Review


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Linking effects of anthropogenic debris to ecological impacts

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Accelerated contamination of habitats with debris has caused increased effort to determine ecological impacts. Strikingly, most work on organisms focuses on sublethal responses to plastic debris. This is controversial because (i) researchers have ignored medical insights about the mechanisms that link effects of debris across lower levels of biological organization to disease and mortality, and (ii) debris is considered non-hazardous by policy-makers, possibly because individuals can be injured or removed from populations and assemblages without ecological impacts. We reviewed the mechanisms that link effects of debris across lower levels of biological organization to assemblages and populations. Using plastic, we show microplastics reduce the ‘health’, feeding, growth and survival of ecosystem engineers. Larger debris alters assemblages because fishing-gear and tyres kill animals and damage habitat-forming plants, and because floating bottles facilitate recruitment and survival of novel taxa. Where ecological linkages are not known, we show how to establish hypothetical links by synthesizing studies to assess the likelihood of impacts. We also consider how population models examine ecological linkages and guide management of ecological impacts. We show that by focusing on linkages to ecological impacts rather than the presence of debris and its sublethal impacts, we could reduce threats posed by debris.

1. Background

Anthropogenic debris (hereafter ‘debris’) contaminates aquatic and terrestrial habitats, degrading most levels of biological organization, but mechanisms linking effects at lower levels (e.g. biochemistry, tissues, organisms) to ecological levels of organization (i.e. populations, assemblages) are poorly understood. Recent analysis of studies about impacts of marine debris [1] revealed more than 80% of studies demonstrated impacts at one or several levels of biological organization (atom–ecosystem). Less than 1% of studies, however, demonstrated conclusive ecological impacts in nature because (i) of difficulties in complex biological systems in demonstrating unambiguously effects were due to debris and (ii) many studies inferred impacts from the presence of debris. Thus, there is confusion between ‘contamination’ of organisms/habitats by debris and assumptions that it causes an impact.

Debris spans a range of materials and sizes, and is often categorized as micro-debris (less than 1 mm) or macrodebris (greater than 1 mm). The latter includes abandoned fishing nets, which can be kilometres in length. Most microdebris considered in the literature is plastic of various forms. Although the majority of the suborganismal effects are from plastics, debris also consists of wood, metal,
rubber, glass and other types of material. Most impacts described at biological scales from individuals to populations occur in the macrobenthos [1].

Contamination of habitats and organisms by debris does not automatically cause biological impacts. Impacts occur when exposure of organisms to debris causes a disturbance of sufficient magnitude to overcome inertia in that level of biological organization, and therefore elicits a response. Like most pollutants [2], most work on debris focuses on toxic effects at lower levels of organization, without investigating ecological effects to populations or assemblages [1]. This is unfortunate, because ecological assessments provide us with critical information about the capacity for debris to alter ecological structures (e.g. assemblages, food-webs) and processes (e.g. competition, predation), including provision of societal services (e.g. maintenance of water quality, biogeochemical cycling, production of uncontaminated food). It also makes it impossible to determine whether management is needed to alleviate impacts.

This issue is not unique to debris; there has been much debate about how best to quantify impacts of contaminants at all biological scales. Sometimes, proposed methods are compared in large-scale field experiments [3], because the best evidence for impacts is experimental [4], and appropriate experimental designs for detecting many types of impacts have long been available [5]. Nevertheless, often, impacts cannot be detected by robust experimental analyses, because there is no information from before a population or assemblage was contaminated to determine the extent of damage, or impacts are too widespread to allow robust comparisons with unaffected areas [6]. Instead, studies involve correlative analyses (e.g. densities of organisms negatively correlated with concentrations of a contaminant in their habitat; see [1]). Such analyses cannot reject the possibility that densities are, themselves, correlated with other environmental features that are positively correlated with contaminants.

Instead, known information about linkages from one level of organization to another can be used to infer impacts [7]. Thus, suppose that it has been demonstrated experimentally that the reduction of enzyme function in a species reduces reproductive capacity of individual animals, and there is robust population modelling to show that reducing numbers of offspring will cause smaller populations. The studies will probably have been done separately for different purposes. Nevertheless, if contaminants are found in areas occupied by this species, then an inference can be made that contamination will lead to loss of reproductive capacity (this first linkage is a generalization from what has been described previously in the literature) and that this will, in turn, lead to a smaller population (a second generalization from previous studies). Although neither linkage can demonstrate that ecological impacts will occur, establishing linkages provides a strong case for inferring impacts to the population. Where such linkages across levels of organization (here, known physiological effects on reproduction and known effects of reproduction on the size of a population) have been established by appropriate study, such inferences can be made with understanding of their likelihood of being correct. This is a common procedure for evaluating environmental pollution, but linkages from one suborganismal level to another are more easily established than are those linking suborganismal levels to impacts at ecological scales [8].

Contamination by debris raises similar issues and problems to those in previous studies about other contaminants leading to pollution and impacts. The issue, in the absence of direct evidence, is how to establish robust and generalizable linkages from biological to ecological levels of organization. Linking impacts of pollutants from lower to higher levels of suborganismal organization is used to integrate, evaluate, interpret and predict impacts on humans, wildlife [9] and ecological assemblages [10]. The existing paradigm is that biochemical changes at subcellular levels in a target organism precede changes to cells and tissues, which in turn affect physiological functions and, ultimately, populations and assemblages [9].

Yet the mechanisms by which debris causes ecological damage are poorly understood. Studies are required at multiple levels of biological organization that consider what is causing observed patterns of change to populations or assemblages at contaminated sites. To achieve this, work must synthesize and link understanding of toxicological consequences to individuals to ecological consequences for populations and assemblages. Understanding can then guide ecological risk assessment and management of impacts by identifying specific variables within relevant organisms for monitoring programmes (hypothesis-driven) to provide earlier warning of ecological impacts and to indicate progress of contaminated systems towards impact or recovery [6,10].

Here, we identify some processes affected by debris using the ‘adverse outcome pathway’ framework [9]. This provides a means to integrate ecotoxicological and medical evidence about the impacts of debris and their linkages across biological scales. Adverse outcome pathways allow scientists to begin to understand the mechanism and sequence by which causally linked biological impacts lead to an adverse ecotoxicological effect. Through this, we shift the focus from traditional endpoints to developing mechanistic understanding of effects of debris at lower biological scales (where most is known) to lesser-known and more worrisome ecological and policy-relevant effects.

2. Suborganismal impacts by microdebris

We demonstrate what can be achieved using examples of contamination of organisms by plastics because, for some species, it is possible to link effects of microdebris at various biological scales (table 1). We note that most work has been done on effects of plastics on species used as standard test-subjects in laboratory experiments, so the range of species involved is small. Nevertheless, suborganismal impacts of particles of debris can be linked across many levels of organization, enabling well-established frameworks for understanding why the effects occur. Establishing such linkages increases our capacity to use information from the more rapid and earlier developmental stages of a problem to predict what is likely to occur at higher levels of organization.

Obviously, linkages across levels can be established without all the intermediate levels being demonstrated (reviewed by Rochman et al. [11]). For example, freshwater algae suffered oxidative stress and reduced rates of photosynthesis when exposed to polystyrene (20 mm, 0–30 g L⁻¹) [18]. Such reductions in photosynthesis can reduce the growth and survival of algae. Despite the wealth of available information, little has been done to link suborganismal effects of debris to ecological impacts. More effort is needed to determine whether or how many of the observed effects actually translate into ecological responses.
3. Evidence for impacts by macrodebris

In contrast to microdebris, there has been more attention to ecological impacts owing to macrodebris. Nevertheless, in an extensive review of the available refereed literature, there were surprisingly few studies where ecological impacts had been demonstrated, rather than simply stated [1]. Some examples are considered here for four major categories of impact: entanglement, rafting, alteration of habitat and ingestion.

(a) Field evidence for impacts on assemblages

Knowledge about the existence, nature, frequency and magnitude of impacts owing to debris can be substantially enhanced and made less uncertain when direct evidence can be gained from field experiments manipulating debris under controlled conditions.

(i) Entanglement or ‘ghost-fishing’

Entanglement by lost fishing nets or traps (‘ghost-fishing’) is a global problem. Such nets are marine debris, and continue to catch and kill animals after being lost from a fishery. Many studies have examined the effectiveness of lost nets in continuing to catch fish and invertebrates. The majority of these have been in unrefereed literature, but have been reviewed extensively [19,20]. Findings depend on types and sizes of nets, nature of habitat, types of organisms caught and the period that nets remain active. Often, catch rates decline rapidly, even though lost nets can continue to trap animals for long periods [21].

Most studies are lists of numbers and types of species caught in nets, without parallel information about any consequences. As a result, it is difficult to determine whether or not ghost-fishing has consequences to populations. The mortality of animals killed by ghost-fishing must be estimated in relation to the sizes of their populations and to the mortality they experience from active fishing and other causes. Even if a lost net or trap catches and kills hundreds of fish or invertebrates per year after it has been lost, this may be trivial compared with the hundreds of thousands killed in the fishery itself [22].

A further complication is that some crabs killed by ghost-fishing would otherwise have been killed in the fishery, making it possible that ghost-fishing per se made little difference. This is particularly difficult for assessing consequences to populations of rare species, because information is so sparse [19].

Despite gaining good information about lost nets and their effects on individuals, experimental analyses have not often led to accurate estimates of ecological effects. Experimental analyses can, however, provide the information necessary to model rates of catch, and therefore fishing mortality, owing to lost gear [23,24]. Direct experimental evidence of mortality rates can, however, be difficult to obtain when dead animals decompose or are consumed by scavengers, and observed deaths probably underestimate mortality [25].

Estimating mortality of individuals does not demonstrate effects on populations for two reasons. The sizes of populations of organisms caught are not known. Often, strandings are used to obtain data, but these are known, in some cases, not to be a reliable indicator of actual rates of mortality in a population [26,27]. So, even where experimental or other estimates of rates of mortality of individuals are possible, some linkage (usually via population modelling) is also required before ecological impacts to populations could be estimated.

Many studies conclude that there are no impacts to populations owing to entanglement (or to ingesting debris, see below) simply on the grounds that populations are showing no change or are increasing (e.g. fur seals) [28,29]. This makes little sense, because, in the absence of entanglement or ghost-fishing, populations might be increasing or increasing faster.

(ii) Alteration/destruction of habitat

Studies of effects of debris on habitat are often correlational and conclusions can be problematic. For example, Uneputty & Evans [30] found more meiofauna and fewer diatoms in areas with litter embedded in sand on a beach than where

<table>
<thead>
<tr>
<th>levels of organization</th>
<th>linkage</th>
<th>reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>subatomic → macromolecular</td>
<td>nanometre-sized polystyrene added to lungs of rats caused oxidative stress and increased concentrations of protein</td>
<td>Brown et al. [11]</td>
</tr>
<tr>
<td>macromolecular → molecular assemblies</td>
<td>lugworms ingesting micrometre-sized PVC showed increased oxidative stress with fewer antioxidants</td>
<td>Browne et al. [7]</td>
</tr>
<tr>
<td>molecular assemblies → organelles</td>
<td>polymethyl methacrylate in mice damaged DNA resulting in increased numbers of cellular micronuclei</td>
<td>Zhang et al. [12]</td>
</tr>
<tr>
<td>organelles → cells</td>
<td>cells of mice that take up nanometre-sized polystyrene suffer apoptosis, necrosis and reduced proliferation</td>
<td>Fröhlich et al. [13]</td>
</tr>
<tr>
<td>cells → tissues</td>
<td>mussels exposed to micrometre-sized polyethylene produced more granulomas in gut-tissues than is normal</td>
<td>von Moos et al. [14]</td>
</tr>
<tr>
<td>tissues → organs</td>
<td>mice exposed to micrometre-sized polymethyl methacrylate caused their bone to breakdown</td>
<td>Clohisy et al. [15], Pearl et al. [16]</td>
</tr>
<tr>
<td>organs → organism</td>
<td>mice ingesting nanometre-sized zinc suffered vomiting, diarrhoea, blocked guts and mortality</td>
<td>Wang et al. [17]</td>
</tr>
<tr>
<td>organ system → organism</td>
<td>lugworms ingesting micrometre-sized PVC with triclosan reduced their feeding and suffered mortality</td>
<td>Browne et al. [7]</td>
</tr>
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there was no litter. These comparisons do not allow any determination of whether this was due to the litter or whether the litter and the assemblage were being affected by other, small-scale environmental variation. Widmer & Hennemann [31] described a negative correlation across beaches in the numbers of ghost-crabs and amounts of litter. They also noted, however, that amounts of litter were positively and numbers of crabs were negatively associated with human activity across beaches, so no causal effect of litter was identifiable.

Odzilek et al. [32] found a negative correlation across beaches between amounts of litter and hatching success of turtles, which were apparently being eaten by crabs when they became trapped in the litter. It may also be true that the amounts of litter and the survival of turtles are both related to some other features of the different beaches.

Very few studies have experimentally introduced debris to determine the amounts of damage caused to a habitat. Katsanekakis et al. [33] put plastic bottles and glass jars into experimental plots of sediment in coves on the Greek coast. Numbers of species of megabenthos increased, and the debris appeared to provide habitat for attachment of sessile species and sites for deposition of eggs.

Lewis et al. [34] tracked unbuoyed lobster pots, which simulate lost pots from the fishery, and measured the damage they did to areas of coral reef. Each pot damaged areas of 1–4.7 m² of reef and reduced the cover of corals by 14–20%, depending on depth.

In each case, alterations of the composition of subtidal assemblages owing to debris were clearly demonstrated.

Such experimental studies of impacts nevertheless require careful design and interpretation. For example, Aloy et al. [35] compared foraging by snails (Nassarius pullus) on a sandy beach in the Philippines. They added plastic bags to the areas of shore to represent debris and found significantly altered foraging behaviour in areas with the most debris (50% and 75% cover), but no effect of 25% cover. There was, however, much less cover (less than 2%) of debris on the shores, so it is impossible to demonstrate that debris in the amounts found in the field would have any effect on the snails.

As a result of the nature or design of experiments, most studies examined were similar and did not provide unambiguous evidence for effects of debris on populations and assemblages because of alteration of habitat. Only coherent and well-designed experiments using realistic amounts of debris will produce this information.

(iii) Ingestion of debris

There have been numerous descriptions of vertebrates (fish; reptiles—especially turtles; birds; mammals) being found dead with plastic or other debris in their guts. Relatively few studies [36,37] have examined and then demonstrated that death was actually owing to the material ingested. There is rarely any information about how representative such cases are of populations, and therefore no indication of any impact (or its magnitude). For example, the Oslo–Paris Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) uses monitoring of plastic in guts of fulmars as a criterion for determining impacts, and there is a precise protocol for handling the birds and their gut contents [38]. There is, however, no indication of how to ensure representative sampling so that data could be used to determine effects on populations.

We found no cases of studies of ingestion that demonstrated any ecological effects because they were not examined with systematic data. Much greater consideration needs to be given to the nature and types of ecological impacts, and how they might be inferred from data and impacts at other biological scales (see section below on proposing hypothetical links).

(iv) Rafting

Natural floating material, such as macroalgae [39,40], often carries diverse assemblages of algae and animals, which can arrive in new areas, potentially as invasive species. Such rafting is a well-known mechanism for dispersal of species. Debris, particularly plastic [41], also carries organisms across oceans, but is more anthropogenic material has no nutritive value [40], so most animals colonizing debris are filter-feeders. Most pieces of debris are unfouled or are colonized by few species [42], and often those are endemic to the area where the debris was found [43]. Debris usually starts its journey without colonizers, whereas floating natural substrata can have a diverse assemblage prior to drifting offshore.

Anthropogenic material may not show similar patterns of stranding to natural material [41], so there is potentially a greater threat of invasion from rafting on debris than on natural material. Nevertheless, there is limited information about where most stranded debris originated, or its path of drift. The probability of invasion by species colonizing drifting litter as opposed to other surfaces, such as natural material or ships, is not known.

Most studies are simply inventories of amounts of litter and the organisms on it. We found no evidence of rafting actually resulting in introductions of alien species, let alone it resulting in any impact. Animals and plants on stranded litter are frequently found dead [41], but whether they were alive when they reached the shore is not known. If not, there is no possibility of serious impacts. Invasive species that have major effects on assemblages (e.g. the alga Caulerpa taxifolia) are spread by discarded fishing gear rather than by active fishing [44].

Lewis et al. [42] argued strongly that floating litter is likely to move species along the same routes through which they disperse naturally on floating material, although the numbers of individuals dispersing across open oceans on floating material are likely to have increased with increased amounts of floating debris. Such transport is slower than via boating, which is more likely to move species across biogeographic boundaries into novel areas while they are still alive [42].

Linkages between the documented presence of algae and invertebrates on debris floating in the open ocean, their arrival, survival and capability of reproduction in novel habitats, and, thus, their subsequent establishment of viable populations, have not been established. Nor is it known what extra threat rafting on debris might pose above dispersal by fouling on boats and transport in ballast water. Whether or not rafting on debris causes ecological impacts cannot possibly be ascertained until some of these linkages are established.

4. Field evidence for ecological linkages

It is possible to make linkages to ecological impacts from experimental evidence about some components of the affected assemblage. Uhrin & Schelling [45] tethered individual wire crab-pots and tyres in areas of saltmarsh,
keeping areas with no attached debris as controls. They sampled replicate experimental areas for 9 (tyres) and 13 weeks (crab-pots), by which time there had been a sustained increase in the number of stalks of cordgrass, Spartina alterniflora, which is a major habitat-forming species. Clearly, this study demonstrated reductions in cordgrass owing to debris. It did not demonstrate impacts on the assemblage of species in saltmarsh, because no other organisms were sampled, but there is evidence that reductions in cover of saltmarsh plants are strongly linked with and often cause major changes in other species [46,47]. From knowledge of linkages between amounts of cordgrass and the structure of associated assemblages, it is possible to infer that ecological effects are affecting more species than just those plants.

5. Linkages from impacts demonstrated in the laboratory

Assuming experimental doses of debris are comparable with those that organisms encounter in habitats, it may be inferred from laboratory studies that effects of debris on individual animals or plants probably have consequences for the assemblages in which they live. For example, mussels exposed to polystyrene (30 nm; 0–0.3 g l⁻¹) ate less [48]. Von Moos et al. [14] found that mussels ingesting polyethylene (less than 80 μm; 2.5 g l⁻¹) had weaker lysosomal membranes in their digestive glands. Reduced feeding and damage to lysosomes cause reduced growth [49]. Reduced lysosomal function in mussels is therefore linked to later reductions in growth. In this case, there is also correlative field evidence that associated assemblages are less diverse in mussel-beds where individuals have smaller scopes for growth [50].

As another example, Triclosan ingested from PVC (230 μm) when present at 5% in sediment increased (55% greater) mortality of lugworms compared with controls, but, importantly, it also diminished the ability of lugworms to burrow and feed [7]. This has potential links to assemblages, because experimental reductions in density and feeding by lugworms increased quantities of silt and algae in sediments, thus altering associated assemblages [51,52].

6. What to do when ecological linkages are not known

There are many problems in establishing linkages from suborganismal impacts or from impacts to individual organisms to predict or identify any effects of debris at ecological levels of populations or assemblages. This has long been an issue for investigations or management of ecological problems caused by chemical contaminants [3]. It is often more expedient not to try to find linkages directly, but to turn to other methodologies for predicting ecological effects or determining how to manage their consequences in the absence of adequate evidence.

(a) Proposing hypothetical links

These will be demonstrated using examples from birds, because there is a large amount of data on these species [53–66]. Observational, correlative and experimental studies have shown impacts of debris on seabirds, affecting tissues, organs and individuals (figure 1). These show that ingested plastic can directly damage the digestive system of birds through ulcers, lesions, perforations and blockages [55]. Correlative evidence shows that birds ingesting larger volumes of plastic weigh less [53,54], grow at slower rates and die sooner [55]. Birds with more plastic in their gut are likely to hold less water and food [67], and contain more chemical pollutants [57,58], some of which are likely (based on studies of other taxa) to be transferred to tissues and suppress immunity [7].

As a result of such observations, concerns are growing about the possible role of plastic as a source of mortality for birds. To guide future research, thought has been put into the sorts of pathways or linkages that could demonstrate how the observed effects on feeding and growth might lead to effects on reproduction or survival, and therefore influence populations. Figure 1 shows a set of inter-related linkages from digestion through growth to reproduction to survival and abundance. These are all realistic potential linkages to loss of individuals or changes to sizes of populations, but they must remain speculative until experimental work is done to test them.

Experiments testing relevant hypotheses about impacts of debris [56] are necessary, because observations and correlative evidence are insufficient to demonstrate causality [53] and the relative importance of direct and indirect effects. Careful thought about possible linkages, as shown here for seabirds, and estimates of how different processes may be disrupted by debris, will identify the experiments needed and determine how likely they are to provide adequate evidence to identify an impact if it is occurring (i.e. their power). There is also experimental evidence (not included here, for brevity) indicating the likelihood of similar linkages for other types of vertebrates (e.g. rodents, goats, turtles, fish) and invertebrates (e.g. crustaceans, molluscs, worms).

(b) Linkages using population models

Several methods (e.g. uncertainty/safety factors [68], species–sensitivity distributions [69] and quantitative structure–activity relationships [70]) have attempted to link effects of pollutants on individuals to their populations, but modelling the dynamics of populations remains the most effective approach [71]. Information on individual responses of organisms to contaminants cannot assess the risks of impacts to populations, because responses vary among individuals [72]. Some individuals can be removed from populations without subsequent effects on the size or growth of populations, because, for example, (i) many individuals would not have reproduced successfully had they stayed alive [73] and/or (ii) sexually mature females contribute more to the population than do juveniles [74]. The size and rate of growth of a population are more ecologically relevant than isolated responses of a subset of organisms, because the former integrates several responses (e.g. mortality rates of juveniles and adults, age at first breeding, number of offspring produced at each breeding attempt and the interval between breeding).

Policy-makers are therefore requesting that populations (and assemblages), rather than just individual organisms, should be protected [75]. Population models provide a means to assess (i) whether or not a population is declining (assessment), (ii) the cause of decline, if occurring (diagnosis), (iii) the parts of the lifecycle requiring managerial action (prescription), and (iv) the likely fate of the population (prognosis) [76]. Such models are based on equations, matrices,
individuals or agents, or integrals, each of which can incorporate scientific understanding about how various abiotic and biotic processes/stressors affect the rate or probability at which individual organisms grow, survive and reproduce, and therefore the resultant dynamics of populations. The requisite information comes from surveys [77] or experiments [78], and is used to build statistical models appropriate for the organism. Models vary in the numbers and types of parameters that need to be estimated, the temporal scales included, and the degree of heterogeneity of the individuals in the population and in their habitat. Estimates of parameters must be critically evaluated in terms of (i) how well they apply to the population being modelled, and (ii) the sample sizes and precision/accuracy of estimates [77]. Data from rigorous experimental studies are the most valuable in determining the relative importance of different stressors that affect a population [77]. Impacts of pollutants on organisms can be assessed only if we understand their life histories, availability of food, density [78], prevalence of parasites, risk of predation [79] and previous exposure to pollutants. The best models for populations affected by pollutants are parameterized for a particular habitat and include actual levels of exposure to the pollutant [80]. Debris such as plastic has a 'life' in the sense that they fragment and do not simply disappear, but have continued effects on target populations (see models for pesticides by Wennergren & Stark [80]), so modelling will be complex.

Sibly [77] outlined a framework to integrate field and laboratory studies using (i) field surveys to identify species that were declining, (ii) controlled experiments with populations in the absence of predation, parasites, competition and migration, and (iii) life-table experiments to identify the life-history stages at which a stressor can cause a decline in the population. Where the stage of life history that is affected can be identified, the molecular, cellular or physiological processes of damage or defence can be investigated in detail. This framework can develop convincing and reliable models for populations exposed to debris. The models could be used to estimate how much change to a population may occur under different scenarios of interaction with debris.

(c) Linkages and management of ecological impacts

Linkages along the life cycle of the debris would, itself, sometimes assist the management of smaller-scale impacts. Consider the potential environmental damage owing to expanded polystyrene. Large pieces may cause visual pollution and may act as habitat, or promote rafting by species. When the polystyrene breaks up, the problems change when animals start to inhale or ingest it. In this case, unwarranted amounts of large material may be managed by removal before the material becomes transmissible to suborganismal levels of organization.
In the USA, populations of some taxa are managed under the Marine Mammal Protection Act and the Endangered Species Act [76]. Other countries have similar legislation to protect particular taxa [81], but the majority of marine taxa are not protected, except by declaration of particular habitats in reserves [82]. Such declarations cannot protect biota from such a diffuse and widespread problem as marine debris [83].

Under the US Marine Mammal Protection Act, models are incorporated as a precautionary measure to manage mortality or serious injury of mammals caused by fisheries [75], using the concept of ‘potential biological removal’ (PBR) [83]. Other countries/jurisdictions use similar calculations with slightly different levels of tolerance [81]. If annual numbers of deaths caused by humans exceed this value, an assessment, diagnosis, prescription and/or prognosis is made [76] to guide the response and future activity. The goal is to reduce anthropogenic mortality to ‘insignificant levels approaching zero’ [83], usually within 5 years. PBR assumes that unbiased information is available on the rate of bycatch, so was never designed to deal with unmonitored fisheries, nor cases where anthropogenic mortality is discovered opportunistically. Managers and scientists are modifying this approach to estimate the number of cetaceans killed by oil spills or ship-strikes (e.g. [84]), but it has not yet been used for debris. Models must also be developed to synthesize pathological data that might link debris to mortality [85].

The US Endangered Species Act is used to determine whether there is evidence that a population is in danger of extinction throughout all, or a large portion, of its range. Gerber & DeMaster [86] developed criteria for delisting a species and defined thresholds for the abundance of a species based upon whether it is threatened, endangered or likely to become extinct. Although such criteria are useful because they incorporate uncertainty, Caswell [76] warned that there were arbitrary elements in the modelling.

Regardless of such problems, using these approaches to estimate the likelihood and potential magnitudes of impacts owing to debris is a useful precautionary action until better information is available.

7. Conclusion
By use of experiments, contamination of habitats and species by plastic debris has been demonstrated to cause ecological impacts to assemblages. In some cases, of course, experiments may be impractical for endangered species (which would invalidate their use in ecological programmes of monitoring). Using other species might be possible, but only if it can be shown that the responses match those of the endangered species. In other cases, ecological responses may only be evident over long periods or over extensive areas.

For microplastic, lethal impacts on animals and plants (from atoms to organisms) and sublethal impacts in humans (atoms to organs) can sometimes be linked through known mechanisms of disease involving oxidative stress, inflammation, necrosis, fibrosis (table 1) and photosynthetic inhibition. Debris may, therefore, have diverse and complex impacts on wildlife and humans by degrading molecular, cellular, physiological and, ultimately, ecological processes. We therefore urge ecologists who wish to understand how and why populations and assemblages are responding to debris to synthesize evidence that links suborganismal, organismal and ecological impacts. This would identify gaps in understanding to guide the design of surveys, population models and experiments. Such linkages can be explored only by integrating evidence across biological scales, but this would provide the understanding to guide assessments of risks and responses to ecological impacts of debris.

As an example, our case study (figure 1) of development of hypotheses about potential impacts to birds suggested that the US government might have underestimated the impacts of debris on populations of albatrosses [67]. We found a number of untested direct (gastric blockage and damage, uptake of pollutants or pathogenic bacteria) and indirect (dehydration, malnutrition, parasites, altered immunity) mechanisms by which ingested plastic could increase rates of mortality and reduce reproductive output [53–66]. So, there is a clear need to determine the magnitude and extent of the potential impacts to individuals in order to understand the actual and potential consequences for populations of birds. If experimental evidence is lacking, then it is precautionary for managers to take a ‘weight of evidence’ approach and act as though debris may pose a threat to populations and assemblages until studies are completed to discount that hypothesis.

More attention to the processes that link suborganismal impacts to ecological responses can also guide population modelling. Models must, however, be constructed to determine whether populations are declining because of debris and which part(s) of the life cycle are being affected. Better models will identify what sorts of management, and at what life stage, could reduce exposure of the organisms to debris, or could mitigate impacts caused by the debris.

A further consideration about establishing linkages is that many methods for quantifying the microdebris are semi- or non-quantitative and cannot be used to determine quantities in tissues or whole organisms, or to assess the likelihood and extent of impacts to habitats. This makes it difficult to determine the ecological risks. Spatial and temporal patterns of presence and amounts of debris are poorly understood, and very little is known about how frequently organisms and habitats are exposed to debris in nature. The situation would be improved if studies of exposure to, or impacts from, debris included estimates of how much debris (including material type, size dimensions, volume, mass) is encountered by organisms in different habitats. Without this information, risk assessments cannot be used and policy-makers will be managing debris using existing laws [87]. The ultimate goal of policies should be to replace problematic products with safer alternatives (before they are used) by tasking ecologists and engineers with working together to identify and remove features of products that (if found as debris in habitats) might cause ecological impacts. Similar approaches are already used to engineer infrastructure ecologically [88] or to make less toxic ‘biocompatible’ medical devices [89].

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