

Quantifying impacts of plastic debris on marine wildlife identifies ecological breakpoints

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1 Quantifying sub-lethal effects of plastics ingestion on marine wildlife
2 is difficult, but key to understanding the ontogeny and population dyn-
3 amics of affected species. We developed a method that overcomes
4 the difficulties by modelling individual ontogeny under reduced en-
5 ergy intake and expenditure caused by debris ingestion. The pre-
6 dicted ontogeny is combined with a population dynamics model to
7 identify ecological breakpoints: cessation of reproduction or neg-
8 ative population growth. Exemplifying this approach on loggerhead
9 turtles, we find that between 3% and 25% of plastics in digestive con-
10 tents causes a 2.5% to 20% reduction in perceived food abundance
11 and total available energy, resulting in a 10% to 15% lower condi-
12 tion index and 10% to 88% lower total seasonal reproductive output
13 compared to unaffected turtles. The reported plastics ingestion is in-
14 sufficient to impede sexual maturation, but population declines are
15 possible. The method is readily applicable to other species impacted
16 by debris ingestion.

Physiological energetics | Conservation | Sea turtles | Stressors | Debris ingestion

1 Plastic debris in marine environments represents a global
2 anthropogenic pressure that, despite heightened aware-
3 ness and preventive regulations (Thompson et al., 2009), is
4 likely to keep increasing for the foreseeable future (Moore,
5 2008; Jambeck et al., 2015). Annually, between 4–12 million
6 tons of plastic waste from land (Jambeck et al., 2015) and 1–2
7 million tons from rivers (Lebreton et al., 2017) find their way
8 into the sea, making plastic items and microplastic particles
9 omnipresent in global oceans (Derraik, 2002; Moore, 2008;
10 Barnes et al., 2009; Cozar et al., 2014; Eriksen et al., 2014).

11 Negative effects of plastic debris on over 700 marine species
12 have already been documented, and sea turtles are among
13 the top three wildlife groups impacted by plastics (Gall and
14 Thompson, 2015). Namely, all seven species of sea turtles in all
15 major oceans have been found to ingest plastic debris (Schuyler
16 et al., 2014a; Nelms et al., 2016; Lynch, 2018), with as many
17 as 52% of all sea turtle individuals ingesting plastics sometime
18 during their lifetime (Schuyler et al., 2016). Plastic debris
19 has therefore been identified as one of the global research and
20 conservation challenges for sea turtles (Hamann et al., 2010;
21 Wallace et al., 2011).

22 Plastics ingestion occurs when sea turtles encounter and
23 ingest debris accidentally (i.e., opportunistically) or mistake
24 debris for prey and ingest it actively (Schuyler et al., 2012;
25 Narazaki et al., 2013; Schuyler et al., 2014b). Data suggest that
26 oceanic turtles are less selective and more likely to encounter
27 floating debris while feeding on pelagic organisms (Schuyler
28 et al., 2012), which makes them vulnerable to opportunistic
29 plastics ingestion (Schuyler et al., 2014a; Nelms et al., 2016;
30 Schuyler et al., 2016). This is in contrast to neritic sea turtles

who are more selective and encounter fewer floating items
 while feeding on benthic organisms, which in turn results in
 predominantly active plastics ingestion (Schuyler et al., 2012).
 During feeding, turtles may also indirectly ingest plastics
 already consumed by their prey (Thompson et al., 2004; Barnes
 et al., 2009; Rochman et al., 2015). Indirect ingestion, in
 addition to environmental debris prevalence and turtle feeding
 ecology, partly depends on prey behaviour. Furthermore,
 indirectly ingested plastics are likely to be fragmented into
 small, hard-to-detect pieces termed microplastics that exert
 both physical and biochemical effects on individuals (Rezania
 et al., 2018). Biochemical effects intensify, whereas physical
 subside, inversely to particle size (Batel et al., 2016), implying
 that larger macroscopic debris causes mostly physical effects.

The physical effects of plastics ingestion cause a range of
 lethal and sub-lethal consequences. Ingested debris blocks,
 damages, or reduces the volume of a turtle's digestive system
 (Gramentz, 1988; Stahelin et al., 2012; Schuyler et al., 2014a),
 thus decreasing the caloric intake at best, and fatally harm-
 ing the turtle at worst (Mascarenhas et al., 2004; Lazar and
 Gračan, 2011; Casale et al., 2016). Sub-lethal consequences
 of plastics ingestion, observed in other species (Connors and
 Smith, 1982; Ryan, 1988; Yamashita et al., 2011) and sea tur-
 tles alike (McCauley and Bjorndal, 1999; Mascarenhas et al.,
 2004; Stahelin et al., 2012), include slower weight gain, de-
 creased fat deposits, less feeding activity, longer gut residence
 times of ingested material, and inflammation. Ingested plastics
 therefore has the potential to substantially reduce the physio-
 logical condition of individuals, thereby lowering their chances
 of survival or reproduction and, in turn, lowering the popula-
 tion growth rate. Our goal here is to lay the methodological
 foundation for quantifying these effects.

Quantifying sub-lethal effects of plastics ingestion is chal-
 lenging. Studies of controlled plastics ingestion involving sea
 turtles are rare and limited to short periods of time (Lutz,
 1990; Narazaki et al., 2013; Fukuoka et al., 2016) or a single
 life stage (McCauley and Bjorndal, 1999). The most elabo-
 rate efforts to date have produced a quantitative link between
 the number of ingested items and mortality (Santos et al.,
 2015; Wilcox et al., 2018), whereas sub-lethal effects have only
 been mentioned as “an area of considerable uncertainty and
 concern” for sea turtles (Gall and Thompson, 2015). Studies
 on quantitative, let alone mechanistic, links between plastics

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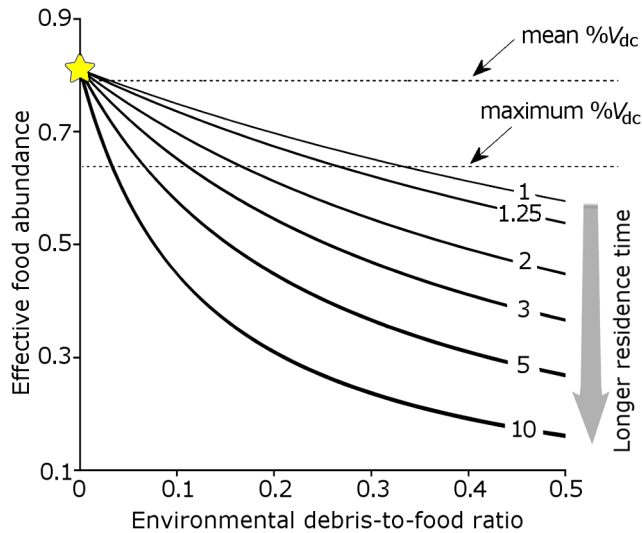


Fig. 1. Presence of plastic debris reduces effective food abundance experienced by sea turtles. As the prevalence of plastics in the environment increases relative to food, animals are more likely to ingest plastics. A direct consequence is that sea turtles experience less food abundance (top black curve) than they would in a pristine ocean (yellow star). The effect is compounded if, in addition, plastic debris has a longer residence time than food (black curves indexed by relative residence times from 1.25 \times to 10 \times). The horizontal dashed lines mark 3% and 25% of debris in digestive contents, which corresponds to mean and maximum reported values for percentage of debris in stomach contents (Frick et al., 2009).

74 ingestion and sub-lethal physiological or life-history effects on
 75 sea turtles are thus hard to come by. Nevertheless, there is still
 76 a lot of information on sea turtles that could be assimilated
 77 by mechanistic modelling approaches. Loggerhead turtles
 78 (*Caretta caretta*) are of prime interest because of the extensive
 79 information on this species from field studies, strong concerns
 80 about the damaging effects of plastics ingestion (Schuyler
 81 et al., 2014a; Nelms et al., 2016; Schuyler et al., 2016; Lynch,
 82 2018), and the status as a potential bio-indicator for monitoring
 83 plastics ingestion in the Mediterranean (Fossi et al., 2018).
 84 Serious effects of plastics ingestion have been documented for
 85 post-hatchlings (McCauley and Bjorndal, 1999; Ryan et al., 2016),
 86 oceanic-stage juveniles (Pham et al., 2017), and juveniles and
 87 adults (Lazar and Gračan, 2011; Campani et al., 2013; Schuyler
 88 et al., 2014a), with some authors quoting loggerhead turtles as
 89 being “specially prone to debris ingestion” (Tomás et al., 2002).
 90 Recent reports put the incidence of plastics ingestion at an average
 91 of 49%, based on 42 studies that examined three or more animals
 92 (Lynch, 2018). The ontogeny of loggerhead turtles has been
 93 characterized from the perspective of physiological energetics and
 94 successfully linked to environmental forcings such as food and
 95 temperature, to capture the observed variability among individuals
 96 within and between populations (Marn et al., 2017b,a, 2019).
 97 Here, we further extend the already validated mechanistic model of
 98 the ontogeny by accounting for plastics ingestion.

100 We mechanistically describe how plastics ingestion affects
 101 the ontogeny of loggerhead turtles, seeking answers to two
 102 primary questions. First, is the reported plastics ingestion
 103 sufficient to impede sexual maturation or egg production of
 104 these turtles? Second, even if individual turtles successfully
 105 mature and reproduce, can the population as a whole maintain
 106 positive growth? In doing so, we identify two ecologically

critical events of interest: one is the cessation of reproduction,
 which we define as an *individual-level* breakpoint; the other is
 non-positive population growth which we define as a *population-level*
 breakpoint. We investigate the extent of plastics ingestion required
 to reach either breakpoint, and how this relates to plastic loads
 currently experienced by sea turtles in the wild. Further methodological
 details are available in the Methods section and Supplementary
 Information (SI) Appendix.

Results

117 Ingesting plastic debris has the effect of reducing effective food
 118 abundance experienced by sea turtles (Fig. 1). We quantify
 119 food abundance as a fraction of the maximum that healthy sea
 120 turtles could ingest during the same time given unrestricted
 121 continuous access to food. The reduction in effective food
 122 abundance occurs because of competition between food and
 123 debris for the limited digestive capacity of gastrointestinal
 124 tract, where the amount of debris compared to food in digestive
 125 contents at any given time is determined by two quantities.
 126 First, a debris-to-food ratio estimates how likely it is for a
 127 turtle to ingest debris instead of food; more debris in the
 128 environment results in proportionately more ingested debris
 129 (Schuyler et al., 2016). Second, the relative debris residence
 130 time estimates how much longer than food ingested debris
 131 remains inside gastrointestinal tract (Lutz, 1990; Mascarenhas
 132 et al., 2004; Valente et al., 2008). (See also SI Appendix,
 133 Fig. S1). As a baseline, our previous analysis has shown that
 134 free-ranging loggerhead turtles experience, on average, food
 135 abundance of 0.81 (Marn et al., 2017b,a). The effect of said
 136 competition between food and debris for the limited digestive
 137 capacity is nearly negligible when there is 3% of debris in
 138 digestive contents, but when there is 25% of debris in digestive
 139 contents, effective food abundance is as low as 0.64. As will
 140 be shown shortly, a drop in food abundance from 0.81 to 0.64
 141 is already enough to considerably affect the life history of sea
 142 turtles.

Ingested debris reduces the energetic scope for growth, mat-

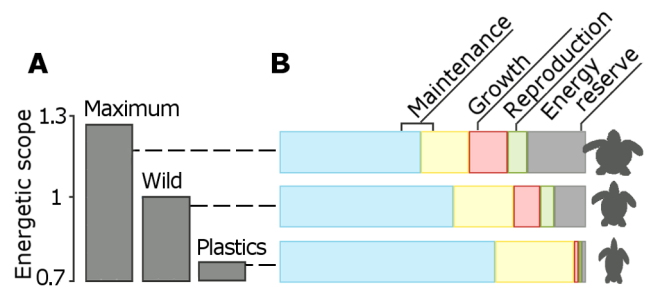


Fig. 2. Ingested debris reduces the energetic scope for growth, reproduction, and reserve accumulation of sea turtles. The decrease happens both in absolute and relative terms. Shown is the total energy budget size and the energy usage of loggerhead turtles at the moment of sexual maturation. **A**, The decrease in effective food abundance from 0.81 for unexposed loggerheads (Marn et al., 2017b) to 0.64 for loggerheads exposed to 25% plastics in digestive contents would mean a 24% reduction in the total energy budget size. Also plotted for reference is the energy budget size at the maximum theoretical food abundance of 1.00. **B**, Beside reducing the total energy budget size, lower effective food abundance leaves a smaller fraction of the energy budget available for growth, reproduction, and energy accumulation. This is because maintenance demands are largely size-driven, and loggerhead turtles mature at approximately the same size (albeit possibly at very different age) as long as food abundance is sufficient for reaching sexual maturation.

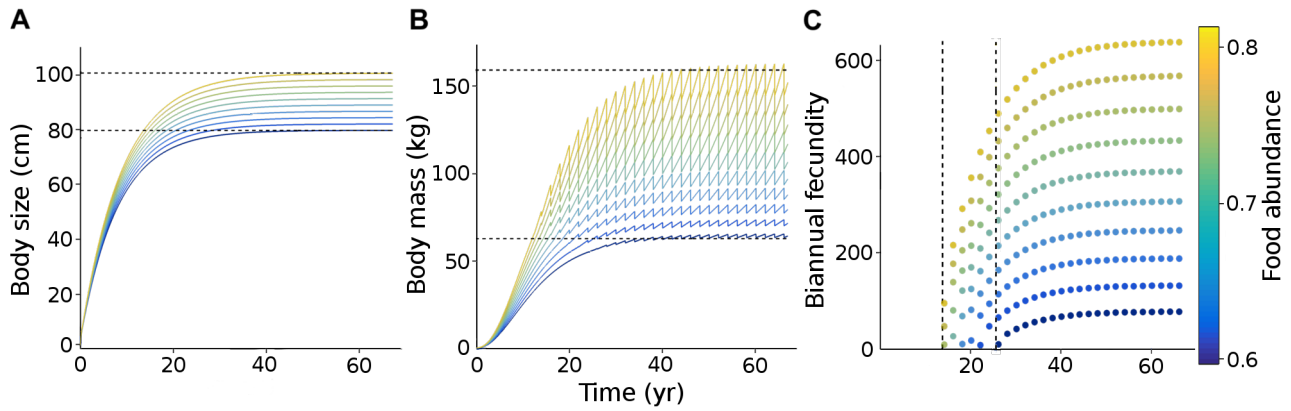


Fig. 3. Ontogenetic development of loggerhead turtles is negatively affected by higher ingestion and longer residence of plastic debris. Ingested debris competes with food for a limited digestive capacity as shown in Fig. 1, thus lowering the effective food abundance below $\approx 80\%$ of the maximum that loggerhead turtles normally experience in the wild. This negatively affects ontogeny. **A**, Body size can decrease from ≈ 100 cm straight carapace length under normal conditions to ≈ 80 cm, when also sexual maturation becomes delayed. **B**, The corresponding reduction in body mass is even more striking, falling from as much as 160 kg to as little as 60 kg. Periodic mass fluctuations are due to allocation to reproduction in adults, culminating in nesting that takes place roughly biannually in loggerhead turtles. **C**, The reduction in, typically biannual, fecundity is the most striking of all as the number of eggs declines from several hundreds to less than a hundred in a nesting season. Despite the large magnitude of the described negative effects, the effects may be difficult to recognize in the field due to imprecise age determination and natural variation between individuals.

144 uration or reproduction, and reserve accumulation (Fig. 2).
 145 For example, a turtle with an average plastics load of 3% in
 146 digestive contents, which is close to the mean reported values
 147 for loggerhead turtles (Frick et al., 2009; Clukey et al., 2017),
 148 experiences $\approx 2.5\%$ reduction in perceived food abundance and
 149 total available energy, causing $\approx 10\%$ lower condition index
 150 and total seasonal reproductive output. With plastics loads in
 151 excess of 25%, which is the reported maximum (Frick et al.,
 152 2009), a fully grown turtle experiences $\approx 20\%$ reduction in
 153 available energy, causing $\approx 15\%$ lower condition index and
 154 $\approx 88\%$ lower total seasonal reproductive output compared to
 155 unaffected turtles. This reduction is due to a smaller energy
 156 budget in absolute terms (Fig. 2A), but also a larger contribu-
 157 tion of maintenance to the budget in relative terms (Fig. 2B).
 158 At the moment of sexual maturation, for example, the decrease
 159 in effective food abundance from 0.81 for unexposed individu-
 160 als to 0.64 for individuals exposed to 25% plastics in digestive
 161 contents corresponds to a 24% reduction in the total energy
 162 budget size. Meanwhile, in unexposed turtles, maintenance
 163 costs comprise 78% of the energy budget, whereas in exposed
 164 ones, as much as 96% of the energy budget. With only 4%
 165 of the daily energy budget left for growth, reproduction, and
 166 reserve accumulation, exposed turtles potentially experience
 167 difficulties allocating energy to egg production simply because
 168 less energy is available for reproduction once maintenance
 169 demands are satisfied.

170 A particular energy budget gives rise to a particular ontog-
 171 eny. For simplicity, the presented simulations describe
 172 North Atlantic loggerhead turtles experiencing a typical con-
 173 stant environment from hatching onward (Marn et al., 2017b);
 174 the percentage of plastics in digestive contents is the only
 175 differing factor between simulations. The ultimate body sizes
 176 of adults, expressed in terms of straight carapace length, range
 177 between 80 cm and 100 cm (Fig. 3A). The corresponding range
 178 of body masses is more dramatic, extending from as little as
 179 60 kg for turtles exposed to plastics to as much as 160 kg for un-
 180 exposed turtles (Fig. 3B). Consistent with observations under
 181 favourable conditions in the wild (Zug et al., 1986; Tiwari and
 182 Bjorndal, 2000; Tucker, 2010; Hawkes et al., 2005), unexposed

183 turtles under simulated conditions mature already at the age of 14, and are at maximum size capable of producing up to 640
 184 eggs in one nesting season (Fig. 3C). The sexual maturation
 185 age of exposed turtles may be substantially prolonged, followed
 186 by reduced egg production. For example, turtles that start to
 187 mature at the age of 26 years produce at maximum size only
 188 75 eggs over a nesting season. When plastics occupies $>30\%$
 189 of digestive contents, effective food abundance drops below
 190 0.61, and turtles never mature or reproduce.

191
 192 Sexual maturation and reproduction are not a guarantee
 193 of population viability. Long-term viability of a population is
 194 secured only if reproduction makes up for mortality, thereby
 195 preventing population declines (see Methods). Our simula-
 196 tions show that the loggerhead turtle population growth, just
 197 like sexual-maturation age, strongly depends on effective food
 198 abundance driven by both the debris-to-food ratio and relative
 199 residence times (Fig. 4). The lowest ratios and residence times
 200 result in the fastest population growth and the youngest sexual-
 201 maturation age. As long as effective food abundance permits
 202 sea turtles to mature by the age of ≈ 22 years, the population
 203 remains viable; longer maturation times cause the population
 204 to decline even though individuals can mature (Fig. 4). Ac-
 205 cordingly, the population-level ecological breakpoint at which
 206 population transits from growth to decline is breached before
 207 the individual-level breakpoint at which individuals stop sex-
 208 ually maturing. Ecological monitoring may thus show that
 209 a population has many reproducing individuals when in fact
 210 their reproductive output is insufficient to offset various causes
 211 of mortality, and the population is headed towards extinction.

212 Discussion

213 Sub-lethal effects of marine debris, though much discussed
 214 (Ryan, 1988; McCauley and Bjorndal, 1999; Barnes et al.,
 215 2009; Mato et al., 2001; Yamashita et al., 2011), have so far
 216 been deemed “particularly difficult to quantify” (Gall and
 217 Thompson, 2015). We have overcome this difficulty by de-
 218 veloping a mechanistic modeling framework that links levels
 219 of biological organisation to quantify the effects of plastics
 220 ingestion on ontogeny and population growth of endangered

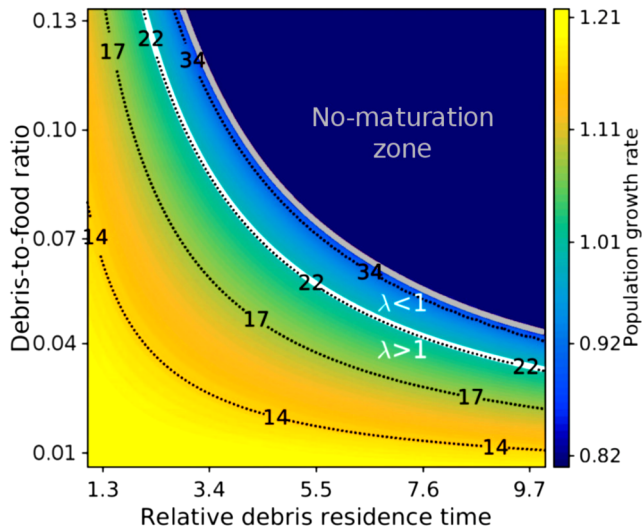


Fig. 4. Reported debris ingestion is unlikely to impede sexual maturation of sea turtles, but population declines are possible. Population growth (colormap) is overlaid with sexual maturation age (contours) as functions of (i) the environmental debris-to-food ratio (y-axis) and (ii) gastrointestinal debris residence times relative to food (x-axis), which determine the percentage of plastics in the digestive contents and consequently the effective food abundance (see Fig. 1). White and gray isocurves respectively delineate zones of population decline (population-level breakpoint) and permanent sexual immaturity (individual-level breakpoint). The two breakpoints are separated by a 25% difference in the debris-to-food ratio or relative residence time. Plastics in turtles is known to equal $\approx 3\%$ of digestive contents on average, but has been observed to exceed 25% (Frick et al., 2009). In the former case, loggerhead turtles in our simulations mature at the age of 14 years, which is well within the safe zone of positive population growth. In the latter case, however, turtles sexually mature only at the age of 25 years; if this became the norm, the population growth would land in the danger zone delineated by a white isocurve in the figure, and the population would be headed to extinction despite the reproducing individuals.

less sensitive to short-term fluctuations. The ontogeny of exposed turtles can be greatly delayed, causing major life-history milestones to be reached at a later age. Accordingly, if measurements of body size and fecundity were supplemented with the age of sea turtles, every individual could be positioned in one or more plots in Fig. 3, and compared against expectations calculated for that population, taking into account the average food abundance and temperature experienced by the turtles. Individuals whose position was closer to the blue (resp., yellow) side of the spectrum of expected ontogenies would then be deemed affected (resp., unaffected) by plastics. Due to inter-individual variability present even after accounting for population-specific characteristics (Marn et al., 2017a, 2019), the results would have to be interpreted in a statistical sense, i.e., based on samples of sufficient size that are currently unavailable.

Gaps in current field data, and by extension knowledge, prompted us to limit simulations to continuous average sub-lethal exposure to plastics. The limitation, however, is not methodological; if high-resolution field data is leveraged with the present methodology, insights into multiple ecological scenarios become possible. To illustrate, we have considered one such scenario: the potentially damaging effects of *short-term spikes* in plastics ingestion on ontogeny and reproductive output of loggerheads. In these additional simulations we mimic reality by making debris ingestion stochastic, with the mean and the maximum percentage of plastics in digestive contents corresponding to the observed values of 3% and 25%, respectively (SI Appendix, Result 2). The periods of favorable feeding conditions allow sea turtles to grow almost as large as if there was no exposure to plastics (SI Appendix, Fig. S5C), yet the periods of occasional starvation decrease condition and thus reproductive output (SI Appendix, Fig. S5D, E). This is worrying because body size in terms of carapace length is the easiest to measure in practice, and thus much more often reported than body mass and fecundity, although a combination of at least two of these quantities would be a better indicator of the true state in the field.

Ecologically important difference in plastics ingestion between pelagic-feeding juveniles and benthic-feeding adults could also be explored with presented methodology and additional data. Debris ingestion is likely to change over a lifetime (Nelms et al., 2016; Schuyler et al., 2014b, 2016), thus raising questions about the realistic exposure in each life stage, and about identifying which life stage is then a bottleneck in the population growth of loggerhead turtles. Similarly, not all species of sea turtles are equally affected by environmental prevalence or ingestion of debris (Nelms et al., 2016; Schuyler et al., 2016; Lynch, 2018). Species with a highly specialized or generally nutritionally poor diet may be at a higher risk by additional energy limitations imposed by plastics ingestion than, e.g., carnivorous loggerheads. Again, answers are methodologically within our reach (Mazaris et al., 2006; Schuyler et al., 2016; Ijima et al., 2019; Stubbs et al., 2020), but remain speculative until high-fidelity field data become available.

Plastics can have ecological consequences extending beyond energy limitation caused by reduction in digestive capacity of individuals. These are most likely to arise from ingested microplastics (Rezania et al., 2018), and manifest as feedbacks between energy allocation and the bioconcentration of leached toxins (Klanjscek et al., 2007). Incorporating ecotoxicology

loggerhead turtles. The results show that amounts of plastics observed in the digestive contents of sea turtles are insufficient to prevent sexual maturation, but population declines as a consequence of plastics cannot be ruled out. Considering indefinite sexual maturation and population decline as individual- and population-level ecological breakpoints, we see that while the former is far from being breached, the latter may have already been breached. A population could therefore be in distress despite a seemingly favorable physiological state of individuals.

Pinpointing indicators of physiological state that are sensitive to plastics ingestion may help identify negatively affected individuals, the prevalence of such individuals in a population, and thereby the population's state. Our framework, for example, predicts that turtles exposed to plastics have a reduced scope for growth, maturation, reproduction, and reserve accumulation. To test this prediction we looked at Fulton's condition factor (Nash et al., 2006) whose values are expected to be lower for exposed individuals due to less reserve accumulation. We used a limited dataset on loggerhead turtles from the Adriatic Sea (SI Appendix, Result 1). Exposed turtles indeed appear to have a lower condition factor than those unaffected by plastics (SI Appendix, Fig. S4), but more data is needed for a decisive conclusion. Ultimately, however, the model suggests that other indicators linked to ontogeny may be more sensitive to the long-term sub-lethal effects of plastics than the condition factor, with the added benefit of also being

309 into the ontogenetic model presented here is conceptually
 310 straightforward (Jager et al., 2006; Jager and Zimmer, 2012),
 311 but uncovering the true nature of such dynamics is not. The
 312 process is, in fact, extremely data-intensive, implying that
 313 biochemical effects on metabolism of plastics ingestion are
 314 probably going to be quantified last.

315 Finally, modular structure and mechanistic underpinning of
 316 the framework allow not only exploration of additional specific
 317 ecological scenarios for the sea turtle related to plastics, but
 318 also generalizations to other species and stressors. The module
 319 for individual ontogeny can directly account for thousands of
 320 species in a rapidly expanding Add-my-Pet database (Marques
 321 et al., 2018), and can be augmented to include a number of al-
 322 ternative or coexistent stressors (Galic et al., 2018). Therefore,
 323 in addition to elucidating consequences of exposure to plas-
 324 tics important for loggerhead turtle conservation, the present
 325 study answers a long-standing call by ecotoxicologists (Kramer
 326 et al., 2011) and conservationists (Cushman, 2006; Mazaris
 327 et al., 2006) for a flexible framework relating environmental
 328 pressures to both individual- and population-level indicators.
 329 Probably the most important feature of the new framework
 330 is the ability to assimilate and make best use of disparate
 331 sources of knowledge ranging from molecular-level processes,
 332 through individual-level data and time series, to environmental
 333 and population-level indicators (Nisbet et al., 2000), taking
 334 us a big step closer to understanding the general ecological
 335 consequences of plastic pollution.

336 Materials and Methods

338 **Overview.** Herein, we showcase a general and flexible modeling
 339 approach aiming to relate the individual-level effects of sub-lethal
 340 exposure to a stressor and the population-level effects of such expo-
 341 sure in order to quantify ecological breaking points. The approach
 342 is general in the sense of pertaining to any wildlife species and
 343 multiple types of stressors. Our focus was on sea turtles and plastic
 344 debris, but we could have easily worked with another endangered or
 345 otherwise-of-interest species whose energy budget is known (Mar-
 346 ques et al., 2018). Similarly, the stressor could have been an abiotic
 347 factor such as temperature or salinity, or anthropogenic substances
 348 such as pollutants or toxicants. The approach is modular: the
 349 outputs of one module serve as the inputs for another module,
 350 while each module’s internal functioning is separate from others.
 351 To describe an individual’s ontogeny we thus used dynamic energy
 352 budgets (Nisbet et al., 2000; Sousa et al., 2008, 2010; Jusup et al.,
 353 2017), although other approaches based on physiological energetics
 354 would do (Nisbet et al., 2012). Similarly, population dynamics is
 355 implementable in any number of ways, e.g., via the Euler-Lotka
 356 equation (De Roos, 2008; Beekman et al., 2019), matrix population
 357 models (Klanjscek et al., 2006; Ijima et al., 2019), physiologically
 358 structured population models (De Roos and Persson, 2002; Diek-
 359 mann et al., 2003), integral projection models (Smallegange et al.,
 360 2017). Finally, we modeled the effects of debris ingestion in the
 361 form of competition between food and plastics for available digestive
 362 capacity. We assumed that ingested debris occupied a fraction of
 363 digestive capacity, but delivered zero digestible energy. Below, each
 364 module is conceptually introduced; mathematical derivations and
 365 other details are in the SI Appendix, Methods.

366 **Ontogeny.** We relied on physiological energetics to generate the
 367 ontogenetic development of individuals as a function of exposure to
 368 external factors (food, temperature, stressors, etc.). Physiological
 369 energetics contrasts energy sources and sinks, and thus determines
 370 the proportion of ingested energy available for growth, maturation,
 371 and reproduction. We used the standard dynamic energy budget
 372 model because of an existing and detailed adaptation to loggerhead
 373 turtles (Marn et al., 2017b,a, 2019), and straightforward application

to more than 2000 other species* (Marques et al., 2018).

The model expresses body size, L , as a function of age, a , via

$$\frac{dL}{da} = \dot{G}(i\text{-state}, e\text{-state}), \quad [1]$$

where $\dot{G} = \dot{G}(i\text{-state}, e\text{-state})$ is a growth function dependent on the
 individual’s state, e.g., body size L (i -state variables), and the state
 of the environment, e.g., the aforementioned abiotic factors and
 anthropogenic substances (e -state variables). A concrete functional
 form of \dot{G} is decided by the animal’s energy allocation scheme or
 simply the energy budget, the side-product of which is also a fecun-
 dity function, $\dot{F} = \dot{F}(i\text{-state}, e\text{-state})$. The energy budget is only
 indirectly inferable by specifying model assumptions and testing how
 well the resulting model fits empirical data. The assumptions of the
 standard dynamic energy budget model, as well as the consequent
 energy budget, are detailed in the SI Appendix, Methods. The
 model parameters for North Atlantic loggerhead turtles produce
 realistic predictions with a good fit to observed data (See Marn
 et al. (2017b,a) and the online Add-my-pet collection* for more
 information on model calibration and comparison to observations).
 Ontogenies were simulated at the temperature of 21.8°C and a range
 of effective food abundance (see Fig. 1).

Population growth. Key ingredients for determining the long-term
 population growth rate, \hat{r} , are fecundity \dot{F} and survival S as func-
 tions of age. This is most evident by writing the Euler-Lotka
 equation of population growth

$$1 = \int_0^{\infty} e^{-\hat{r}a} \dot{F}(a) S(a) da. \quad [2]$$

Generally, however, fecundity and survival depend on i -state and
 e -state variables, with body size L often being the dominant vari-
 able that drives major ecological events (De Roos and Persson,
 2002). To calculate the loggerhead population growth, we se-
 lected size as the dominant variable driving major ecological events,
 and obtained the survival curves by integrating size-dependent
 hazard rates found in the literature (Heppell et al., 2003; Sasso
 et al., 2006) (SI Appendix, Fig. S6). The individual-level energy
 budget gives access to environment-dependent fecundity function
 $\dot{F} = \dot{F}(L, e\text{-state}) = \dot{F}(L)$, and by means of Eq. (1), to age-length
 relationship $L = L(e\text{-state}; a) = L(a)$. The composition of func-
 tions $\dot{F} = \dot{F}[L(a)]$ is then directly insertable into Eq. (2) to yield a
 population growth rate sensitive to the specifics of ontogenetic devel-
 opment. Analogous reasoning applies to composition $S = S[L(a)]$
 with the exception that survival is, at best, partly inferable from
 the individual’s energy budget in the form of aging-related hazard
 (van Leeuwen et al., 2010) because natural (diseases and preda-
 tion) or anthropogenic (direct exploitation or collateral damage)
 hazards almost always dominate. We reported population growth
 in terms of $\lambda = \exp(\hat{r}/\hat{r}_0)$, where we set the reference growth rate
 to $\hat{r}_0 = 1 \text{ y}^{-1}$. The population is thus growing if $\lambda > 1$, declining if
 $\lambda < 1$, and stagnating if $\lambda = 1$.

Debris ingestion. We described food abundance in terms of quan-
 tity $0 \leq f < 1$ representing a fraction of the maximum ingestion rate
 for individuals of a given size. The value of f reflects food supply
 in the environment because $f = 0$ when supply ceases, and $f \rightarrow 1$
 when supply is unlimited. Assuming that individuals spend time
 searching for and then clearing food (Jusup et al., 2017), the rela-
 tionship between f and food supply X becomes $f = X/(X + K_X)$,
 where K_X is known as the half-saturation constant for food. This
 constant is a compound parameter containing information on how
 aptly food is searched for and cleared by the animal. Therefore,
 if plastic debris is mistaken for food, some of the searching and
 clearing efforts are in vain, and the half-saturation constant should
 increase in the presence of plastics. We modeled this by assuming
 that a fraction of digestive capacity is occupied by ingested debris
 (SI Appendix, Methods). Instead of f , turtles experience effective
 food abundance f_{eff} , with K_X replaced by $K_{\text{eff}} = K_X(1 + Y/K_Y)$,
 where Y is the supply of plastics in the environment, and K_Y is an
 analogue of the half-saturation constant, but now for plastic debris
 instead of food. Quantity f_{eff} is related to food abundance f via

$$f_{\text{eff}} = \frac{f}{1 + fR\frac{Y}{X}}, \quad [3]$$

*https://www.bio.vu.nl/thb/deb/deblab/add_my_pet

441 where dimensionless R is the gastrointestinal residence time of
 442 plastics relative to food, and Y/X is the environmental plastics-to-
 443 food ratio. Previous work (Marn et al., 2017b) shows that loggerhead
 444 turtles in the wild experience food abundance $f = 0.81$. Ranges of
 445 simulated R and Y/X were determined from passage time of plastics
 446 relative to food (Lutz, 1990; Mascarenhas et al., 2004; Valente et al.,
 447 2008), and environmental debris prevalence (Moore et al., 2001;
 448 Figueiredo and Vianna, 2018) (SI Appendix, Methods). The debris-
 449 to-food ratio implicitly assumes uniform mixing of plastic debris and
 450 food in the ocean, and that the higher likelihood of debris ingestion
 451 due to higher debris occurrence (Schuyler et al., 2016) results in
 452 more debris in the digestive system relative to food.

453 Ecological literature often expresses the influence of plastics
 454 on sea turtles in terms of the percentage of debris in digestive
 455 contents, $\%V_{dc}$. Under the assumption that digestive capacity is
 456 indiscriminately occupied by both, we obtain the simplest relation
 457 between quantities R , X/Y , and $\%V_{dc}$:

$$458 \quad \frac{\%V_{dc}}{1 - \%V_{dc}} = R \frac{Y}{X}. \quad [4]$$

459 The simple form is intuitively appealing because either $Y/X = 0$ or
 460 $R = 0$ leads also to $\%V_{dc} = 0$, whereas either $Y/X \rightarrow \infty$ or $R \rightarrow \infty$
 461 leads to $\%V_{dc} = 1$. In other words, debris in digestive contents
 462 is negligible in a plastics-poor environment or if the clearance of
 463 plastics is much faster than that of food. Conversely, debris over-
 464 whelms gastrointestinal tract in a plastics-rich environment or if the
 465 clearance of plastics is relatively slow. By using the percentage of
 466 debris in digestive contents as the target quantity for simulations,
 467 we bypass the uncertainties linked to preferential plastics ingestion
 468 of specific life stages, as well as different types of plastic debris hav-
 469 ing different residence times. The chances of debris ingestion and
 470 variability of residence time by the plastics types does not matter as
 471 long as relative abundance of ingested plastics remains reasonably
 472 constant. The simulated percentages in stomach contents have been
 473 truth-grounded against necropsy data (Frick et al., 2009; Lynch,
 474 2018); presently, the mean and the maximum value of $\%V_{dc}$ are 3%
 475 and 25%, respectively (Frick et al., 2009).

476 **Data Accessibility.** The data associated with the paper has been
 477 made available via HAL (hal-02880223) and associated code via
 478 GitHub[†] and Add-my-Pet repository[‡].

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† <https://github.com/ninamarn/marineDebrisTurtle>

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690

1 **Supplementary Information for**
2 **Quantifying impacts of plastic debris on marine wildlife**
3 **identifies ecological breakpoints**

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7 **This PDF file includes:**

8 Figs. S1 to S6

9 References for SI reference citations

Supporting Methods

Model equations for ontogenetic development. To simulate realistic ontogenies of individual sea turtles, we relied on physiological energetics, and specifically the standard Dynamic Energy Budget (DEB) model (Sousa et al., 2008, 2010; Kooijman, 2010; Jusup et al., 2017) (Fig. S1). This model traces the individual's state—its size, life stage, and condition (i -state variables)—as a function of age, depending on physiological characteristics that are mirrored in the values of the model parameters. The environmental conditions, such as food abundance and temperature, act as forcing variables (e -state variables).

The rates of change of a sea turtle's i -state variables are given by a set of three differential equations. Eq. (1) for length L (in cm) tracks energy used for growth, Eq. (2) for life-stage indicator E_H (in J) tracks cumulative energy invested into maturation, and Eq. (3) for condition E (in J) tracks the amount of available energy reserve:

$$\frac{dL}{dt} = \frac{\dot{p}_G}{3L^2[E_G]}, \quad [1]$$

$$\frac{dE_H}{dt} = \begin{cases} \dot{p}_R & \text{if } E_H < E_H^p \\ 0 & \text{otherwise} \end{cases}, \text{ and} \quad [2]$$

$$\frac{dE}{dt} = \dot{p}_A - \dot{p}_C, \quad [3]$$

where $[E_G]$ (in $J\text{ cm}^{-3}$) is the volume-specific cost for growth, and E_H^p (in J) is threshold maturity for sexual maturation. Length L is not a measurable quantity itself, but connects to often measured straight carapace length L_{SCL} via shape factor δ_{SCL} , i.e., $L = L_{\text{SCL}}\delta_{\text{SCL}}$. Below, we define energy flows \dot{p}_* (in $J\text{ d}^{-1}$) in terms of i -state variables.

A sea turtle acquires energy through assimilation:

$$\dot{p}_A = \{\dot{p}_{Am}\}L^2f, \quad [4]$$

where the performance of the turtle's digestive system is reflected in the surface-specific maximum assimilation rate, $\{\dot{p}_{Am}\}$ (in $J\text{ d}^{-1}\text{ cm}^{-2}$), while food abundance f , $0 \leq f < 1$, is the ratio of the current ingestion rate to the maximum ingestion rate at unlimited food supply. Assimilated energy gets mobilized from reserve to power metabolic processes at a rate determined by the utilization energy flow:

$$\dot{p}_C = [E] \frac{\dot{v}[E_G]L^2 + [\dot{p}_M]L^3}{[E_G] + \kappa[E]}, \quad [5]$$

where $[E] = E/L^3$ is energy reserve density, parameter \dot{v} (in cm d^{-1}) is energy conductance, and dimensionless parameter κ , $0 < \kappa < 1$, is a fraction of utilization flow directed to somatic maintenance and growth. Finally, the sea turtle grows at a rate:

$$\dot{p}_G = [E_G] \frac{\kappa\dot{v}[E]L^2 - [\dot{p}_M]L^3}{[E_G] + \kappa[E]}, \quad [6]$$

provided there is mobilized energy left after satisfying the somatic maintenance needs, i.e., $\kappa\dot{p}_C > [\dot{p}_M]L^3$.

As indicated, fraction $\kappa\dot{p}_C$ of mobilized energy is used for somatic maintenance and growth, while the remaining fraction, $(1 - \kappa)\dot{p}_C$, is allocated to maturation in embryo and juveniles, and to reproduction in adults: $\dot{p}_R = (1 - \kappa)\dot{p}_C - \dot{k}_J E_H$, where \dot{k}_J (in d^{-1}) is the maturity maintenance coefficient. Life-stage transitions from embryo to juvenile, and from juvenile to adult, occur at maturity thresholds E_H^b and E_H^p , respectively. Investment into reproduction starts after puberty (E_H^p) and determines fecundity in adults: $\dot{F} = \kappa_R \dot{p}_R / E_0$, where dimensionless parameter κ_R is the efficiency of energy conversion from reserve to eggs, and E_0 (in J) is the energetic value of an egg. Fig. S2 visualizes the energy allocation of a free-ranging loggerhead turtle experiencing food abundance $f = 0.81$.

In the described frameworks, body mass is not an i -state variable, but rather a function of length L , reserve E , and energy E_R stored for reproduction between two nesting seasons. Quantity E_R is calculated only for adult individuals using $E_R = \int \dot{p}_R dt$, where integration is performed over predominantly biannual remigration period (Hawkes et al., 2005). Body mass is calculated as:

$$W = d_V L^3 + \rho_E (E + E_R), \quad [7]$$

where $d_V \approx 1\text{ g cm}^{-3}$ is body density and ρ_E (in g J^{-1}) is a mass-energy coupler for reserve. The second term in this equation indicates the animal's condition, which is best seen through the connection with Fulton's condition factor (Nash et al., 2006):

$$\text{CF} \propto \frac{W}{L_{\text{SCL}}^3} = d_V \delta_{\text{SCL}}^3 + \rho_E \frac{E + E_R}{L_{\text{SCL}}^3}. \quad [8]$$

The proportionality constant is $C_0 = 100\text{ cm}^3\text{ g}^{-1}$ if wet body mass is expressed in grams and straight carapace length in centimeters.

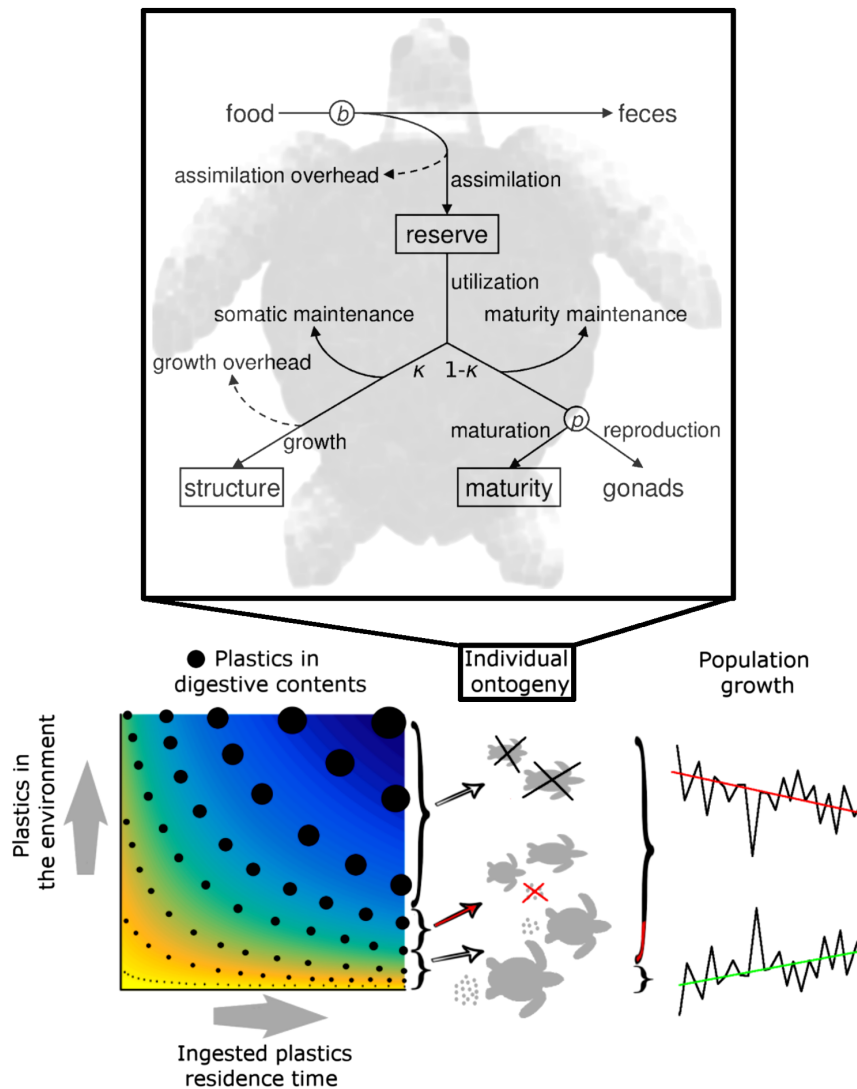


Fig. S1. Schematic representation of the proposed modeling framework. More plastic debris in the environment or a longer ingested-plastics residence time result in more of the digestive capacity being occupied by plastics instead of food. This affects individual ontogeny by reducing the energy assimilated into reserve, and thereafter the energy available for growth and maturation (or reproduction) after all maintenance costs are paid. In response, individuals may grow smaller, their condition may decrease, they may lose the ability to reproduce, and they may even die under the most extreme circumstances. At the population level, moreover, reproducing individuals are no guarantee of positive growth because fecundity may be insufficient to compensate for mortality losses, in which case the population declines. Individual ontogeny depending on the state of the environment and the state of the individual is captured using the standard Dynamic Energy Budget model (box), but with parameter values adjusted to loggerhead turtles.

48 **Model parameters and the generality of physiological energetics.** The values of model parameters pertaining to
 49 the North Atlantic population of loggerhead turtles at the temperature of 21.8 °C are: $[E_G] = 7322 \text{ J cm}^{-3}$, $E_H^b =$
 50 $2.535 \cdot 10^4 \text{ J}$, $E_H^p = 9.875 \cdot 10^7 \text{ J}$, $\{\dot{p}_{Am}\} = 868.11 \text{ J cm}^{-2}$, $\dot{v} = 0.0791 \text{ cm d}^{-1}$, $[\dot{p}_M] = 13.01 \text{ J cm}^{-3}$, $\dot{k}_J = 0.0013 \text{ d}^{-1}$,
 51 $\kappa = 0.729$, $\kappa_R = 0.95$, $\delta_{SCL} = 0.39$, $E_0 = 178.56 \text{ kJ}$, and $\rho_E = 1.552 \cdot 10^4 \text{ g J}^{-1}$ (Marn et al., 2017b). Data from all
 52 life stages used to calibrate the model, as well as the potential updates to model parameters, are freely accessible in
 53 the Add-my-Pet collection* (Marques et al., 2018). A discussion on the model calibration and model predictions
 54 in the context of the reported values for North Atlantic loggerhead turtles, including inter-individual variability in
 55 growth, maturation, and reproduction due to experienced food abundance and temperature, are available in the
 56 literature (Marn et al., 2017b,a). We deliberately decided to work with model parameters pertaining to the North
 57 Atlantic population of loggerhead turtles, but other populations or species from the Add-my-Pet collection, e.g., the
 58 Mediterranean population of loggerhead turtles (Marn et al., 2019), green turtles (Stubbs et al., 2019), or any other
 59 species exposed to plastics ingestion would have been equally acceptable. Testifying to the generality of the approach
 60 is the fact that the Add-my-Pet collection currently contains over 2000 entries.

*https://www.bio.vu.nl/thb/deb/deblab/add_my_pet

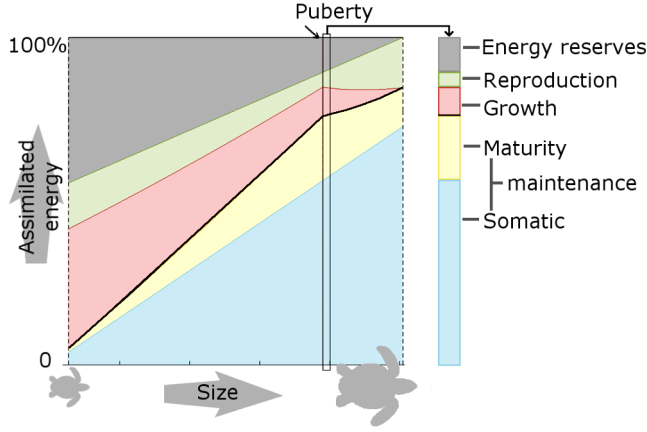


Fig. S2. Energy budget of a loggerhead turtle throughout its ontogeny. Shown is the allocation of assimilated energy to main ontogenetic processes: maintenance, growth, maturation or reproduction, and reserve buildup. Further emphasized is a snapshot of the energy budget at sexual maturation. Maintenance demands are size-driven, leaving large turtles with less energy for growing and accumulating reserve. Investment into maturation halts when turtles sexually mature. Instead, investment into reproduction begins. Here, food abundance is set to $f = 0.81$, corresponding to what free-ranging loggerhead turtles experience in the wild.

61 **Plastic ingestion and effective food abundance.** Food abundance f introduced in Eq. (4) depends on food density
 62 in the environment, X (in g dm^{-3}), via the Holling type-II functional response (Holling, 1959):

$$f = \frac{X}{X + K_X}. \quad [9]$$

63 Here, K_X (in g dm^{-3}) is the half-saturation constant for food. Relatively small K_X characterizes animals with low
 64 ingestion demands and/or fast foraging rate, and such animals reach near-satiation at low food densities. The opposite
 65 holds for relatively large K_X .

66 In the presence of debris in the environment, the animal's effective half-saturation constant K_{eff} becomes larger
 67 than K_X . Specifically, if Y (in g dm^{-3}) denotes environmental debris density, then the relationship between K_{eff} and
 68 K_X is:

$$K_{\text{eff}} = K_X \left(1 + \frac{Y}{K_Y} \right), \quad [10]$$

69 where K_Y is a debris analogue of the half-saturation constant. The animal thus perceives food abundance $f_{\text{eff}} =$
 70 $X / (X + K_{\text{eff}})$ that is lower than f whenever $Y > 0$. Below, we derive Eq. (10) via the competition of food and
 71 plastics for the limited digestive capacity of gastrointestinal tract.

72 **Digestive capacity.** Inspired by the concept of Synthesizing Units (Kooijman, 1998, 2006), we modeled energy uptake
 73 by assuming that ingested food and debris compete for the limited digestive capacity of the gastrointestinal tract.
 74 Parts of the capacity are either: (i) occupied by food, (ii) occupied by plastics, or (iii) free (unoccupied), which we
 75 respectively denote with fractions θ_X , θ_Y , and θ_- . By definition $\theta_X + \theta_Y + \theta_- = 1$. The dynamics of digestive capacity
 76 is determined by a binding rate (\dot{b}_i in $\text{dm}^3 \text{g}^{-1} \text{d}^{-1}$), and a release rate (\dot{k}_i in d^{-1}), where $i = X$ for food or $i = Y$ for
 77 debris. The ratio of the two rates gives the half-saturation constant: $K_i = \dot{k}_i / \dot{b}_i$. The dynamics of the three fractions
 78 can then be written as a set of three ordinary differential equations (ODEs):

Dynamics	Eq.	Equilibrium densities	Eq.
$\frac{d\theta_X}{dt} = \dot{b}_X \theta_- X - \dot{k}_X \theta_X$	[d1]	$\theta_X^* = \theta_-^* \frac{X}{K_X}$	[e1]
$\frac{d\theta_Y}{dt} = \dot{b}_Y \theta_- Y - \dot{k}_Y \theta_Y$	[d2]	$\theta_Y^* = \theta_-^* \frac{Y}{K_Y}$	[e2]
$\frac{d\theta_-}{dt} = -\dot{b}_X \theta_- X + \dot{k}_X \theta_X - \dot{b}_Y \theta_- Y + \dot{k}_Y \theta_Y$	[d3]	$\theta_-^* = \left(1 + \frac{X}{K_X} + \frac{Y}{K_Y} \right)^{-1}$	[e3]

80 The equilibrium digestive capacity fractions are marked with an asterisk.

81 Only the occupied part of the digestive capacity is responsible for energy assimilation. Analogously to Eq. (4),
 82 we have: $\dot{p}_A = \{\dot{p}_{Am}\} L^2 (\theta_X^* + \kappa_Y \theta_Y^*)$, where κ_Y , $0 \leq \kappa_Y < 1$, quantifies the energetic contribution of debris (Y)
 83 relative food (X). The effective food abundance is $f_{\text{eff}} = \theta_X^* + \kappa_Y \theta_Y^*$. Assuming debris has no energetic value, $\kappa_Y = 0$.

84 Inserting Eq. (e3) into Eq. (e1) finally yields:

$$f_{\text{eff}} = \theta_X^* = \frac{X}{X + K_X \left(1 + \frac{Y}{K_Y}\right)}, \quad [11]$$

85 from where it becomes obvious that $K_{\text{eff}} = K_X (1 + Y/K_Y)$ as specified in Eq. (10).

86 Comparing Eq. (9) and Eq. (11) leads to $f_{\text{eff}} = f / \left(1 + f \frac{K_X}{K_Y} \frac{Y}{X}\right)$. Here, Y/X is an environmental debris-to-food
 87 ratio. The ratio of half-saturation constants, K_X/K_Y , is best understood as follows. Because digestive capacity is
 88 indiscriminately occupied by food and debris, we have $\dot{b}_X = \dot{b}_Y$. We also have $\dot{k}_X > \dot{k}_Y$ because plastics is more inert
 89 than food. Accordingly, quantity $R = \frac{K_X}{K_Y} = \frac{\dot{k}_X}{\dot{k}_Y} > 1$ is a ratio of clearance rates for food and plastics that shows how
 90 much longer plastics occupy digestive capacity relative to food. Existing literature, in line with inequality $R > 1$,
 91 posits that it takes more time to egest plastics than digested food (Lutz, 1990; Mascarenhas et al., 2004; Valente
 92 et al., 2008). We finally arrive at an expression that relates food abundance f in the absence of plastics to effective
 93 food abundance f_{eff} in the presence of plastics:

$$f_{\text{eff}} = \frac{f}{1 + fR\frac{Y}{X}}. \quad [12]$$

94 **Plastics in digestive contents.** We have shown in Eq. (12) that effective food abundance in the presence of plastics
 95 is critically determined by two dimensionless factors: (i) the environmental debris-to-food ratio, Y/X , and (ii) the
 96 residence time of debris relative to food, R . The former factor quantifies the prevalence of debris in the environment,
 97 while the latter factor quantifies how much longer debris stays in the gastrointestinal tract relative to food. Although
 98 some indications of environmental debris prevalence and the relative debris residence time exist in the literature (Lutz,
 99 1990; Moore et al., 2001; Mascarenhas et al., 2004; Valente et al., 2008; Barnes et al., 2009; Figueiredo and Vianna,
 100 2018), field ecologists often measure the proportion of debris in digestive contents, $\%V_{\text{dc}}$ (Frick et al., 2009; Clukey
 101 et al., 2017; Lynch, 2018). To use available data in a consistent way, we need to establish a relationship between
 102 quantities Y/X , R , and $\%V_{\text{dc}}$, while satisfying several constraints. First, if $Y \ll X$ then the presence of plastics is
 103 negligible and $\%V_{\text{dc}} \rightarrow 0$, whereas if $Y \gg X$, the presence of plastics is overwhelming and $\%V_{\text{dc}} \rightarrow 1$. Similarly, if
 104 $R \ll 1$ then plastics is quickly cleared from the gastrointestinal tract and $\%V_{\text{dc}} \rightarrow 0$, whereas if $R \gg 1$ then plastics
 105 bioaccumulate in the gastrointestinal tract and $\%V_{\text{dc}} \rightarrow 1$. The simplest relationship that satisfies these constraints is
 106 (Fig. S3):

$$\frac{\%V_{\text{dc}}}{1 - \%V_{\text{dc}}} = R \frac{Y}{X}. \quad [13]$$

107 Based on the known residence times of food from 9 to 13 days (Valente et al., 2008), and of debris from several
 108 weeks to several months (Lutz, 1990; Mascarenhas et al., 2004), we simulated relative debris residence times as
 109 the ratio of debris and food residence times in the range between $1 \leq R \leq 10$. The values of the environmental
 110 debris-to-food ratio are unknown at present for sea turtles, but in some areas more plastic particles than plankton
 111 have been observed (Moore et al., 2001) implying $Y/X > 1$; in simulations we always varied Y/X sufficiently to reach
 112 or even exceed the point at which individuals are unable to sexually mature. The different combinations of R and
 113 Y/X result in a range of $\%V_{\text{dc}}$ values (Fig. S3) and thus effective food abundances illustrated in Fig. 1 of the main
 114 text.

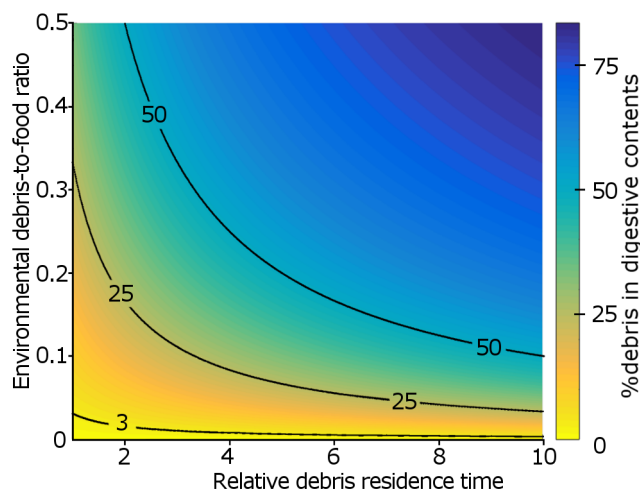


Fig. S3. Environmental debris-to-food ratio and relative debris residence time control the proportion of debris in digestive contents. Shown is the percentage debris in digestive contents, $\%V_{dc}$ (colormap), as a function of debris-to-food ratio in the environment, Y/X (y-axis), and the residence time of food relative to ingested debris, R (x-axis). The representative values of Y/X are currently unknown. We estimated the representative values of R , $1.25 \leq R \leq 10$, based on the known residence times of food and debris in gastrointestinal tract of loggerheads (Lutz, 1990; Mascarenhas et al., 2004; Valente et al., 2008). The displayed range of $\%V_{dc}$ values between $0 \leq \%V_{dc} < 84\%$ is illustrative and exceeds the range used in simulations. Of note is that existing data on plastics loads are plagued by uncertainties. For North Atlantic loggerhead turtles, the mean and the maximum reported percentage of plastics in stomach contents equals 3.2% and 25.7%, respectively (Frick et al., 2009), yet the stomach is unlikely to be representative of the whole gastrointestinal tract in which ingested debris may bioaccumulate (Tomás et al., 2002; Campani et al., 2013). Measuring (i) debris in digestive contents and (ii) relative residence times, and then inverting the map shown here, may quantitatively constrain the presently unknown bio-availability of plastics to sea turtles.

Supplementary Results

Supplementary Result 1. Turtles exposed to plastics are expected to have lower condition than comparably sized unexposed turtles. Based on Eq. (8), for example, we predict a North Atlantic loggerhead juvenile of 40 cm straight carapace length at effective food abundance of $f_{\text{eff}} = 0.64$ to have condition factor of $CF \approx 12$. A similarly sized juvenile at food abundance of $f = 0.81$ for unexposed loggerheads is expected to have a condition factor of $CF \approx 14$. This difference in condition factors corresponds to $\approx 11\%$ less body mass for exposed turtles relative to unexposed ones, which in turn corresponds to $\approx 19\%$ less reserve because the difference in body mass is entirely due to less accumulated reserve. Available data, albeit limited, seem to support the model's prediction (Fig. S4). Specifically, we obtained from the Pula aquarium in Croatia sufficient information to estimate the condition factor of 77 loggerhead turtles, of which 8 individuals have been found to have ingested plastics. The average condition of exposed turtles is indeed lower by close to the predicted amount. However, more data is needed to establish statistical significance with sufficient power; the small number of exposed turtles may cause type II error with a large probability.

Supplementary Result 2. In the main text, we simplified simulations by assuming a constant exposure to plastics. A more realistic scenario is that sea turtles experience a time-varying environment such that the periods of no exposure interchange with the periods of potentially high exposure. To explore such a scenario, we performed additional simulations in which the proportion of debris in digestive contents, $\%V_{dc}$, is log-normally distributed with parameters $\mu = 0.6$ and $\sigma = 1.0$ (Fig. S5A). This generates a realistic mean of $\%V_{dc} \approx 3\%$, covers a realistic range of plastics loads between zero and 30%, and mostly the load is low to moderate, with only occasional spikes (Fig. S5B) just as reported in the literature (Frick et al., 2009; Schuyler et al., 2014; Clukey et al., 2017; Lynch, 2018).

Based on the assumed distribution for the proportion of debris in digestive contents, we calculated new effective food abundance experienced by sea turtles every 30 simulation days and performed smooth interpolation for days in between. Spikes in plastics load were large enough to make effective food abundance occasionally insufficient to satisfy the basic maintenance needs, i.e., the animal could enter starvation. The formal condition for starvation is $\kappa \dot{p}_C < [\dot{p}_M] L^3$, from which point the energy deficit is compensated for by draining the energy stored for reproduction (i -state variable E_R). If such energy is unavailable, the animal satisfies only the basic maintenance needs from reserve (i -state variable E). In the latter case, Eq. (5) no longer holds, and instead $\dot{p}_C = [\dot{p}_M] L^3 + \dot{k}_J E_H$. We assumed biannual reproduction (Hawkes et al., 2005) during which variable E_R is emptied until there is not enough energy to produce a whole egg. The average energetic value of an egg calculated by the model equals 178.56 kJ, and is close to that reported in literature (Hays and Speakman (1991)). The predicted reproductive output is expressed for the whole season as, for example, 600 eggs—in reality this would correspond to between 4 and 5 clutches (Tucker, 2010).

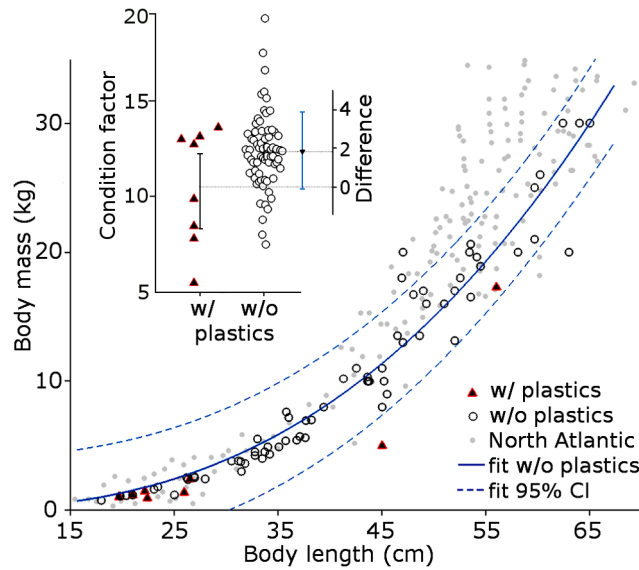


Fig. S4. Debris ingestion lowers condition. The model predicts that similarly sized loggerhead turtles who have been exposed to plastics should have lower mass, and thus lower condition than unexposed turtles. Using data on 77 loggerhead turtles obtained from the Pula aquarium in Croatia, we found preliminary evidence favoring this model prediction. Specifically, the average condition factor of eight exposed turtles is significantly lower than the average condition factor of the remaining 69 turtles (one-sided t-test, test statistic 2.35, p-value $p=0.011$). The empirical difference in condition factors (≈ 2) is in line with the model-predicted difference. The inset shows the estimated condition factors, including the average values and the confidence intervals obtained with the dabest package (Ho et al., 2019). Also shown is the original data in a body-mass vs. body-size plot, where the red triangles depict eight exposed individuals, while the white circles depict 69 other individuals. These turtles have been found in the Adriatic Sea and thus belong to the Mediterranean population. For comparison, gray dots depict similarly sized North Atlantic turtles (Wabnitz and Pauly, 2008).

145 of 120 to 150 eggs per clutch (Tiwari and Bjorndal, 2000; Hawkes et al., 2005).

146 Due to the stochasticity of plastics loads, the results vary between simulation runs. Here, we illustrate one typical
 147 realization. The decrease in body size of turtles exposed to plastics, who experience effective food abundance f_{eff} ,
 148 compared to unexposed turtles, who experience food abundance $f = 0.81$, is small (Fig. S5C). The small decrease in
 149 body size is explained by the average effective food abundance of $f_{\text{eff}} = 0.79$ being close to $f = 0.81$. Favorable feeding
 150 periods allow sea turtles to grow relatively large. In contrast, intermittent periods of high plastics loads worsen the
 151 condition of exposed turtles throughout their lifetime (Fig. S5D). With less energy available to accumulate reserve,
 152 the reproductive output declines (Fig. S5E). These results are worrying because body size in terms of carapace length
 153 is the easiest to measure in practice, and thus much more often reported than body mass and fecundity, although a
 154 combination of at least two of these quantities would be better indicators of the true state in the field.

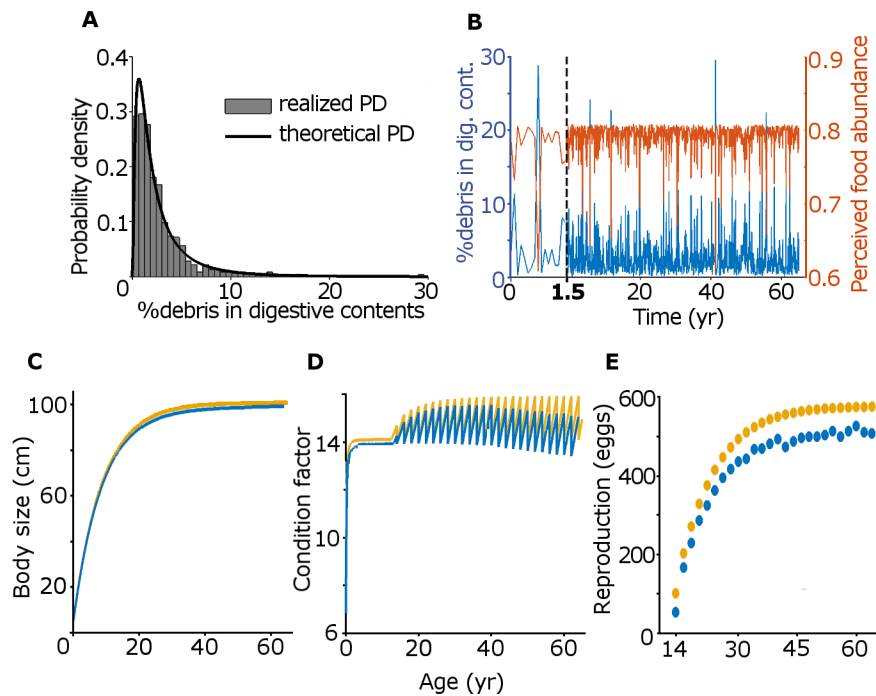


Fig. S5. Physiological response of loggerhead turtles to time-varying plastic exposure. **A**, We assumed that plastic load was log-normally distributed (PD - probability distribution) with parameters $\mu = 0.6$ and $\sigma = 1.0$. This produced the mean of $\%V_{dc} \approx 3\%$, with most values being relatively small, but occasional large spikes remained possible. **B**, As a consequence of the assumed probability distribution for $\%V_{dc}$, the average effective food abundance in the shown simulation was $f_{eff} = 0.79$. This compares to $f = 0.81$ for turtles unexposed to plastics. **C**, Because the average effective food abundance for exposed turtles was close to that for unexposed turtles, the ultimate body size of 99.3 cm straight carapace length in the scenario with time-varying exposure is close to 100.8 cm straight carapace length in the no-exposure scenario. **D**, As in the main text, plastic ingestion affects body mass and the corresponding condition factor more strongly than body size. The displayed difference in condition factor arises because turtles exposed to plastics accumulate less reserve. Oscillations in the adult stage are due to reserve accumulation for reproduction. During a nesting season this energy is converted into eggs. Nesting seasons occur biannually [Hawkes et al. \(2005\)](#). **E**, Reproductive output of exposed turtles is also notably lower than that of unexposed turtles. The difference varies between any two nesting seasons depending on the realized plastic load.

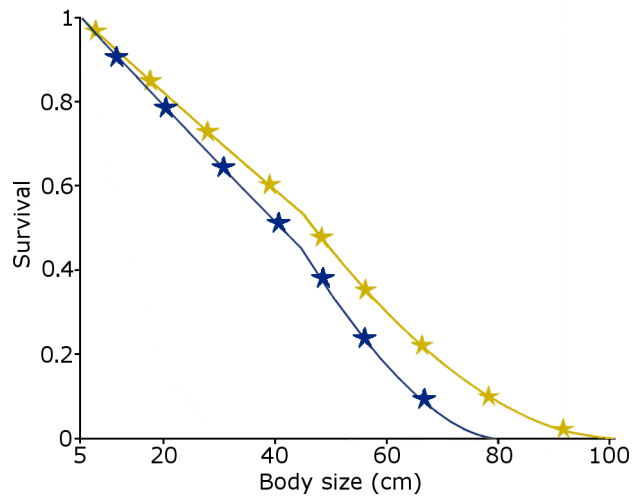


Fig. S6. Survival curves for North Atlantic loggerhead turtles. The curves show the probability of survival from size at hatching to size indicated on the x-axis. We obtained the curves by integrating size-dependent hazard rates found in the literature (Heppell et al., 2003; Sasso et al., 2006). The yellow curve is for unexposed turtles at food abundance $f = 0.81$, whereas the blue curve is for turtles exposed to plastics at effective food abundance $f = 0.64$. We focus on North Atlantic loggerhead turtles, but survival estimates for Mediterranean loggerheads are also available (Casale and Heppell, 2016).

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