Culex*
352 *pipiens*; Linnaeus 1758), to transport microplastics (2 and 15 µm) from aquatic to terrestrial
353 and atmospheric systems (Al-Jaibachi et al., 2018). For micro-organisms, transport may be
354 relatively localised, yet larger organisms (e.g. cetaceans) may facilitate long distance transport.
355 Such processes are likely responsible for distributing plastic across the landscape and
356 potentially generating plastic pollution in regions previously unaffected by non-biological
357 fluxes of plastics. These processes, however, are unlikely to be significant relative to
358 redistribution by physical processes (e.g. winds and tides). The interaction between organisms
359 and plastic transport is an emergent field of research, requiring further attention.

360 **4. Ecological effects of plastics**

361 Ecological impacts on biota from exposure to plastic may stem from an array of mechanisms.
362 While current literature predominantly reports physical impacts on biota or ecosystem function,
363 chemically-related effects, facilitated by the adsorption properties of plastic surfaces, are also
364 likely (Fig. 2).

365 One of the largest bodies of observational evidence for the lethal effects of plastic pollution
366 lies in records of entanglement and external physical damage. Although the majority of
367 information available implicates large plastic items, for example fishing nets and rope (e.g.
368 (Jacobsen et al., 2010), these physical effects also pose a problem for small organisms. For

369 example, zooplankton exposed to microplastic fibres (1.7×10^4 – 5.4×10^5 fibres L⁻¹), were
 370 observed with antennal and carapace deformities resulting from external damage (Ziajahromi
 371 et al., 2017). The concentrations utilised within this study, however, do not represent
 372 environmentally relevant concentrations. Observations in terrestrial systems have also
 373 identified the lethal effects of entanglement on American crow (*Corvus brachyrhynchos*;
 374 Brehm, 1822) nestlings (Townsend & Barker, 2014). The effects of entanglement, however,
 375 occur at the individual level, and there remains limited evidence to suggest that these
 376 potentially lethal impacts support significant effects across populations. Furthermore, the
 377 effects of plastic exposure on sensitive tissues have generally been carried out at concentrations
 378 exceeding those observed within natural environments (Phuong et al., 2016).

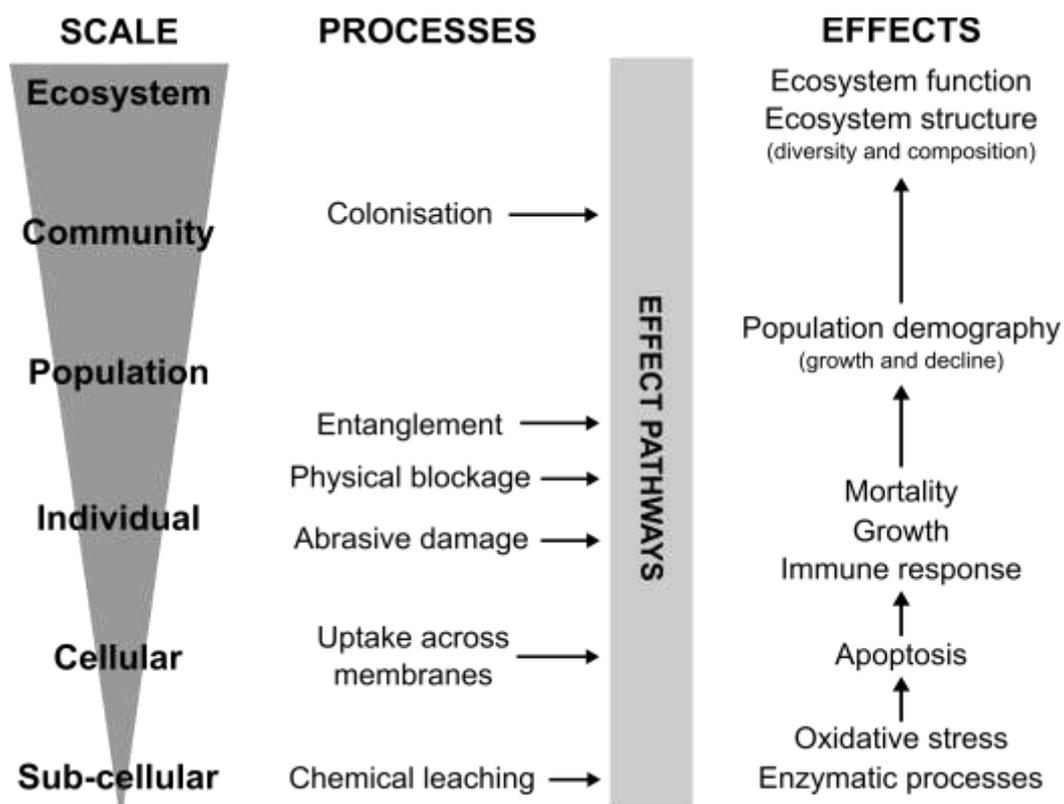


Fig. 2. Observed and predicted mechanistic effects of microplastic exposure in natural environments. Potential mechanistic effects are determined from theoretical and empirical studies, as well as perceived mechanisms of action which have yet to be investigated. Bold effects and responses are those that have been investigated within the literature.

379 The ingestion of plastic has also been a focus of existing research with the severe effects (e.g.
380 reduced growth and mortality) of plastic blockages in the digestive tracts of organisms
381 attracting attention (Derraik, 2002; Gall & Thompson, 2015). These effects are observed across
382 the biosphere, although they have so far been infrequently recorded on a small number of
383 individuals. A range of more subtle effects, however, may be generated by plastic ingestion.
384 The ingestion of plastic maintains the potential to generate reductions in the adsorption of
385 nutrients by the organism (based on reduced uptake of nutrients and intake of actual food
386 items), alterations in the gut microbiota and also reduce the energy budget of organisms leading
387 to several subsequent impacts, including reduced feeding, decreased activity, reduced
388 reproductive output and eventually mortality (see Wright *et al.*, 2013a; Au *et al.*, 2015; Watts
389 *et al.*, 2015; Zhu *et al.*, 2018). Thus far, exposure to a range of plastic types, sizes and shapes,
390 has generated relatively limited adverse effects on aquatic organisms, including fish and
391 invertebrates (Foley *et al.*, 2018). As a specific example, a battery of six freshwater
392 invertebrates exhibited limited responses in growth, reproduction and survival to polystyrene
393 microplastics (20–500 μm) at concentrations of 0–40% sediment dry weight (Redondo-
394 Hasselerharm *et al.* 2018). However, the complexity of plastics make effects difficult to predict
395 as the shape, size and type of polymer can influence particle toxicity. For example, microfibers
396 have been shown to have a greater adverse effect than microbeads due to entanglement and
397 carapace damage in water fleas (*Ceriodaphnia dubia*; Richard, 1894) (Ziajahromi *et al.*, 2017).
398 In addition to physical effects, plastics can also leach toxic compounds, generating effects
399 within organisms that come into contact with plastics. Plastics are complex compounds with a
400 variety of added chemicals (plasticisers, hardeners, flame retardants, surfactants and synthetic
401 dyes) to give them their specific properties. Over time these plasticisers leach out and can often
402 act as toxic or endocrine disrupting chemicals within the environment (Hermabessiere *et al.*,
403 2017). A wide range of toxic compounds have been identified as plastic additives, including

404 bisphenol a (BPA), nonylphenol, polybrominated flame retardants and phthalates
405 (Hermabessiere et al., 2017). These leachates have been shown to negatively affect
406 development in the early life stages of invertebrates (Nobre et al., 2015), whilst also generating
407 reproductive abnormalities in a range of organisms (Browne et al., 2007).

408 Plastics may act as vectors within the environment, facilitating the enhanced transport of
409 persistent organic pollutants (POPs) and other chemicals through biotic and abiotic
410 components of ecosystems (Ziccardi et al., 2016). The “vector effect” has predominantly been
411 portrayed as detrimental, with a range of harmful substances adsorbed to the surfaces of plastics
412 (Koelmans et al., 2016) and the possibility to potentiate the toxicity of other chemicals, e.g.
413 triclosan (Syberg et al., 2017). The role of microplastics in organic chemical bioaccumulation,
414 however, is unclear. While previous studies have shown increased bioaccumulation of
415 chemicals when adsorbed to plastics (Bakir et al., 2014a, 2014b), recent evidence suggests that
416 the role of microplastics in chemical transfer to organisms may be negligible when compared
417 to other natural organic matter (Koelmans et al., 2016). Further to this, only a small fraction of
418 contaminants appear to adsorb to the surface of common microplastics (polyethylene and
419 polypropylene), with only hydrophobic compounds shown to consistently adsorb to particles
420 (Seidensticker et al., 2018). Other studies have indicated that the presence of plastics during
421 contaminant exposure maintains variable effects. For example, polystyrene microplastics (0.4–
422 1.33 mm) under provided a “cleaning” mechanism, whereby pollutants, in this case PCBs, are
423 transferred from the tissues of the organisms to the microplastic particles (Koelmans et al.,
424 2013). In another study, the addition of polyamide microplastic particles (15–20 µm) to
425 experimental chambers reduced the aqueous concentrations of BPA, leading to a reduction in
426 the levels immobilisation of *Daphnia magna* (Straus, 1820) in comparison to exposure to only
427 BPA (Rehse et al., 2018). The degree to which chemicals sorb to plastics is also highly variable
428 and dependent upon the environmental conditions (e.g. salinity, temperature, pH and organic

429 matter), chemical characteristics and plastic type (Teuten et al., 2009). Although other
430 substrates may provide a greater influence on the bioaccumulation of pollutants, the sorption
431 of pollutants to plastics may enable the transfer of pollutants over greater distances compared
432 to organic pollutants associated with denser sediment particles (Nizzetto et al., 2016).

433 The surface of plastics provides a suitable substrate for colonisation by microbial and
434 invertebrate communities (McCormick et al., 2016; Reisser et al., 2014). Within urban river
435 systems, plastics have been identified as a unique and important substrate for the colonisation
436 of aquatic microbial biofilms (McCormick et al., 2014). Similar findings have been presented
437 within marine systems, with diatoms, phytoplankton and cyanobacteria colonising plastic
438 particles suspended within the water column (Oberbeckmann et al., 2016; Reisser et al., 2014;
439 Zettler et al., 2013). While in some instances the microbial communities on these plastic
440 particles maintained comparable species richness and evenness to communities present on
441 natural substrates (Zettler et al., 2013), other studies (e.g. McCormick et al. 2014) demonstrated
442 that microbial communities inhabiting microplastic particles maintained a different taxonomic
443 structure to those present in the water column and on suspended organic matter. An increasing
444 body of research has also identified the colonisation of plastic particles by harmful microbes,
445 which could lead to further deleterious effect upon organisms interacting with these particles
446 (Keswani et al., 2016). For example, the ingestion of these particles may expose organisms to
447 a range of adverse effects derived from harmful microbes and lead to long-range transport of
448 these microbes to regions that would not normally be found (Kirstein et al., 2016; Viršek et al.,
449 2017). Further to this, recent studies have indicated that the intense interactions within
450 microbial communities on microplastic particles enables the increased plasmid transfer
451 between phylogenetically-diverse bacteria, potentially facilitating the spread of antibiotic
452 resistance across aquatic systems (Arias-Andres et al., 2018).

453 While individual-level effects are widely demonstrated for macro- and in some cases micro-
454 plastics, evidence for population and food web level effects remains restricted. As highlighted
455 by Koelmans *et al.* (2017), a range of issues currently limit our understanding of the ecological
456 risks resulting from exposure to plastic pollution. The majority of current individual-level
457 assessments suffer from three dominant limitations; (i) the absence of ecologically relevant
458 metrics, (ii) a limited understanding of organism-plastic encounter rates for given exposure
459 concentrations, and (iii) the restricted development of dose-response relationships across
460 suitable concentration ranges. As a result, the individual-level and in some cases population
461 effects identified within contemporary experimental assessments are not directly applicable to
462 natural systems. Developing an improved mechanistic understanding of the effects of plastic
463 pollution, as well as following lessons learnt in previous environmental toxicology assessments
464 (e.g. non-monotonic relationships, mixture effects, indirect effects) is likely to improve our
465 understanding of the ecological risks posed by plastic pollution.

466 **5. Understanding plastic-biota links**

467 The mechanisms through which plastic exposure effects occur are strongly dependent upon the
468 characteristics of plastic particles, including size, shape, colour and polymer type (Lambert et
469 al., 2017). As an example, polyvinyl chloride is generally more toxic than polyethylene and
470 polypropylene, due to the greater toxicity of its additives and subsequent leachates (Lithner et
471 al., 2012). The diversity of physical and chemical characteristics exhibited by plastic particles,
472 throughout their lifecycle and as they degrade in natural systems, means that the potential
473 ecological effects resulting from plastic pollution are extremely variable.

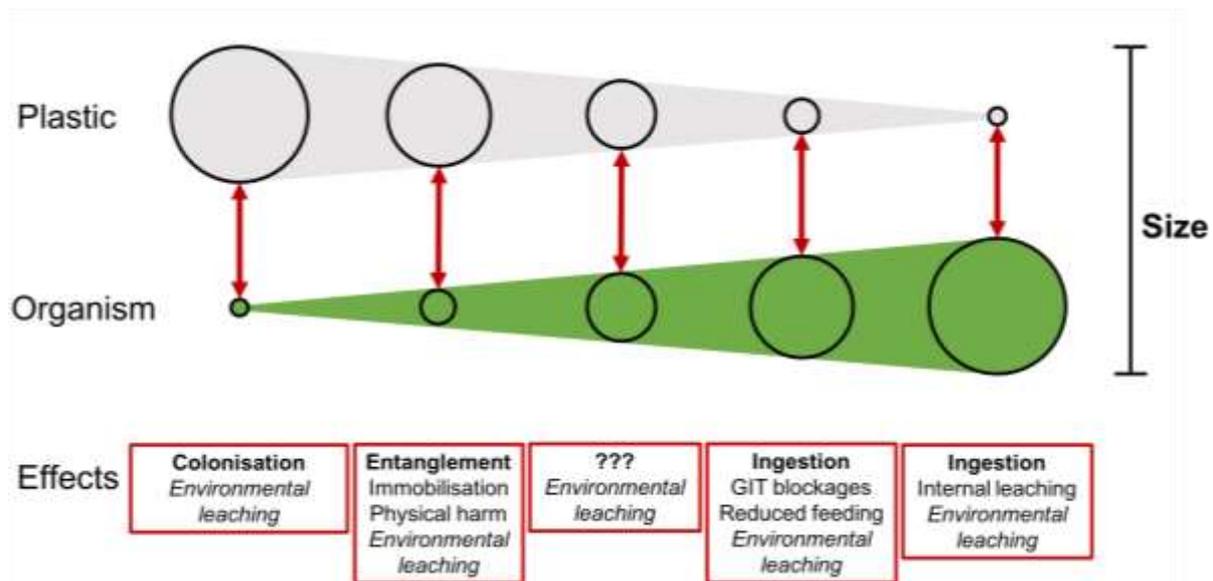


Fig. 3. Conceptual relationship between the organism-to-plastic size ratio and the dominant effects derived from direct interactions between organisms and plastic pollution at these scales. These general relationships are independent of actual size, yet bounded by the maximum sizes of both plastic particles and organisms across the globe. Examples of potential effects at different size ratios are presented in red boxes. **Bold text** indicates the nature of organism-plastic interactions, *italic text* indicates indirect effects.

474 The relationship between organisms and plastic size appears particularly important in
 475 determining the nature and severity of ecological effects (Fig. 3). Plastics significantly larger
 476 than the target organism can provide a novel substrate for colonisation for the smaller
 477 organisms (as described for microbial communities (Reisser et al., 2014) and invertebrates
 478 (Davidson, 2012)), or become a cause for entanglement and associated effects for larger
 479 organisms (Gall & Thompson, 2015). Plastics of large, yet ingestible size classes present the
 480 potential for gastrointestinal blockages (Gall & Thompson, 2015). Finally, particles that are
 481 ingestible in size, yet too small to present physical risks (e.g. digestive blockages and
 482 entanglement) propose a large range of potential effects, including the leaching of toxic
 483 chemicals directly to organisms (e.g. Teuten *et al.*, 2009). These general rules provide a good
 484 indication of the potential effects of different plastic particles, however, it should be noted that
 485 organisms are able to interact with all sizes of plastic pollution, with wide range of possible

486 effects not detailed above. Furthermore, a range of indirect effects are also presented by
487 particles of various sizes (Fig. 3). As an example, chemicals from macro-plastics leach into the
488 surrounding environment, providing the potential to indirectly affect organisms through the
489 uptake and subsequent effects.

490 Thus far, the observed effects of plastic pollution are mainly limited to the size classes utilised
491 in experimental manipulations (0.04–500 μm) (Foley et al., 2018) or the size classes observed
492 in fatalities in natural systems (0.3–10 m) (Jacobsen et al., 2010). Thus, the nature, mechanisms
493 and severity of effects across the spectrum of plastic sizes is unknown. Further research
494 investigating the interactions between organism size, plastic characteristics and ecological
495 effects is important for developing a comprehensive knowledge of ecological risks posed by
496 plastic pollution.

497 **6. Plastic pollution in a social and economic context**

498 Plastic presents a number of societal benefits, and has promoted a range of technological
499 advances. However, increasing awareness of potential environmental impacts, predominantly
500 focused on marine systems (Thompson, 2017), is also highlighting potential knock-on effects
501 across a range of economic sectors, including the water industry, tourism and fishing. Data are
502 geographically restricted, yet indicate the potential for widespread socio-economic effects of
503 plastic pollution.

504 Fishing activity (commercial and recreational), in particular, is negatively impacted by plastic
505 debris, reducing and damaging catches (Thompson, 2017); for example 86% of Scottish fishing
506 vessels surveyed had incurred restricted catches as a result of marine litter (Mouat et al., 2010).
507 Furthermore, entanglement within marinas and harbours appears a significant problem, with
508 70% of surveyed marinas and harbours reporting that users had experienced incidents with
509 litter (Mouat et al., 2010). Contamination of fish stocks may also provide a significant
510 economic cost, although concentrations of plastic within individual fish is relatively low (e.g.

511 1–2 pieces per organism: Foekema *et al.*, 2013; Lusher *et al.*, 2013). Nevertheless, the negative
512 perception of this contamination by consumers may be enough to affect the marketability of
513 commercial organisms (GESAMP, 2016).

514 Another economic sector significantly impacted by plastic pollution is tourism. Public
515 perceptions of plastic pollution is likely to influence where people choose to visit. For example,
516 visitors to coastal regions cited the presence of litter as a factor influencing the locations they
517 visited (Brouwer *et al.*, 2017). To mitigate the negative effects of litter local authorities
518 implement cleaning operations, which within the UK is estimated to cost £15.5 million
519 annually (Mouat *et al.*, 2010). The combination of removal costs and potential reductions in
520 tourism present a major concern the tourism industry.

521 Expenses are also incurred through increased research and development relating to water
522 treatment methods, damages to equipment and blockages of infrastructure. In particular,
523 cosmetic wipes have been shown to cause problems – blocking sewage infrastructure and
524 generating private and public effects (Drinkwater & Moy, 2017). The net costs of plastics to
525 the water industry are, however, difficult to calculate as removal and blockages occur alongside
526 other problematic items (e.g. fat, grease and organic pollutants).

527 Human health is potentially impacted by plastic pollution. Beach litter has been shown to cause
528 physical harm (Werner *et al.*, 2016), nevertheless, the vast majority of these incidents relate to
529 metal and glass as opposed to plastic. Psychological effects of plastic litter are also observed
530 with negative effects on the ‘restorative value’ generated by visiting a polluted habitat (Wyles
531 *et al.*, 2016). The health of individuals may also be affected by any of the suite of effects
532 highlighted in the previous section *Ecological effects of plastic*. This includes the transport of
533 potentially harmful microbes and chemicals (see Keswani *et al.*, 2016), as well as the physical
534 effects of plastic ingestion. More work is nevertheless required to detail the specific health
535 risks to human populations generated by global plastic pollution.

536 **7. Plastic pollution as an agent of global change**

537 The relative impact of plastic pollution on ecosystems in comparison to other global stressors
538 is poorly understood. Contextualising the effects of plastic pollution within a multi-stressor
539 environment is an important development and to date, the importance of plastic effects in
540 comparison to urbanisation, habitat fragmentation, other pollutants, increased temperatures,
541 hydrological changes and invasive species, for example, is unknown. Within the terrestrial
542 environment, nevertheless, recent investigations across soil ecosystems, plastics have been
543 identified as a potential agent of global change, altering the function of soils (water retention,
544 microbial activity, soil structure and bulk density) and affecting their role in the function of the
545 wider environment (de Souza Machado et al., 2018b). Furthermore, microplastics have been
546 shown to potentiate the effects of other xenobiotic pollutants, in this case the antimicrobial
547 chemical triclosan (Syberg et al., 2017). The interactions between other stressors and plastic
548 pollution therefore provides the potential to generate negative effects across natural
549 ecosystems. Future mitigation and management strategies will require a better understanding
550 of the relative importance of global pressures, and also their interactions.

551 **8. Future research at the catchment-scale**

552 Understanding the movement of plastic through hydrological catchments is an important step
553 in determining the source to sink dynamics of plastics within natural systems. This review
554 highlights that catchment-scale assessments are currently limited to theoretical assessments,
555 but also provides a framework to structure future investigations, with hypotheses already
556 generated by theoretical models. Supporting existing studies with comprehensive field-based
557 and experimental datasets is the logical next step in developing a comprehensive body of
558 research assessing catchment-scale transport and effects of plastic pollution. To date, empirical
559 studies have focused on individual ecosystems providing an analysis of plastic distribution and
560 plastic-organism interactions. Catchment scale assessments are an important next step for

561 research. Detailed below, are several important developments required to facilitate the advance
562 of catchment-scale investigations.

563 **Methods for tracing plastic transport processes.** Contemporary empirical assessments are
564 not able to elucidate the sources and pathways of plastic particles, as once particles enter the
565 environment tracing sources becomes problematic. Furthermore, the longer particles are
566 exposed to physical, chemical or biological processes, the more their transformation
567 exacerbates difficulties identifying sources. Novel methods of tracing plastics have yet to be
568 developed, yet using tracer studies to support existing models will allow for directed research
569 projects attempting to bridge current knowledge gaps.

570 **Hotspots and sinks of plastic pollution.** Knowledge surrounding the distribution of plastic
571 pollution across catchments is limited. Understanding where and how high plastic
572 concentrations arise in space and time is required for assessments detailing how plastic
573 concentrations may vary across hydrological catchments. The importance of such
574 developments is further emphasised by a recent study which identified the highest global
575 concentration of microplastics recorded within riverine sediments (517,000 MP m⁻²) (Hurley
576 et al., 2018). Assessments of heterogeneity are required at a range of spatial scales, from local
577 patch-dynamics at centimetre to metre scales, to comparisons between entire habitats and
578 ecosystems. Understanding spatial variation and potential sinks of plastic will allow for an
579 improved understanding of transport processes leading to the deposition of plastics across the
580 landscape, and importantly provide more accurate risk maps for biota.

581 **Quantification of source contributions.** Although estimates exist for the net contribution of
582 plastic from specific ecosystems, e.g. freshwater (Lebreton et al., 2017) and terrestrial (Horton
583 et al., 2017a) systems, the importance of specific sources in contributing to these plastic
584 burdens across these environments is poorly understood. Further study of plastic sources, in
585 particular diffuse contributions, is required to better resolve the source-flux-sink nexus within

586 catchments, detailed in previous sections. Developing more accurate methods of quantification,
587 designed to detect low concentrations of plastic and nano-plastics will enable the detection of
588 a wider range of plastics (e.g. tyre dust), allow for an improved understanding of plastic
589 pollution across catchments and bridge the current gap between estimated inputs of plastic into
590 catchments and measured environmental concentrations. Through investigating the
591 characteristics and concentration of plastics released from each potential source, a mixing-
592 model type assessment can be used to understand the entrance and flux of plastics within
593 catchments (Fahrenfeld et al., 2018). Further to this, determining the specific contributions
594 from sources will enable targeted mitigation, ultimately aimed at preventing the entrance of
595 plastics into the natural environment.

596 **Determining the applicability of catchment assessments.** Catchment-scale assessments are
597 dependent upon catchment characteristics, including but not limited to: size, relief, land cover,
598 water quality, hydrological connectivity and geomorphological features. The degree to which
599 plastic studies within individual catchments are applicable across the wider landscape is
600 unknown. To answer this question, multiple catchment assessments are required to determine
601 the relative importance of catchment-specific processes (e.g. hydrological flow paths,
602 subsurface characteristics and catchment geology) in comparison to more generalisable
603 characteristics (e.g. land cover, population density, human activities). An understanding of the
604 importance of processes at a range of spatial and temporal scales, is also required in order to
605 appreciate the extent to which relationships are applicable across catchments.

606 **9. Conclusions**

607 Our understanding of the effects of macro-plastics within ecosystems indicates the potential
608 negative effects of these pollutants. Knowledge regarding the nature and severity of effects
609 derived from smaller plastic particles, at environmentally relevant concentrations, however,
610 remains restricted. The array of mechanistic effects identified by studies nevertheless indicate

611 the potential for adverse effects within natural systems. The significant potential for effects
612 coupled with recent research indicating the relative global ubiquity of plastics provides a
613 perceivable risk to a range of ecosystems. In spite of this, we are only starting to understand
614 the fluxes and pools of plastics within a range of ecosystems. This knowledge is nonetheless
615 fundamental for mitigating existing and future plastic pollution. It is apparent that further
616 research is required to better understand the interactions between plastic pollution and
617 organisms in many ecosystems. Furthermore, a comprehensive understanding of potential
618 ecological risks presented by plastics remains absent with a range of potential adverse effects
619 remaining unexplored. The existing ecological risk presented by plastic pollution is estimated
620 to continue into the future as a result of predicted increases in production of plastics, the
621 significant persistence of plastic particles and the degradation of existing plastic pollution
622 generating increases in micro- and nano-plastic concentrations across the globe.

623 **Conflicts of interest**

624 The authors declare no conflicts of interest.

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