PHILOSOPHICAL TRANSACTIONS B

Phil. Trans. R. Soc. B.
doi:10.1098/not yet assigned

Global change, parasite transmission and disease control: lessons from ecology

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Keywords: Infectious disease; climate change; stressors; resilience; sustainable control

Summary

Parasitic infections are the norm in wildlife, livestock and human populations, and healthy ecosystems tend to be rich in parasites. Yet, their negative impacts can be extreme. Understanding how both anticipated and cryptic 'systems change' might affect parasite transmission at an individual, local and global level, both directly and indirectly, is critical for sustainable control. Here we highlight and synthesise evidence regarding potential effects of global change on parasite transmission in natural host-parasite systems, which could inform more refined and sustainable parasite control programmes in domestic animals or humans. Many examples from diverse terrestrial and aquatic ecological systems show how abiotic and biotic factors can interact additively, multiplicatively or antagonistically to modify effects of global change on parasite transmission, including through altered habitat structure, biodiversity, host demographics and evolution. Despite this, few studies of managed systems explicitly consider higher-order interactions, or the effects of parasite evolution, which might either conceal or exaggerate

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measured impacts of control actions. We call for a more integrated approach to investigating transmission dynamics, which recognizes these complexities and makes use of new technologies for data capture and monitoring, and to support robust predictions of altered parasite dynamics in a rapidly changing world.

Introduction

The current epoch of ecological time is driven by human interference [1]. Multiple anthropogenic stressors – including climate change, pollution, ocean acidification, habitat loss and fragmentation, urbanization, agricultural expansion and intensification, together with other changes in the use of water and land resources – are directly or indirectly impacting all species on earth (e.g. [2–5]). These changes may lead to the crossing or corrosion of critical thresholds, or 'planetary boundaries' [6,7], that induce physiological stress or complete system dysfunction, with negative consequences for individuals, populations and species. Such processes will have significant impacts on parasite natural history and infectious disease risks.

Substantial changes to human demography over the coming decades will have major impacts on land-use practices, including farming, and the natural environment. By 2050 it is estimated that almost half the world's population will live in the tropics, and approximately 66% are likely to be living in urban contexts [8,9]. Millions of individuals are also expected to migrate during their lifetimes due to factors associated with the urban-rural cycle, extreme weather events, economic necessity, water and food security, and conflict [10]. Increased patchiness of wealth associated with urbanization combined with disrupted social structures has already changed the entire landscape of Neglected 'Tropical' Diseases (NTDs). These diseases are no longer exclusively prevalent in less developed countries; instead they infiltrate impoverished areas of all countries, including those in the G20, giving rise to the global pattern of 'Blue Marble Health' [11]. Taken together, these observations and projections give a strong signal to all epidemiologists: the future is both uncertain and rapidly changing, representing a new era of health challenges in the 21st Century that is unprecedented in human history. To keep pace, we need to develop a predictive understanding of how patterns of parasite transmission amongst animals and humans could change in response to the multiple, interacting stressors being placed on the global ecosystem [12]. From this understanding we need to create improved decision support systems that allow for sustainable control and management of hosts, vectors and parasites. However, the wide range of relevant anthropogenic stressors, the enormous diversity of parasite taxa, life history traits and infection strategies, and the range of possible functional responses and interactions between them – coupled with simultaneous responses among host populations – make this a hugely challenging task.

One route to reducing the scale of the health challenge we now face under conditions of uncertainty is to draw upon proxy observations from comparable systems [13]. In terms of parasitology, much of the relevant proxy information has been drawn from wildlife disease ecology, which has tended to pay more attention to the issues of global change than comparable studies on human and domestic animal parasites. In this paper we demonstrate how ecological studies of parasites in wildlife may be used to enhance our understanding of stressors arising from global change, which are likely to be important in the context of parasitic infections of humans and domestic animals.

First we consider the influences of some major abiotic and biotic stressors associated with global change and, second, how these stressors might affect parasite life cycles, transmission and ecology. In doing so, we highlight that the abiotic-biotic distinction is blurred, particularly as many stressors also act simultaneously and indirectly on parasites through their hosts. Third, we explore how parasites may respond to the evolutionary pressures of such stressors. Finally, we consider how these complex impacts of global change potentially militate against the sustainable control of parasites affecting animals and humans, and make suggestions for improved understanding and control in an uncertain future world.

Section 1: Anthropogenic abiotic and biotic stressors affecting parasite transmission

The majority of ecosystem stressors are driven by industrialization combined with human population growth. These in turn are responsible for increased resource use and generation of waste products, many of which have negative impacts on the environment in complex direct and indirect ways, which may subsequently affect disease risks. For example, the combustion of fossil fuels for energy production and for powering transportation modes significantly contributes to air pollution and carbon dioxide emissions, which promotes climate change. Changing land uses, including farming for food, further contributes to climate change and both are considered major drivers of biodiversity loss. Broad scale biodiversity loss, latitudinal and altitudinal host range expansions and retractions, reduced wildlife population sizes and more limited habitat connectivity are subsequently affecting host interactions and changes in parasite transmission.

Climate change

The multiple components of climate change, including temperature, precipitation and atmospheric CO_2, have been extensively studied individually [14–16], but the interactions between these environmental stressors and the consequent effects on parasite transmission are complex. Thus, there is considerable uncertainty about how future climate variation and change will affect disease dynamics [17–19]. Multiple stressors might affect different life-history traits, potentially influencing both parasite and host fitness ([20]; see Section 2). Higher temperatures, for example,

85 often increase parasite growth, reproduction and infectivity [21,22], yet can also increase mortality
86 [23,24]. Likewise, while temperature elevations accelerate the replication of arthropod-borne
87 viruses in their insect or tick vectors, they also affect vector mortality and biting rate, making the
88 net effect of temperature increase on transmission difficult to predict without detailed knowledge of
89 each component in the system [25]. Other hosts may respond to increased temperatures by
90 exhibiting acclimation, adaptation or be forced to shunt resource investment into various life-
91 history components, resulting in thermal preference shifts. Poikilothermic hosts are particularly
92 vulnerable to temperature shifts, but also show remarkable adaptations. Some fish, for example,
93 exhibit adaptive behavioural traits to reduce transmission risk, by selecting thermal conditions that
94 are detrimental to parasites ('behavioural fever'; [26]) or selecting flow conditions that minimize
95 fitness costs of infection and potentially reduce transmission [27,28].
96
97 Disentangling anthropogenic environmental change from that of natural variation is problematic,
98 particularly for indirect effects and naturally rare events such as extreme weather conditions or
99 disease outbreaks [29,30]. The relationship between environment and transmission is also complex.
100 Different environmental parameters may have additive, multiplicative or antagonistic as well as
101 nonlinear effects on transmission, and themselves be inter-correlated or vary at different spatial or
102 temporal scales, with such effects difficult to measure [31,32]. Some relationships are a
103 consequence of transmission mode. For example, flooding events might be a key driver of some
104 water-borne disease epidemics [33], whilst drought conditions cause hosts to aggregate at sites
105 where water is available, amplifying transmission and triggering outbreaks of vector-borne diseases
106 such as African horse sickness and Rift Valley fever [34,35]. Range shifts due to climate warming
107 might change the distribution of vector borne diseases, including malaria [36] and Rift Valley fever
108 [37]. However, climate change is not spatially homogeneous and could render previously suitable
109 areas unsuitable for transmission and *vice versa* [38]. Host-vector-parasite interactions could also
110 decouple if the rate of range shift differs between the host and parasite, leading to a decline in some
111 diseases [39]. For example, tick-borne encephalitis virus (TBEV) transmission is sustained only
112 when temperatures result in synchronous feeding of larvae and nymphs [40]. Projected temperature
113 rises might desynchronize feeding and shrink the area within which TBEV persists [41]. Even the
114 immediate effects of change in temperature and rainfall on parasites are therefore complex and
115 strongly modified by host factors.
116
117 *Pollution*
118 Pollutants can cause sub-lethal physiological stress to hosts and hence reduce their capacity to
119 withstand parasite invasion and/or proliferation, potentially increasing infection levels indirectly
120 (e.g. [42]). However, pollutants also impact parasites themselves, and in aquatic ecosystems, both

121 the infective stages of parasites and their intermediate hosts can be highly sensitive to their effects
122 [43]. Heavy metals can inhibit the release of trematode cercariae from molluscan hosts, as well as
123 impair their swimming behaviour and longevity [44–46]. Pharmaceutical pollutants are widespread
124 stressors likely to affect host susceptibility to disease. The scale of this threat is increasingly
125 apparent in aquaculture: in Chilean salmon farms alone, hundreds of tonnes of antibiotics are used
126 annually [47]. Eutrophication – another important stressor of aquatic ecosystems, arising from
127 excessive nutrient input – is associated with elevated intermediate host densities, parasite fecundity,
128 and increased prevalence of certain pathogen infections [48]. However, as yet, there is no overall
129 consensus on its consequences for general patterns of infection [49,50].
130
131 Other forms of pollution are less well studied with regard to disease transmission. Whilst it is
132 known that light pollution can impact the structure and function of ecosystems via cascading effects
133 [51], and that natural light cycles govern both relevant parasite life history traits (e.g. egg hatching;
134 [52]) and intermediate host behaviours (e.g. zooplankton diel migration; [53]), studies on the effects
135 of light pollution on human parasite transmission remain limited [54]. Although the introduction of
136 electricity to socio-economically developing communities has overall human health benefits, night
137 lighting inevitably attracts certain insect vectors and increases human night time activity, which is
138 implicated in higher incidences of leishmaniasis and malaria in some regions [54]. In other insect-
139 vectored diseases, artificial lighting may have a less overt effect on transmission dynamics:
140 triatome bugs, the vectors of Chagas' disease, typically avoid well-lit areas and artificial lighting
141 may be driving Chagas transmission towards a sylvatic cycle [54]. Noise pollution, a known stress-
142 induced modulator of the immune response [55] that can significantly affect behaviour and
143 predator-prey interactions [56], has not yet been considered in terms of infectious diseases. The
144 gaps in knowledge concerning the impacts of all types of pollution on parasite transmission are
145 considerable, and without this information it is challenging to assess its importance across host-
146 parasite systems.
147
148 *Habitat loss and fragmentation*
149 Habitat alteration due to climate change or anthropogenic activity poses a major threat to
150 ecosystems, often leading to substantial biodiversity loss, ecosystem functioning and services, and
151 reduced resilience to external stressors [57–61]. This in turn may alter host-parasite interactions, by
152 either increasing [62–65] or decreasing [66–68] infection levels, depending on nuances of host and
153 parasite life history (see Section 3). The effects of habitat change can even have contrasting effects
154 on closely-related parasite species infecting the same host; for example, sunbirds in disturbed
155 habitats exhibited increased prevalence of *Plasmodium lucens* but decreased prevalence of *P.*
156 *megaglobularis* [67]. Habitat loss and fragmentation also increases the frequency of 'edges' -

157 transition zones between habitats [69,70] - which are typically exposed to more extreme climatic
158 conditions than interior sites [70]. Habitat edges often promote increased species diversity (i.e. the
159 richness and/or relative abundances of species [71]), resulting in altered levels of interspecific
160 competition and parasitism [72–75]. How the differential effects of edge versus interior sites
161 impacts parasitism varies between host-parasite systems; infections may significantly increase [73],
162 decrease [74], or be unaffected [76]. Although maintaining connectivity generally benefits
163 biodiversity, it can also facilitate disease spread [77] so modular populations might be favoured in
164 terms of vulnerability to disease (but see [78]).
165
166 *Host density and farming intensification*
167 Over the past 50 years, there have been unprecedented changes in farming practices and associated
168 land use [79]. Although forestry currently occupies about 30% of total land area, the impact of
169 deforestation and land use intensification, especially on soil degradation, is significant. Growth in
170 crop production and livestock has been driven by the requirement for higher yields. Livestock
171 production is the largest user of agricultural land, accounting for more than 30% of the Earth's ice-
172 free terrestrial area [79,80] but aquaculture is the fastest growing food sector [81]. Farming
173 typically relies on concentrating and containing inbred hosts, which can increase host exposure to
174 and facilitate parasite transmission [82,83]. Host density is particularly important for tick-borne
175 pathogens [84], as these vectors are relatively immobile and host-parasite contact frequencies tend
176 to be driven by changes in host abundance and/or behaviour. Chronic stress induced by high
177 stocking densities in aquaculture can have important implications for fish immunocompetence [85],
178 but relationships with infection levels are variable. Whilst high host densities can promote greater
179 parasite population densities, the number of conspecific parasites per host may be reduced [86].
180 This 'dilution effect' (see *Biodiversity Loss*, below) is illustrated by a reduction in directly
181 transmitted sea lice at the high host densities in salmonid cage aquaculture [87]. Positive effects of
182 high host density on transmission can be attenuated by mixing susceptible and resistant hosts in
183 rotational grazing systems [88], showing the importance of multiple hosts in modifying infection
184 pressure. However, in aquaculture where hundreds of thousands of hosts are contained together this
185 is not yet possible [89], partly because of the need to track farmed fish in the event of an accidental
186 release, and also because of concerns about disease transmission between farmed and wild stocks
187 (and *vice versa*).
188
189 *Urbanisation*
190 While density-dependent transmission of human parasites may be expected to increase with high
191 host densities and ownership of companion animals, decreased human-wildlife contact and better
192 sanitation in cities generally point to lower levels of disease transmission among such populations

(e.g. [90,91]); although there are exceptions. Dengue, for example, is more prevalent in urban areas due to the provision of suitable human-created microhabitats for the *Aedes* mosquito [92]. Urban environments with high human densities are potentially more vulnerable to water-borne or faeco-orally transmitted parasites if investment in sanitation infrastructure is neglected or disrupted due to socio-economic unrest. Poverty is an important related factor; a study of the contiguous cities of Laredo (US) and Nuevo Laredo (Mexico) on the US-Mexican border found that dengue transmission was strongly affected by income, and hence access to technologies such as air conditioning [93]. Overall, pathogens likely to thrive as a result of urbanisation tend to be either those for which transmission is strongly density-dependent, or those with vectors or reservoirs which are themselves well-adapted to urban environments. The net effect on parasite burdens will be highly case-specific and difficult to predict, especially where urbanisation is rapid and strong interactions with rural populations persist [94].

An additional issue regarding urbanization or changing land use is related to potential wildlife conflicts. Most emerging and re-emerging human infectious diseases (EIDs) are zoonotic, typically with origins in mammalian wildlife [95,96] or interactions between wildlife and domestic animals [97,98]. This might increase further as habitat loss forces the co-occurrence of wildlife and humans; although this could be offset by the greater effects of biodiversity loss (see below). Australian urbanization, for example, has detrimentally affected many possum species, but the brush tail possum *Trichosurus vulpecula* is often attracted to human dwellings for shelter, nesting and the steady food supplies of urban gardens and rubbish bins. This in turn has led to increases in the transmission of zoonoses such as *Toxoplasma gondii*, *Leptospira* spp. and *Cryptosporidium* spp. [99].

Associated with urbanization, is increased road building. Approximately 60% more roads are projected by 2050 compared to 2010, mostly in developing countries [100], potentially making road building one of the most significant drivers of future environmental change [101]. Road building has already increased the risk of some diseases associated with human development (e.g., agricultural intensification), with an increase in the number of human hantavirus cases reported following completion of a highway through the Brazilian Amazon [102]. Such large scale road building will almost certainly further facilitate bushmeat hunting in the most biodiverse regions of the planet [103] and change the scale at which people are able to move wild animals out of newly exploited areas and into commodity chains, thereby increasing public health risks.

Biodiversity loss

228 Current extinction rates are estimated to be 100-1,000 times greater than background levels [104],
229 with biodiversity loss being one of the hallmarks of the Anthropocene [105]. Loss of host diversity
230 can reduce disease risk directly or indirectly through the associated loss of parasite diversity [106].
231 For example, reduced risk of African sleeping sickness in humans [107] has been related to the loss
232 of wildlife host biodiversity (reviewed in [108]). Wildlife biodiversity has often been correlated
233 with human infectious disease risks. Examples include correlations between mammalian
234 biodiversity and global biogeographic patterns of human infectious diseases [109], elevated
235 likelihood of observing emerging infectious diseases [95] and increases in human pathogen richness
236 and prevalence for some diseases [110]. However, in these cases it can be difficult to separate cause
237 from correlation since areas with high levels of biodiversity are also characterised by other,
238 unrelated, risk factors for disease transmission such as climate and poverty. Nevertheless, the fact
239 that most human infectious diseases have origins in animals, mostly wildlife, support suggestions
240 that these correlations are mechanistically reasonable and that one large scale consequence of
241 biodiversity loss could be an overall reduction of disease transmission.
242
243 Wildlife biodiversity loss can, however, also increase disease risk. In some ecosystems the number
244 of transmission-competent hosts is 'diluted' by abundant non-competent hosts, so the chance of a
245 vector feeding on a suitable host, or of a motile infective parasite successfully contacting a
246 transmission-competent host, may be reduced. When members of a host community are lost due to
247 – for example – habitat loss, the risk of disease to a focal host (e.g. humans) could rise. This
248 appears to be the case for Lyme disease in North America [111–113] and there is support for
249 generality across multiple systems [114]. In addition, generalist host species that cope more
250 effectively with human pressure may exhibit greater reservoir competence, or the capacity to
251 transmit pathogens, such that biodiversity loss could select for species that contribute to higher
252 levels of parasitism [115]. Nevertheless, many studies continue to demonstrate that the dilution
253 effect is likely to be of limited generality [116–119], and the net contributory effect of biodiversity
254 (and its loss) to disease risks requires the balance of costs and benefits to be more thoroughly and
255 objectively addressed [120,121]. The notion that wildlife biodiversity can provide an important
256 service in regulating the risk of infectious disease is attractive and has received widespread
257 exposure, although because the interactions that result in transmission events can be complex, the
258 evidence for widespread effects continues to be mixed. In many cases, community composition
259 including relative abundance, rather than biodiversity loss, is a greater predictor of disease risk
260 dilution [116–119].
261
262 Biodiversity loss, and its implications for disease risks, may also be experienced at the individual
263 host scale, with subsequent impacts on micro- and macroparasitic infection and transmission. All

multi-celled organisms are colonised internally and externally by communities of bacteria, eukaryotes, archaea and viruses [122]. These microbiota play a critical role in host health, particularly the gut microbiota and its involvement in immune system development and function [123,124]. In vectors such as *Anopheles* mosquitoes and triatome bugs, an enriched midgut microbiota stimulates up-regulation of immune genes that inhibit microparasite development; however, reduced microbiota diversity arising from direct antibiotic treatment or by ingestion of antibiotics circulating in a blood meal, is associated with increased microparasite infection of the insect host [125–127]. Moreover, microbiota depletion increases survival and fecundity of the vector itself, potentially exacerbating microparasite transmission [127].

The effects of anthropogenic stressors and within-host biodiversity loss on enteric helminths are highly species dependent. Certain antibiotics remove *Syphacia* pinworms and other gut helminths in lab mice, as a direct effect on the parasites themselves or through altering microbial composition, yet other antibiotics have the opposite effect on *Aspiculuris* pinworms, with treated hosts harbouring nearly twice as many worms as controls (reviewed in [128]). Similarly, there are direct links between the loss of bacterial diversity and truncation of helminth life cycles. Eggs of hookworm *Trichuris muris* require a structural component of Gram-negative bacteria from the host's gut to trigger a signal transduction cascade to stimulate hatching [129]. The nematode *Heligmosomoides polygyrus bakeri* exhibits bacterial dependence for larval development; reared in axenic conditions, the nematodes do not survive beyond second stage larvae [130]. This suggests that transmission of both *T. muris* and *H. polygyrus* is unlikely to be successful if gut microbiota diversity is inadequate, though confirmation is required from *in vivo* studies. These examples illustrate the potential importance of internal and external biodiversity to parasite transmission and maintenance, and suggest that impacts of biodiversity loss could be more far-reaching than is currently recognised.

Altered interspecific interactions

Changes in host interactions, often linked to the stressors listed above, can drive disease emergence in new hosts. We have already highlighted this problem in association with increased human-wildlife contact, but this in turn might be altered by a range of non-human interactions. The fundamental role of parasites in food webs [131,132]; thus, anthropogenic changes that reduce the density of higher trophic level species that feed on larval parasite stages [133] could directly increase disease transmission to competent hosts. Parasites may also indirectly disrupt predator-prey interactions [134] and abiotic factors may affect trophic transmission by altering host foraging activity [22,135]. In addition to these altered parasite-predator-prey interactions, parasites can affect native-invasive host interactions (e.g. [136,137]); newly invading hosts either bring novel

pathogens with them, or – having escaped their own native parasites – dilute the pool of susceptible native hosts [138].

Finally, parasite-parasite interactions affect transmission, with many studies highlighting the complex interactions of co-infecting parasites in wildlife (e.g. [139]) and livestock (e.g. [140]). As well as parasites having their own microbiota, they can serve as hosts for hyperparasites, the occurrence and life-history of which is likely to be influenced by environmental changes [135,141]. Abiotic or biotic stressors may even drive symbionts to adopt parasitism, for example where there is high competition on the host (e.g. [142]). Artificial manipulation of species interactions can be used in biocontrol, as in the case of *Wolbachia* infection of mosquitoes, which reduces their vectoring capacity [143].

Interacting abiotic and biotic factors

The above list is not comprehensive, but rather highlights some of the key abiotic and biotic factors that may act together as 'cocktails' of stress, with implications for increasing or decreasing disease risks. Identifying the direct and/or indirect factors responsible for changes in disease risk is challenging because multiple stressors act simultaneously on both parasites and their hosts. Depending on habitat and season, the peak impact of different abiotic stressors can occur in or out of phase with one another; thus, whilst some organisms may be exposed to multiple stressors simultaneously, others will experience them sequentially. Yet, the consequences of multiple, interacting environmental threats for parasite transmission remain unclear: when they co-occur temporally and spatially, their combined effects may be additive, antagonistic or synergistic [144,145]. For example, whilst elevated seawater temperatures increase mortality rates of oyster larvae, this can be offset by simultaneous acidification, which reduces the growth of pathogenic bacterial infections [146]. On coral reefs, the interaction between ocean acidification and warming contributes to coral bleaching and reduced disease resistance, leading to increased pathogenicity of existing pathogens and the emergence of new diseases [147]. These two examples are rare, because compared with terrestrial and freshwater systems, marine systems are often neglected with regard to assessing the impact of environmental stressors [148].

Section 2: How might parasite life history traits modulate responses to abiotic and biotic stressors?

Given the complexity of the possible effects of global change on parasite transmission, understanding the factors that drive responses across parasite taxa is essential for more general predictive ability. Here we consider the diversity and complexity of parasite life cycles, since the number and diversity of hosts underpins not only how parasites might respond to environmental

change, but also their relative fitness and resilience to environmental change at different life stages [149].

Parasite life cycles exhibit remarkable diversity in form and complexity. Whereas some parasites can complete their life cycle infecting a single host organism, others must negotiate their way through several host species in a particular sequence in order to achieve reproductive success. Life cycles with greater complexity rely on biodiverse and integrated communities, and as such may be highly sensitive to the loss of individual components, in the form of hosts, vectors or species interactions required for transmission [106,150]. The level of life cycle flexibility and host specificity is also likely to influence the sensitivity of parasites to changing environments, and their ability to prosper in perturbed ecosystems.

Life cycle flexibility

The use of paratenic hosts, which are not necessary for parasite development but can sustain parasites and make them available to subsequent obligate hosts, may positively influence transmission if environments become unsuitable or if non-native species outcompete and drive native obligate intermediate hosts locally extinct [151]. An example is provided by two sister species of *Bothriocephalus* cestode, of which only one (*B. gregarius*) utilises a facultative paratenic host. Whereas paratenic hosts enhance the probability of *B. gregarius* successfully infecting definitive host fish, resource competition within paratenic hosts lowers infection intensities, and smaller progeny are produced relative to *B. barbatus* [151]. Consequently, reduced energy expenditure on growth enables *B. gregarius* to invest more in reproduction and dispersal, increasing the likelihood of re-establishment in a new population of intermediate hosts [151–153]. Alternatively, if populations of the definitive host of *B. gregarius* were to rapidly decline, the paratenic host might potentially replace this host [151].

Parasites that have the capacity to truncate their life cycle may be advantaged under fluctuating environments [154]. For instance, if an obligate host is temporarily unavailable due to seasonally-induced migration or anthropogenic activity, developmental requirements for the absent host would be disadvantageous [155,156]. Flexibility in host use may therefore allow parasites to cope with seasonal variation in host availability; for example, *Gymnophallus choledochus* normally employs a three-host cycle in summer, but switches to a two-host cycle during winter [157]. In other species, a host may be lost permanently due to strong selection pressures, such as the lack of predators facilitating onward trophic transmission; this ‘missing host hypothesis’ could explain the two host life-cycle of schistosomes [155]. In a more extreme example, *Mesostephanus haliasturis* can forgo

371 sexual reproduction by completing development, via asexual reproduction, in its snail host [158].
372
373 Other parasites employ progenesis (precocious sexual maturation) to shorten their life-cycle. For
374 example, host diet and increased temperature can induce progenesis in *Stegodexamine anguillae*
375 metacercariae via secreted host-stress signals [154–156]. Thus progenesis may benefit immature
376 trematodes when transmission to definitive hosts is compromised by the health of an intermediate
377 host [156]. For other parasites, such as the hyperviviparous gyrodactylids, progenesis is the norm,
378 and the first-born offspring always develops asexually from the parental worm while it is still an
379 embryo [159]. This adaptation, together with a direct life cycle, facilitates invasion of new habitats:
380 a host only needs to be infected with a single *Gyrodactylus* worm to initiate an epidemic [160].
381
382 Life cycle plasticity is particularly advantageous when facing increasing environmental and host
383 uncertainty. Monogenean parasites of the genus *Polystoma*, which typically infect the urinary
384 bladder of frogs, exhibit life cycle dimorphism, with parasites maturing in either 3 weeks or 3 years
385 [161,162]; precocious maturation (neoteny) on the tadpole gills occurs when environmental
386 conditions are unsuitable for normal development [163]. Although release of eggs from the slower
387 growing bladder morphs is induced by the mature host's gonadotropin secretions during breeding
388 season, both the timing of parasite egg-hatching and tadpole development are sensitive to ambient
389 temperatures and chemical environments [164]. Disrupted host chemical balance and light
390 intensity, for instance caused by pollution, may shift the equilibrium between morphs to favour
391 either the neotenic or the slow growing phenotype [163,164]. Similarly, phenotypic plasticity in the
392 life cycle of the common dog parasite, *Toxocara canis* is reportedly dependent upon the
393 physiological status of the host [165], though host drug treatment might have a hidden influence.
394
395 *Specialist vs generalist life cycles*
396 The evolutionary divergence of parasites has generated varying degrees of specialisation in parasite
397 traits within different habitats and hosts, some of which are more likely than others to enhance
398 parasite success in unstable environments [149,166,167]. Although it is logical to predict that
399 generalist parasites are more resilient to global change than specialists [115,168], this is very
400 context dependent [152]. Furthermore, if global change results in new conditions that are stable,
401 parasites that are locally adapted might develop more specialist tendencies [169].
402
403 Zoonotic parasites demonstrate varying degrees of host specificity due to transmission via three,
404 non-mutually exclusive life-cycles: sylvatic, domestic and anthroponotic [4,149,170]. Host
405 specialisation arises due to parasites' investments towards infectiousness and longevity in particular
406 hosts. For example, the nematodes *Trichinella britovi* and *T. spiralis*, both found throughout

Europe, possess sylvatic and domestic (swine) host cycles. However, their epidemiology differs due to their higher adaptability to either swine (*T. spiralis*) or carnivore (*T. britovi*) hosts [4]. Nonetheless, re-establishment of *T. spiralis* in a red fox (*Vulpes vulpes*) population, decades after its elimination from domesticated swine in Northern Ireland, demonstrates how host diversity increases resilience to anthropogenic farming activity; i.e. by providing alternative sylvatic reservoir hosts until preferred domestic hosts become vulnerable to infection [166,171,172].

Parasite longevity

Parasite life span, and the time spent inhabiting different hosts, will influence the susceptibility of parasites to environmental changes, and the type of responses that are most likely to arise. Whereas short-lived parasites with rapid life cycles may be more capable of evolving adaptive response to chronic directional changes in environments, long-lived individuals may be better equipped to withstand acute, transient perturbations. The life span of parasitic worms can be hugely variable; among the nematodes it can range from 3 days in free-living *Rhabdias bufonis* adults to 20 years for *Loa loa* (reviewed by [173]); among cestodes, *Taeniarhynchus saginatus* can live in humans for 35 years [174]; and schistosome life spans of 20-30 years are documented [175], though mean longevity in optimal hosts is in the range of 5 years [176]. Parasites with viviparous reproduction, such as *Gyrodactylus* spp., tend to have the shortest life spans (few days) with age not only determining reproductive output but also reproductive mode [177]. For all species, timing of pre-patent and patent periods varies and reproductive output typically declines with parasite age and host status (reviewed by [173]). Aside from the longevity of mature worms, it is essential to consider persistence and resilience of environmental stages when considering how any particular parasite population will respond to global change.

Parasite reproductive strategies

Long-lived parasite species tend to be iteroparous (e.g. *Loa loa*) whilst other parasites exhibit semelparity (e.g. the human pinworm *Enterobius vermicularis*). Within a parasite species, timing of reproduction is intricately linked to biological and environmental factors, and for many species transmission is seasonal; in extreme cases this can be incredibly brief. For example, *Polystoma integerrimum* transmission only occurs during the host breeding season [178], and in the related species *Pseudodiplorchis americanus*, transmission can be restricted to just 3 h per year, being entirely dependent on monsoon rains creating suitable habitats [179]. If the rains fail, the adult parasites can re-absorb nutrients from ovoviviparous larvae held *in utero* [179], but the long-term implication of this strategy is unknown. Similarly, disrupted weather patterns threaten other seasonally transmitted parasites, such as brood parasitic birds, which risk phenological mismatch with their hosts [180].

Reproductive strategies of endoparasites, in particular, are determined by trade-offs in energy investments against other life history traits [167]. Schistosomes are the only digeneans whose adult stages are exclusively dioecious and dimorphic [181,182]. Only male *Schistosoma mansoni* retain hermaphroditic traits, implying they are energetically costly, and may have restricted female body-form specialisation required for efficient egg dispersal [152,183,184]. Evolution of dioecy in schistosomes via host-parasite co-evolution demonstrates resilience to long-term environmental changes, however slowly-evolving adaptations may be disadvantageous in the face of short-term perturbations [181,184]. For both hermaphrodite and dioecious parasites, hybridization provides another tool in the parasite's ability to adapt to changing environments [185].

Life cycle determinants of global change effects on parasites

Life-history theory predicts that, whilst parasites with direct life-cycles have fewer energetic restrictions imposed by intermediate hosts and can invest more energy towards growth and reproduction [152,153], their dependence on a single host for reproduction might jeopardize survival. In contrast, indirect life-cycles offer increased likelihood of 'rescue' for parasites, which may alter host specificity via the addition or exclusion of hosts [151,155,186]. Alternatively, parasites that demonstrate increased specialisation of specific developmental stages, such as the dimorphic stages of *Polystoma integerrimum*, can inhabit a wider range of host environments and increase the probability of reproductive success [155,163,164]. Finally, dependence upon specific vectors or intermediate hosts for dispersal and reproduction renders parasites extremely vulnerable to both spatial and temporal climatic changes [2,187,188]. Recent studies suggest that parasite life history traits may be enhanced by climate shifts and anthropogenic stressors associated with on-going global change [189], thus providing relatively benign parasites with the potential to become increasingly pathogenic. However, while this is considered a serious threat to wildlife communities already facing mounting population pressures, conclusions are usually derived from assessments of single stressors or single parasite life stages, while the net effect to the parasite and host's whole population are rarely determined [190].

The parameters that characterize the life histories of individual parasite taxa are likely to play a critical role in determining their relative resilience in the face of changing environments. Parasite life cycles range in complexity from direct life-cycles with a single host species to those with multiple intermediate and facultative paratenic hosts. The diversity of life cycles and life histories, coupled with variable flexibility and specificity of the parasite, mean that there are likely to be winners and losers among parasites in perturbed environments. Whereas increased life cycle complexity might leave indirectly transmitted parasites susceptible to environmental change, if they

acutely affect an obligate host population, the existence of multiple intermediate and/or reservoir hosts in a life cycle [156] and facultative paratenic hosts, may provide a parasite with greater scope for adaptation [152,191]. Parasite survivorship and fecundity are the two key life-history traits that impact parasite fitness, and therefore transmission. Such traits will be subject to environmental stressors, such as drug exposure, that vary over time [192,193]. In the longer term, where stressors inhibit parasite transmission, they are likely to also impose selection pressure on life history traits.

Section 3: Evolutionary change

Parasites are perhaps uniquely predisposed to rapid evolution under global change. Not only are effective population sizes large and generation times typically short, but transmission imposes an exceptionally strong filter to exclude maladaptation: infective stages either find a host or die. Genotypes better suited to transmission under particular conditions will presumably be strongly selected for, with unpredictable variation in climate or host availability encouraging genetic diversity and within-genotype flexibility in key life history traits. The potential for parasites to out-evolve their hosts suggests that increasing, rather than decreasing, parasite risks and burdens will be the norm under global change. However, the complex interactions of current stressors, as discussed thus far, can also act upon parasites at the genetic level, complicating predictions and leading to unexpected future infection patterns. Observations of parasite evolution in response to changing environments in nature are rare, but results from a few example systems are offered here to illustrate the potential diversity of parasite adaptive responses to global change.

Resilience and plasticity

The complex links between existing environmental variation and disease transmission [194–196] suggests that identifying the impact of anthropogenic activities on the evolutionary responses of parasites over and above natural variation might be challenging. Models predict that increasing seasonal climate variability will drive the evolution of greater resilience of pathogens to environmental fluctuations [31]. This has been demonstrated, with more extreme monsoon rainfall patterns linked to the rise in dominance of a strain of cholera resilient to water quality and quantity fluctuations [31]. Similarly, plasticity in parasite traits is likely to evolve in response to increased climatic variability, exemplified by the evolution of a plastic transmission strategy in *Plasmodium relictum* that has seen reproductive rates increase during periods of vector availability, thereby maximising transmission [197].

Human management of host species and treatment strategies (see Section 4) are also important drivers of pathogen resilience. For example, selection pressure has resulted in altered strain dominance of the potato cyst nematode *Globodera rostochiensis*. Earlier planting of potatoes to

allow growth in months historically too cold for larval invasion is now linked with a faster developing, more fecund strain of the parasite [198]. But by far the most pervasive evolutionary phenomenon due to intervention practices is that of increased drug resistance, increasingly seen in parasites of humans [199,200] and livestock [201] including aquaculture species [202,203].

Infectivity and virulence

Habitat change can strongly influence host-parasite interactions, shifting parasite diversity, abundance and transmission dynamics (discussed in Section 1). Evolving parasite infectivity and virulence may contribute to factors underlying these observations. Habitat fragmentation leads to smaller, patchier and more isolated populations [3]. In host-parasite interactions, infection and transmission will become more localised under such conditions. Theory and empirical data indicate that this can lead to the evolution of reduced parasite infectivity because of self-shading. This effect arises because, as individual susceptible hosts are rapidly infected locally by virulent parasites, they are surrounded by other infected hosts, which will reduce opportunities for further transmission of horizontally transmitted parasites [204,205] and parasites that use mixed transmission strategies [206]. In contrast to habitat fragmentation, intensification of farming practices is predicted to drive evolution of increased virulence; higher host availability reduces the adaptive cost of increased virulence due to host mortality [89]. Key evidence for this is the recent increase in pathology and mortality due to *Flavobacterium columnare* in densely stocked Finnish freshwater fish farms, linked to the emergence of more virulent, infective strains of the pathogen [207].

Bet hedging

Unpredictable conditions, such as the timing of host availability, should favour parasites that can produce offspring that vary in their life history or transmission strategies [208]. Spreading the risk, or 'bet hedging', allows parasites to increase the chances that at least some of their progeny will survive and infect a competent host. It is reasonable to expect that parasites will increasingly adopt such bet hedging strategies to ensure survival in rapidly changing environments. The nematode *Nematodirus battus* historically exhibited a single generation per year with overwintered eggs hatching in spring to coincide with arrival of newborn lambs [209,210]. Evolution of multiple generations per year [211] via the production of autumn-hatching eggs that do not require vernalisation, has mitigated against asynchrony between larval presence and the availability of susceptible hosts in years with early warm springs [212]. However, anthropogenic changes may also hinder the evolution of parasite bet hedging strategies. Variation in life cycle traits (e.g. rate of development, egg laying and hatching) of the fish louse *Argulus foliaceus* infecting farmed fish is lower than in wild populations, likely as a result of reliable host availability in fish farms compared to natural ecosystems [213].

551

552 *Host switching*

553 Global change might constrain host-parasite coevolution, if the benefits of new mutations that
554 enhance fitness (selective sweeps) are not realized in a rapidly changing environment [214].
555 Alternatively, host-switching is a potential parasite adaptation to global change [215,216], should
556 the availability of preferred hosts be decreased via geographical range shifts, phenological
557 asynchrony, human management or control strategies. In some cases, switching to alternative hosts
558 may not be optimal for parasite development, leading to reduced parasite offspring or survivorship
559 and thereby reduced probability of transmission. This may limit how much host switching actually
560 occurs in changing environments. However, Jones et al. [217] showed that, whilst costs of prey
561 switching for a parasitoid, a particular kind of parasite, were severe in the first instance, these costs
562 were ameliorated over successive generations. Furthermore, the force of selection will play a key
563 role in the drive to host switch. In the case of the Guinea worm (*Dracunculus medinensis*), an
564 extremely simple but effective control programme that filtered the copepod vectors from
565 contaminated drinking water, effectively blocked transmission [218] and reduced the number of
566 human cases from an estimated 3.5 million cases in 1986 to just 126 in 2014. However, in 2015,
567 459 infections were recorded for the first time in dogs [219,220] suggesting a potential host-
568 switching event, possibly driven by the effective control measures blocking transmission to humans
569 [220].

570

571 The introduction of invasive host species generates unique opportunities for non-native parasite
572 communities to come into contact with new hosts, and considerable potential for host switching.
573 Classic examples of this include the introduction of squirrel parapoxvirus into UK red squirrels
574 (*Sciurus vulgaris*; [136,221,222]) and crayfish plague (*Aphanomyces astaci*) into European crayfish
575 (*Astacus astacus* [223]). Host switching from introduced to native hosts appears equally common in
576 parasites with direct and indirect life cycles, and worryingly the majority of those reported are more
577 virulent in native hosts than in the co-introduced invasive host [224]. Although, considering that we
578 still know very little about parasite speciation, it is difficult to predict future outcomes, and other
579 mechanisms, such as niche specialization [225] and hybridization [226], could also affect both
580 speciation and host range.

581

582 *Multiple evolutionary targets for adaptation*

583 The above examples demonstrate how the effects of human activity and climate change are varied
584 and far ranging with respect to parasite evolution. Targets of evolution are already altering the
585 epidemiology of parasites; resilience, strain variation in phenology, bet hedging in key life history
586 traits, and host switching, all demonstrate that through past unpredictability in transmission,

587 parasites are well adapted to future changes in climate and host availability. As the evidence for the
588 anthropogenic effects on parasite adaptive responses builds, we must now consider the evolutionary
589 capabilities of pathogens as an integral component to predicting the future landscape of host-
590 parasite interactions under pressures of global change. This will be particularly important when
591 considering the consequences of parasite control programmes; arguably the greatest selective
592 pressure faced by parasites in their evolutionary history.
593
594 **Section 4: Control programmes and predictive epidemiology in a changing world**
595 The evidence presented above offers substantial insights into the challenges faced by advocates and
596 managers of control programmes that target specific pathogens affecting humans or domestic
597 animals because of perceived or actual threats to health and/or productivity. Such intervention
598 programmes are possible because of developments in our understanding of the life cycles and
599 ecology of parasites affecting humans and livestock, primarily gathered in the Victorian era [227].
600 Early optimism amongst health practitioners in wealthy countries that such strategies would
601 eradicate infectious diseases continued up until the middle of the twentieth century (reviewed by
602 [228]); yet despite early (and enduring) optimism, relatively little success in terms of eradication
603 has been made and among the 'Neglected Tropical Diseases', only Guinea worm is scheduled for
604 eradication (most likely because only low-tech solutions are necessary to interrupt the transmission
605 cycle; see Section 3).
606
607 There are many reasons for the lack of success in eradicating infectious diseases, including – for
608 malaria - the resurgence of infection after programmes are abandoned due to the development of
609 pathogen resistance to insecticides and/or medicines (e.g. [229]). For other infections, including the
610 major geohelminth species (*Ascaris*, *Trichuris*, hookworm) and schistosomiasis, a lack of
611 knowledge about the abiotic and biotic factors that underpin the precise spatial distribution of
612 infection, plays a key role in our failure to eradicate infections. This lack of knowledge stems from
613 life-history traits of the parasites themselves (see Section 2), since they have not only evolved to
614 occupy a niche within the water-sanitation related behaviours of humans, but the symptoms of
615 infection (including diarrhoea, nausea and fever) often have low specificity and are shared with a
616 wide range of pathogens. In resource-poor areas where parasitic infections are common, the lack of
617 differential diagnostic information coupled with non-life-threatening symptoms mean that many
618 infections go unnoticed or unreported.
619
620 In the absence of detailed ecological information, there has been a concerted effort to distribute
621 human medicines through 'Mass Drug Administration' (MDA) programmes in areas of high
622 transmission [230], aided by donations from large pharmaceutical companies. These MDA

623 campaigns rely largely on the presumptive treatment of putatively exposed individuals [231]. The
624 expectation, translated from the outputs of mathematical models, is that repeated MDA will reduce
625 the size of the parasite population and simultaneously reduce levels of morbidity attributable to
626 infection [231]. Recent analysis of NTDs in Africa suggests, at first glance, that the MDA strategies
627 have succeeded in reducing the number of NTD infections; however, problems remain in terms of
628 attributing causality to the success of these programmes [230]. First, the historical data are
629 imprecise and patchy; the diagnosis of malaria and other infections has been characterized for
630 decades by a lack of sensitive and/or specific tools [232]. Second, co-incidental environmental
631 change over recent decades may have altered the characteristics of soil and water phases across the
632 global south, potentially confounding the effects of MDAs. Global climate models reveal an ever-
633 changing pattern of land surface temperature, rainfall and vegetation cover across the surface of the
634 planet [233]. Thus, contemporaneous environmental changes could act to decrease transmission,
635 giving the illusion of successful MDA programmes.
636
637 Multiple laboratory and field observations, modelling exercises and meta-analyses have identified
638 important abiotic and biotic factors that govern the free-living and vector-borne stages of parasites
639 (see Section 1). Seasonally-variable environments are also important in determining the aggregation
640 of animal parasites (e.g. [234]) and have been shown to determine the distribution of malaria and
641 hookworm in some regions [235,236]. What remains unknown, however, is how patterns of global
642 change across decadal scales have influenced the transmission of parasitic infections. Whilst the
643 substantial post-1997 downturn in malaria infections has undoubtedly been accelerated by large-
644 scale control interventions, environmental changes that have reduced the vector population or
645 climate-sensitive parasite life-stages over extended periods may have also contributed. Droughts in
646 Africa are increasingly common [237], and we cannot exclude the possibility that prolonged
647 periods of low-rainfall have contributed to the downturn in transmission of malaria and other
648 parasitic infections, given the reliance of vectors and parasite transmission stages on water
649 availability.
650
651 This last point illustrates how cryptic factors continue to be influential. Most campaigns do not
652 routinely collect individual patient data once the delivery programme is established, and without
653 this it is not easily possible to differentiate the effects of MDA from those of environmental change.
654 MDA programmes themselves are stressors on host-parasite systems as result of imposing selection
655 pressures, altering host microbiota, disrupting life-cycles of vectors and intermediate hosts that
656 have co-evolved with parasites. Thus the impacts of these programmes are also modified by
657 combinations of abiotic and biotic factors (Fig. 1).
658

In cases where pharmaceutical interventions have had a clearer effect in reducing infectious disease prevalence, the challenge is now to understand whether this success will be sustained over time in the context of environmental change. River blindness, caused by the nematode *Onchocerca volvulus* carrying the *Wolbachia* bacteria [238], was introduced into South America by the *Simulium* black fly, which has been treated with periodic ivermectin administration since 1991. Although it is unlikely that river blindness will ever be eradicated globally, prevalence has now fallen to 4% of those at risk in the endemic population [219]; however, the implications of changing environments for the long-term efficacy of this and other treatment programmes are not well understood.

Evolutionary implications of control programmes

There are many possible challenges to the sustainability of MDA programmes, including the potential reduction in impact over a decadal scale due to evolution. The impetus for MDA campaigns stems from mathematical models of parasite life cycles first developed in the second half of the twentieth century (e.g. [239]), which have an underlying assumption that neither parasites nor vectors undergo significant evolution that would reduce the impact of intervention. More recent attempts that consider either diminished drug efficacy, and/or greater transmission rate lead to the common output that MDA programmes do not eliminate parasite populations even after decades of intervention. Evolution of the parasite and/or vector populations along a specific life-history trajectory has not been a common feature of these models.

In terms of evidence from wildlife studies, it has been argued that long-term control programmes may do more harm than good, since any treatment imposes a selection pressure on a parasite population. Some treatments may promote the evolution of more virulent pathogen strains [240] as illustrated by evolving rodent malaria parasites in mice immunized with a candidate human malaria vaccine [241]. This in turn might lead to enhanced transmission as shown in chickens vaccinated against Marek's disease virus [242]. Poulin [192] suggests that shortening the duration of parasite infection by drug treatment negates any advantage to a parasite being long-lived, and survivorship of a parasite may trade-off against other traits which affect infectivity, such as age to maturity and fecundity (see Section 2) resulting in an increase of transmission. Virulence-longevity trade-offs might explain increased horizontal transmission of some diseases on hospital wards [243], while nematodes of horses appear to respond to drug treatment by shortening their development period in the host [244]. According to Day and Read [245], the optimal approach for combating the evolution of drug resistance is to use the highest, safe, dose or the lowest, effective dose. High dose medications are effective only if all pathogens can be killed (as in the case of HIV). If a small number of microbes are likely to evade treatment (already resistant to treatment, or if drug

695 resistance arises *de novo*), then high doses of medication may allow resistant microbes to survive
696 and spread by the very act of killing off drug-sensitive microbes [245].
697
698 A key feature of parasite control programmes is that they are based on retrospective evidence of the
699 hazard or risk. Programmes typically re-visit periodically the same communities, distributing
700 medicines and implementing other intervention measures (e.g. bed nets). Areas within control
701 programmes are susceptible to the effects of the stressors discussed above, plus the added stressor
702 of the selective pressure imposed by the intervention. One consequence of this 'perfect storm' of
703 stressors may be the generation of so-called 'hotspots' of transmission [246]. This hypothetical
704 effect does not preclude a reduction in transmission intensity, but does imply that individuals who
705 are normally resident in areas of intervention are persistently exposed and harbouring infection.
706 Other interpretations of the hot-spot observations are possible – including the lack of engagement
707 or disenfranchisement with control programmes.
708
709 **Section 5: Conclusion**
710 The changes in parasite ecology and epidemiology discussed above (Sections 1 and 2) have
711 profound implications for the monitoring and control of health in managed systems, which must
712 themselves adapt if altered challenges are to be attenuated. The subtle, 'covert' ways in which
713 multiple drivers of global change can affect parasite transmission are complex when considering
714 individual stressors, whilst the impact of interacting stressors on future disease risk remains largely
715 unknown. The current practice of making iterative changes in management strategy, based on
716 accumulated evidence of infection patterns, is too static to keep up with the increasing uncertainty
717 around transmission patterns. At the same time, advances in information and communication
718 technology open up new data collection modalities. Organising and applying such data streams
719 could provide novel and powerful ways of gathering real-time understanding of changing
720 transmission, and adapting control practices accordingly.
721
722 The zenith of adaptive management would track and react to not only parasite transmission but also
723 evolutionary processes, including those of host populations, such that transmission functions are re-
724 evaluated as life history parameters change (see Sections 3 and 4 above). This would require
725 repeated confrontation of alternative transmission models with available data, and inferring shifts in
726 key parameters from model fits. In principle this is already possible, but automation of this process
727 and the availability of sufficient, robust and timely data present challenges to implementation.
728 Citizen Science has been leveraged to gather real-time information on the distribution of invasive
729 plant diseases [247], while mobile phone networks have been used effectively to gather data on the
730 changing epidemiology of diseases in livestock and humans (e.g. [101,248,249]). Involved

731 professionals such as farmers and veterinary laboratories are also a source of specific surveillance
732 data [250], which could be collated more quickly to track epidemiological patterns, and update
733 models accordingly. For example, confirmed diagnoses of *Nematodirus* infections in lambs are
734 currently used to populate web-accessible maps that are updated daily during high-risk periods
735 (www.scops.org.uk). Linking these data with dynamic models, modified according to observations,
736 for example arising from bet-hedging (see Section 3), would further increase our capacity to assess
737 disease risk.
738
739 A major challenge in dynamic data-driven model fitting is the reliability of data collected from
740 disparate sources, and not always by professionals using verified methods. However, the decline of
741 expensive, centrally funded monitoring stations and systems, both for meteorological and disease
742 data, limit the alternatives. Networks of privately collected meteorological data (e.g. forecast.io) are
743 increasingly available and may usefully compensate for the loss of, and sometimes exceed the
744 capability of, official sources. Similar networks for data on phenology or even parasitic infections
745 could be envisaged. Separate observational or experimental data will always be needed and can
746 constrain fitted parameters within plausible ranges, or select parameters most likely to be open to
747 parasite evolution. Models of parasite transmission dynamics that are validated, updated with shifts
748 in epidemiology and evolution, and whose outputs are accessible to end users, could form the
749 backbone of a new wave of decision support systems that maximise the opportunities afforded by
750 advances in modelling and new sources of data. In addition, by involving the public in disease
751 monitoring, we can promote disease awareness, improving social-ecological resilience [251].
752
753 We suggest that a truly predictive understanding of the effects of global change on parasite
754 transmission will therefore need to incorporate the evolutionary consequences of changes imposed
755 by combinations of abiotic and biotic stressors acting at various locations under conditions of
756 migration, habitat loss and fragmentation. These are themselves difficult to predict, especially since
757 the experiments required to fuel such predictions would tend to remove 'extraneous' variation that
758 could actually be core to complex evolutionary trajectories under global change. Reverse
759 engineering of models to estimate parameter alterations, which are needed to maintain parasite
760 fitness under change scenarios, and then assessing these predictions for biological plausibility,
761 could provide more adaptive predictions. In any case, models of parasite population dynamics that
762 neglect the possibility of evolution of transmission strategies will have a short shelf-life under
763 global change, and greater attention ought to be paid to this challenging area.
764
765 Differences in transmission ecology, parasite life history and the ecology of intermediate hosts and
766 vectors will clearly play a key role in determining the sensitivity of infections to abiotic and biotic

767 stressors [196]. Monitoring these stressors at high spatial and temporal resolution, perhaps using
768 remotely sensed products (e.g. [252]), is likely to be of considerable help in improving our
769 understanding of how diseases might spread in the future. However, while there is a move away
770 from using keystone species in the general ecological field as early warning indicators of vulnerable
771 ecosystems in favour of monitoring the balance between diversity, functional groups and
772 connectivity, it would be naïve to take this approach for infectious diseases. The impact of
773 infectious diseases, particularly EIDs is context dependent ('the devil is in the detail'), and the
774 importance of particular parasite species and strains will change over space and time, and so at least
775 for the moment, targeted disease monitoring and surveillance at appropriate spatio-temporal
776 resolution is still necessary.
777
778 **Authors' Contributions**
779 All authors attended the 2015 British Ecological Society (BES) Transmission Research Retreat,
780 where initial ideas for this manuscript were conceived. JC coordinated writing and drafted the
781 article with MB. All authors contributed examples, text and approved the final version of the MS.
782
783 **Acknowledgments**
784 We thank the BES Parasite and Pathogen Ecology and Evolution Special Interest Group for
785 organizing the 2012 Ecology Meets Medicine Retreat and the 2015 Transmission Retreat.
786
787 *We have no competing interests.*
788
789 **Funding**
790 This work was partially funded by the Welsh Government and Higher Education Funding Council
791 for Wales through the Sêr Cymru National Research Network for Low Carbon, Energy and
792 Environment AquaWales Project.
793
794
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1485 Figure 1 Stress-response impacts on parasite control programmes
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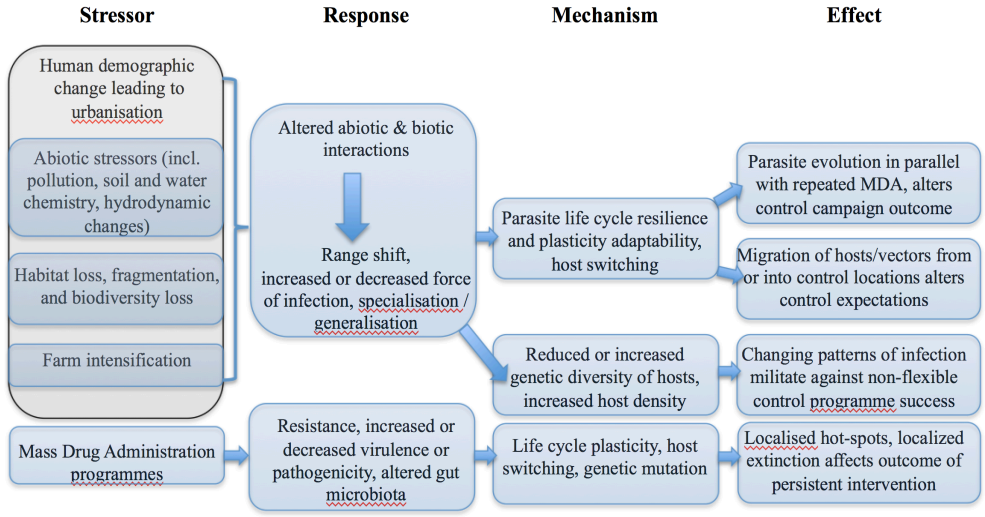

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