



Drivers of aquatic zoosporic parasites in natural and artificial environments

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Abstract

This review, based on more than 500 scientific studies published between 1880 and 2025, summarises current knowledge on the complex relationships between aquatic zoosporic parasites (AZP) and their hosts, the physical and chemical properties of the environment and the many surrounding organisms, to outline the key drivers of AZP diversity, abundance and infection success. It emphasises the urgent need to focus research on the influence of global climate and undesirable anthropogenically driven changes, such as eutrophication, pollution and biological invasions. Knowledge gaps are identified, along with research needs to decipher the diverse responses of AZP to ongoing changes in abiotic and biotic drivers. We believe that the summarised data and highlighted open research questions will benefit future research and help to better conserve natural biodiversity and manage artificial aquatic systems.

Keywords Abiotic drivers · Aquacultures · Biotic drivers · Climate change · Global warming · Host-parasite systems

Introduction

Parasitism and parasites are widely recognised as important drivers in the ecology and evolution of species, communities and ecosystems (Poulin 2007). They play an important role in maintaining the health of ecosystems through various ecological processes, including host population dynamics, energy flow and biodiversity (Hudson et al. 2006; Lafferty 2006; Miki et al. 2011). For centuries, the focus has traditionally been on terrestrial parasites that infest humans, animals or crops (e.g. Elsheikha and Khan 2011; Parwan et al. 2023), while far fewer studies have addressed their aquatic counterparts. In recent decades, however, parasites from different aquatic environments that have host-attached vegetative stages, but can reproduce asexually and infect the hosts through free-living flagellate spores (zoospores), and are therefore summarised under the term aquatic zoosporic parasites (AZP), have gained increasing attention in

the scientific community. AZP have been described from various marine and inland waters and numerous aquaculture systems (e.g. Kühn et al. 1996; Reñé et al. 2021b; Masigol et al. 2023; Seto et al. 2023; Van den Wyngaert et al. 2025). They play an important ecological role in nature by regulating host density, genetic diversity and evolution, but they can also have a negative impact on aquaculture, leading to severe production losses and increased operating costs (e.g. Lafferty et al. 2008; Rasconi et al. 2012, 2014, 2022; Gsell et al. 2013b; Carney and Lane 2014; Kagami et al. 2014; Grossart and Rojas-Jimenez 2016; Schmeller et al. 2018; Asatryan et al. 2019; Tao et al. 2020; Gavrilović et al. 2022). Some AZP significantly affect total global biodiversity as invasive species, some of which are considered among the most destructive and highly infectious pathogens, putting at risk global populations of a wide range of hosts. Such highly impactful AZP include the amphibian pathogen *Batrachochytrium dendrobatidis* and the crayfish plague *Aphanomyces astaci* (e.g. Sewell et al. 2021; Basso et al. 2026). Although their taxonomy is currently undergoing a

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comprehensive revision, recent studies suggest that AZP species belong to phylogenetic lineages of seven phylum-level clades of early diverging fungi, i.e. Blastocladiomycota, Chytridiomycota, Aphelidiomycota, Rozellomycota and three unknown phylum-level clades, but also Oomycota, Labyrinthulomycota and Phytomyxea, as well as protists from Alveolata (Perkinsea). (Scholz et al. 2016; Van den Wyngaert et al. 2025). The hosts of AZP also come from different phylogenetic groups – from autotrophic algae and plants to heterotrophic organisms such as invertebrates, fish and amphibians. In addition, the hosts belong to different ecological groups and occupy different ecological niches, from plankton to benthos, and their spatial and temporal occurrence varies greatly, making the study of AZP even more complex.

Although the life cycles of host–parasite systems are closely linked, each participant can respond differently to changes in environmental variables (Gehman et al. 2018). The occurrence, maintenance and effectiveness of AZP infections are determined by a variety of ecological factors operating in aquatic ecosystems. Today, the health of these ecosystems is increasingly threatened by anthropogenic stressors, with climate change playing an increasingly important role: nowadays the Earth's average surface temperature has increased by 1.3–1.4 °C above the pre-industrial level (1850–1900) (Rodhe 2025), with changes in precipitation and a measured average sea level rise of 21–24 cm since 1880 (Lindsey 2022). Moreover, it has been shown that since 1970, global warming has been increasing at a roughly linear pace (Rohde 2025) and it is predicted that, depending on the level of greenhouse gas emissions (lowest possible or very high), at a warming of 1.5 °C, the global mean sea level would rise at least 0.3 m, or as much as 2 m above 2000 levels by 2100 (Lindsey 2022). In addition to this incremental warming with a gradual increase in temperature, climate change also manifests itself through episodic climate events such as heatwaves, which are increasing in frequency and severity (Claar and Wood 2020). It is widely believed that the rapid change in mean surface temperatures, with a projected increase of 1.4–5.8 °C over the next 75 years and increasing heat waves (Nelson 2005), is not only changing weather patterns but also disrupting the usual balance in nature. All gradual and pulse-like changes are altering aquatic habitats at an unprecedented rate and inevitably affecting their inhabitants in diverse and complex ways. There is growing concern about the role of climate change in influencing the life cycles and transmission patterns of AZP and their interactions with hosts. However, the specific relationship between climate change and AZP remains poorly understood. This knowledge gap emphasises the need for targeted research into how environmental changes affect their reproduction and spread, and what this means for the future health of aquatic ecosystems. In this

review, we summarise the current state of expertise, the gaps in knowledge and highlight further research topics on the key drivers of AZP biodiversity, abundance and distribution. Taking into account the specific physical and chemical characteristics of their most important habitat, we first summarise the most important abiotic factors and then turn to the diverse, nonlinear biological interactions that influence AZP (Fig. 1). This does not mean that we give less importance to biological driving forces. In addition to discussing AZP drivers with a focus on the effects of current and future environmental change, we pay particular attention to other undesirable but strongly anthropogenically enhanced events such as eutrophication and biological invasions (e.g. Wells and Flynn 2022). By understanding all these drivers, we can better predict the future risks of aquatic diseases and implement strategies to manage their impact on biodiversity and ecosystem health under changing climate and increased anthropogenic pressure.

Materials and methods

All information was compiled and reviewed using scientific databases such as Web of Science, Scopus, PubMed, Google Scholar, Research Gate and Academia. A large number of keywords were used, comprising the Latin and common names of the AZP and their hosts, combined with all terms relevant to the factors including specific organismal interactions. To keep the scope of the review manageable, we refer whenever possible to other reviews and summarising articles that provide more details and references. In this way, the review covers more than 500 scientific publications published in the period 1880–2025.

AZP terminology and classification has been harmonised with our recent publication on AZP biodiversity (Van den Wyngaert et al. 2025). Taxonomic affiliations and synonyms, both for the host and for the AZP, are only given when they appear for the first time in the text. All Latin names of AZP have been checked in the Index Fungorum and in the relevant modern taxonomic literature. For the term 'driver', we accept the Millennium Ecosystem Assessment (MA) definitions summarised by Nelson (2005, p. 74): 'A driver is any natural or human-induced factor that directly or indirectly causes a change in an ecosystem. A direct driver clearly influences ecosystem processes. An indirect driver has a more diffuse effect by modifying one or more direct drivers'. The definitions and order of drivers in the aquatic environment generally follow the classical hydrobiological sources (e.g. Hutchinson 1957; Wetzel 2001; Reynolds 2006). For each driver, the available information is organised in terms of hosts from autotrophs to heterotrophs and from freshwater to marine habitats.

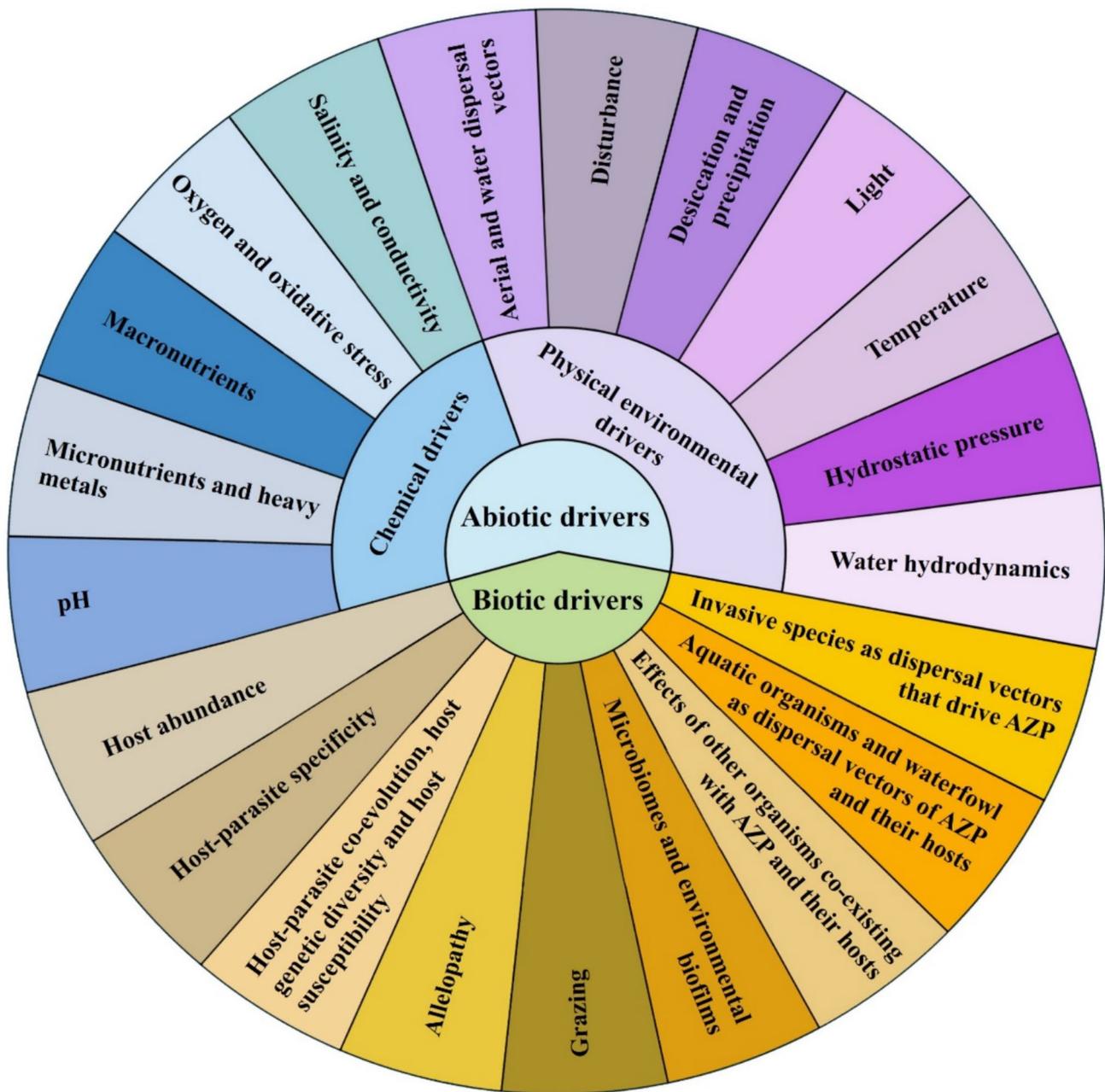


Fig. 1 Abiotic and biotic drivers of AZP

Results

Abiotic drivers

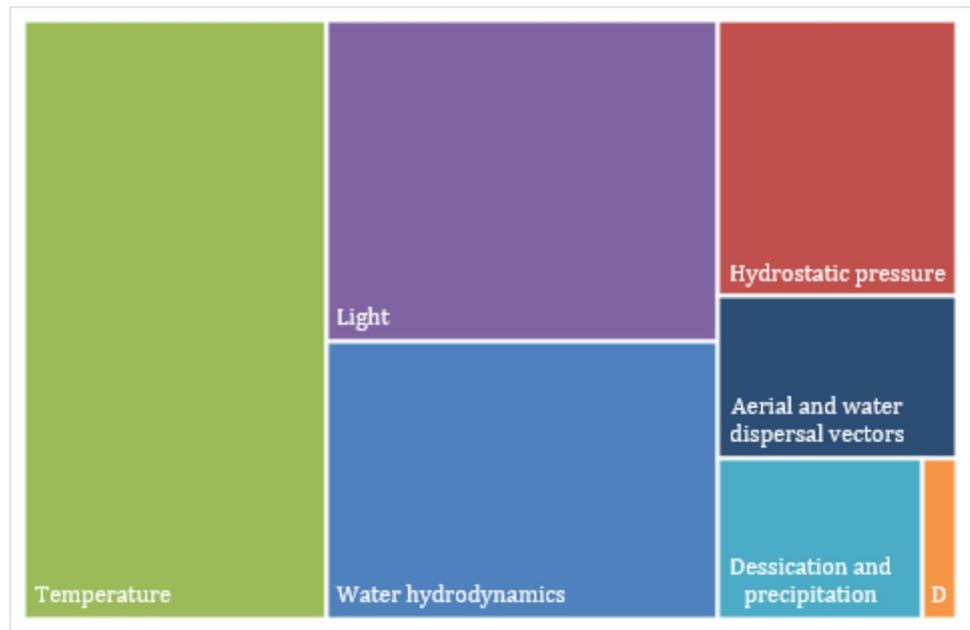
Aquatic habitats cover about 72% of our planet, with the oceans accounting for 97.4% of all water on Earth, while lakes, rivers and atmospheric water account for only 0.02% (Reynolds 2006). The aquatic environment can be characterised by different physical states and is subject to different physical forces which also influence the chemical

composition of the water layers. The most important abiotic factors that regulate the sinking, swimming, movement and dispersal of AZP and their hosts are summarised below.

Physical environmental drivers

The growth, reproduction and distribution of hosts and parasites and their interactions are strongly influenced by physical variables (Wolinska and King 2009; Kordas et al. 2011) – Fig. 2. Some environmental conditions can promote the

Fig. 2 Physical drivers of AZP organised according to the degree of knowledge based on the number of publications on the topic. *D* disturbances



spread of parasites and diseases (Altizer et al. 2013) while others can provide hosts with conditions that offer a refuge from disease (Gsell et al. 2023).

Water hydrodynamics In all aquatic habitats, water masses are in constant motion, which is represented by turbulent and laminar flows (Reynolds 2006). Turbulence is characterised by chaotic changes in pressure and flow velocity, whereas in laminar flow, water flows in parallel layers without interruption between them (Batchelor 2000); it can be attenuated or amplified by basin morphometry, counter-currents, internal waves, wind and possibly tides, as well as by deflection due to the Earth's rotation (Coriolis forces), and thus determines the vertical and horizontal displacement of organisms (Reynolds 1997, 2006). Solar heating, which accounts for 99% of the heat in water bodies, changes the temperature and density gradients in the water layers and also plays a role. With strong warming and little wind, there is a difference in large and deep waters between the mixed surface layer (epilimnion) and the underlying water (hypolimnion), which is limited by the so-called thermocline (Reynolds 2006). In this way, the epilimnion is completely dynamic, and the vertical distribution of hosts and their parasites should match the highly variable vertical differentiation of the water column. In contrast, the water in shallow pools is completely mixed by the wind (Reynolds 2006).

Water turbulence and laminar flow (current) are also important for the horizontal distribution of certain groups of aquatic host organisms and their parasites. Overall, much less evidence has been collected for the spatial occurrence and biodiversity of AZP on planktonic algae in running waters, such as streams and rivers, than for standing inland

waters (lakes, ponds, bogs, etc.) and oceans. The current and high turbulence may lead to lower species diversity and abundance of AZP in running waters, but the low number of AZP records from these habitats could also be explained by fewer field surveys. In any case, to the best of our knowledge, the available data refer to larger rivers with slower current velocities (Fabbro and Duivenvoorden 1996; Stoyneva 1998; Maier et al. 2013, 2016; Maier and Peterson 2014) and to some estuarine waters (Kazama 1972; Van Wichelen et al. 2006). Laboratory experiments have shown that high turbulence stimulates the growth rates of the large colonial pennate diatom *Asterionella formosa* in spring, allowing the alga to outcompete AZP and thus prevent the development of chytrid epidemics in rivers (Maier et al. 2013). In contrast, the reduction in flow velocity by barriers such as hydroelectric dams provides new niches with longer retention times, allowing diatoms to bloom and ensuring extended interaction time for chytrids to infect hosts (Maier et al. 2013; Maier and Peterson 2014). The lack of turbulence also enhances the attachment of AZP to host cells in marine habitats (Llaveria et al. 2010). Unlike algal AZP, parasites of insects, fishes and amphibians from different taxonomic groups have more often been recorded from rivers, streams and springs (e.g. Martin 1984; Czczuga and Godlewska 2001; Combe et al. 2022; Jacinto-Maldonado et al. 2024).

In standing waters, there have been a few studies on the influence of water turbulence on host–parasite interactions and infection success, showing that it can have different effects depending on the strength of the turbulence and the species considered. Intermediate levels of turbulence in a poorly mixed waterbody can increase encounter rates between hosts and parasites, raising the chance that a

parasite will find a suitable host (Zhang et al. 2024). AZP transmissions and infections are reduced under conditions of low turbulence and water stratification because stratification periods are often characterised by low nutrient concentrations and significantly lower phytoplankton abundance (Wetzel 2001), which reduces host density and thus the parasite infection rate. This was exemplified by consecutive epidemics of the chytrid *Zygorhizidium planktonicum* on the pennate diatom *Ulnaria acus* (Syn. *Synedra acus*), which occurred during the winter overturn, but were not observed in surface waters during spring stratification when host populations declined until the subsequent winter bloom (Doggett and Porter 1996). According to these authors, mixing of water layers can resuspend resting spores of chytrids, which are important for triggering AZP infections. Similarly, summer stratification strongly decreased AZP infections of zooplankton, such as *Daphnia pulicaria* by the chytrid *Polycaryum laeve*, while high turbulence in winter enhanced them (Johnson et al. 2009). In contrast, high water turbulence can significantly reduce chytrid infections in phytoplanktonic hosts that prefer stratified water, such as the prokaryotic blue-green algae (cyanoprokaryotes), more commonly known as cyanobacteria, as demonstrated by *Rhizophidium* sp. infections of filamentous *Planktothrix agardhii* (Wagner et al. 2023). Furthermore, in the case of AZP infections of *Planktothrix agardhii* in shallow water bodies, wind-driven high turbulence may hinder parasite transmission owing to impaired chemical communication or resuspension of sediments that physically prevent or reduce chytrid infections (McKindles et al. 2021b). The reduced infections in highly turbulent conditions could be due to the host-seeking behaviour of zoospores, as it was presumed in the case of the centric diatom *Coscinodiscus granii* and its parasitoid *Pirsonia diadema* (Kühn and Hofmann 1999). Strong turbulence, which leads to mixing of the water layers, can also result in the displacement of phytoplankton beyond the epilimnion in deep water bodies, where light limitation can reduce the transmission success of chytrids (Bruning 1991c, d) and other AZP.

Hydrostatic pressure Hydrostatic pressure results from gravity acting on all particles within a water column at a given depth and is determined by the weight of the overlying water per unit area (Lalli and Parsons 1997). As hydrostatic pressure is a downward force that increases with water depth, it is a significant abiotic factor influencing the host–parasite system in various waterbodies (Lalli and Parsons 1997). Therefore, actual depth is considered important for the distribution of aquatic organisms (Reynolds 1997, 2006; Xiao et al. 2021), and AZP are not exempt.

Few data support the presence of AZP in less explored benthic habitats strongly influenced by hydrostatic pressure. For example, AZP from different chytrid genera have

been detected in marine micro- and macrophytobenthos hosts (Karling 1943; Gleason et al. 2011a; Gachon et al. 2017; Scholz et al. 2014; Ilicic et al. 2022, 2024; Murúa et al. 2020, 2023). Chytridiomycota and other AZP groups, such as Perkinsea, were consistently detected in the sediment, and their community structure remained stable over time (Chambouvet et al. 2011; Orsi et al. 2013; Xu et al. 2016, 2018; Picard 2017; Reñé et al. 2021b), suggesting that sediments contain a reservoir of diverse AZP that persist in a latent phase until favourable conditions allow their activation (Picard 2017; Reñé et al. 2021b; Fernández-Valero et al. 2022).

Temperature Temperature is one of the key drivers of environmental changes (e.g. seasonal events) and physiological traits in all organisms, including hosts and their parasites (Lafferty 2009; Wolinska and King 2009; Frenken et al. 2017b; Rajarajan et al. 2025). The response to changes in this physical factor depends on the optimum temperature of both host and parasite, as well as the prevailing temperature in their shared environment (Gsell et al. 2023). Therefore, temperature changes can strongly influence the success of AZP infections. For example, according to the thermal mismatch hypothesis (Cohen et al. 2017), AZP infections can be more successful when temperature conditions differ from the host's thermal optimum. Shifts in the timing of seasonal events caused by global warming can lead to a phenological mismatch (Stenseth and Mysterud 2002). Such phenological asynchrony occurs when the seasonal peak of consumer demand for a resource does not correspond with the peak of its availability (Gronchi et al. 2023). Desynchronisation of species-specific responses was among the earliest observations of trophic relationships in aquatic systems (Gronchi et al. 2023), but it also affects host–parasite relationships. If the thermal tolerance range of the host is greater than that of its parasite, temperature extremes at low or high levels may provide a disease-free refuge for the host, either because the parasite is inactive (e.g. in a dormant stage) or because it dies, being unable to withstand the temperature (Gsell et al. 2023). Experiments with *Asterionella formosa* and its chytrid parasite *Zygorhizidium planktonicum* confirmed that the narrower thermal tolerance range of the AZP provides the host with 'cold' and 'hot' thermal refugia, resulting in very little or no infection (Van Donk and Ringelberg 1983; Ibelings et al. 2011; Gsell et al. 2013a). Differences in temperature response and the resulting cold refugia for the host from chytrid infection have also been described for another freshwater phytoplankton, the filamentous cyanobacterium *Planktothrix rubescens* (Rohrlack et al. 2015; Rohrlack 2018; Wierenga et al. 2022). The hot refugium for the host has been hypothesised for *Planktothrix agardhii*: its infections by the chytrid *Rhizophidium* sp. mainly occurred between 19 °C and 23 °C (Wagner et al. 2023), while high

temperatures (> 27 °C) inhibited the parasites (McKindles et al. 2021b).

The shape and amplitude of the thermal responses of algal hosts and AZP can be significantly altered by other interacting environmental factors, such as light or nutrient availability (Bruning 1991a, c, d; Wierenga et al. 2022). For example, warming accelerates the end of the spring bloom of the diatom *Synedra* sp., but the development of its chytrid parasites may have been restricted owing to increasing phosphorus limitation during the bloom (Frenken et al. 2017a). In cultures of the green freshwater flagellate alga *Haematococcus lacustris* (Syn. *Haematococcus pluvialis*), which is used for astaxanthin production, infections by *Paraphysoderma sedebokerense* (Blastocladiomycota) increased exponentially with temperature until an optimal temperature (30 °C) was reached, but could not establish at the low (< 15 °C) and high (40 °C) ends of the thermal tolerance range (Hoffman et al. 2008).

Temperature changes in the water column may not only create conditions that exceed the thermal tolerance limits of parasites and hosts but also influence the hydrodynamic environments in which they interact, as well as related light and nutrient regimes. Global warming, particularly the increasing frequency and intensity of heatwaves, is leading to greater summer stratification in both marine and deep freshwater ecosystems, affecting all food web components (e.g. Smale et al. 2019). These conditions reduce the vertical exchange of nutrients and passive phytoplankton, potentially resulting in greater nutrient depletion in the upper water layer (Boyce et al. 2010) and lower productivity, which in turn could lead to fewer and less nutritious hosts for parasites. While targeted studies on the effects of heatwaves on AZP are generally lacking, studies on other parasites have shown very different responses depending on factors such as the study area, host and parasite physiology and differences in their life cycles (Li et al. 2024).

With ongoing climatic changes, rising temperatures, changes in ice occurrence and shifts in the timing of snowmelt are also expected (Sharma et al. 2019). Decreasing snow and ice cover can directly lead to habitat loss for ice and snow algae and their associated parasites (Nakanishi et al. 2025). Loss of winter ice cover may also indirectly affect host–parasite interactions, as demonstrated by the loss of the cold-temperature refugium from chytrid infections of *Asterionella formosa* at low temperatures owing to warmer winters (Ibelings et al. 2011).

Temperature also controls dormancy and subsequent activation of resting spores, playing an important role in AZP periodicity. Both field and laboratory studies of the *Ulnaria acus*–*Zygorhizidium planctonicum* system revealed that resting spores of the parasite remained dormant at reduced hypolimnion temperatures (< 10 °C) but were activated and

reached 70–80% germinability when temperatures increased to 10–16 °C (Doggett and Porter 1996).

The effect of temperature on amphibian chytrids and their infection dynamics is particularly well documented (e.g. Boyle et al. 2003; Turner et al. 2021; Haver et al. 2022). Optimal growth of the chytrid *Batrachochytrium dendrobatidis* occurs between 17 °C and 25 °C, and water temperatures below 17 °C slow fungal growth rates (Piotrowski et al. 2004), while high temperatures (> 27 °C), as with algal chytrids, kill the parasite within a few hours to days (Johnson et al. 2003; Piotrowski et al. 2004). This enables amphibian tadpoles to overcome chytrid infections (Geiger et al. 2011). Temperature also determines the interactions of *Batrachochytrium dendrobatidis* with other organisms, such as zooplankton feeders and the protective microbiome of amphibian skin (Haver et al. 2022, see below), and may explain the importance of seasonality in its infection dynamics (Clare et al. 2016).

Whether parasites have a higher or lower thermal adaptive capacity to global warming than their hosts requires further research. Some studies show no parasite adaptation to elevated temperatures in phytoplankton–fungus systems (Schampera et al. 2022), while other research predicts that AZP can adapt faster than their large-bodied hosts owing to their smaller size and shorter generation times (e.g. Cohen et al. 2017; Gsell et al. 2023). In contrast to the faster adaptation rates of smaller parasites, larger hosts may have a broader temperature tolerance range, and their acclimatisation capacity is positively correlated with body size (Rohr et al. 2018). This greater acclimatisation capacity could allow large-bodied hosts to use a wider temperature tolerance range than their parasites (Rohr et al. 2018; Gsell et al. 2023), potentially giving them an advantage under global warming.

Light In ecological studies, the term ‘light’ usually refers to the visible solar electromagnetic radiation spectrum with wavelengths from 400 to 700–750 nm, also known as photosynthetically active radiation (PAR), and ultraviolet radiation (UVR) with wavelengths of 280–400 nm (Zhang et al. 2020). Light intensity, i.e. the number of photons arriving at a defined surface per unit time, is highly variable at each location, with daily and annual fluctuations (Reynolds 1997). After underwater light attenuation, the spectrum and intensity of light perceived by aquatic host–parasite systems depend on their vertical position in the water column (Reynolds 2006) and on water transparency, which is influenced by the presence of organic and inorganic particles or the associated water turbidity (Wetzel 2001). Thus, underwater PAR and UVR penetration and water transparency depend on all physical, chemical, biological and even watershed processes, but they also regulate many biological, physi-

cal and chemical processes in aquatic systems (Zhang et al. 2020).

Host–parasite interactions and coexistence show considerable variability owing to differing light requirements of the host algae, which vary not only between species from different evolutionary lineages and taxonomic groups but also between algae from the same taxonomic group and even from the same genus (Round 1981). For example, the chytrid *Rhizophyidium megarrhizum* infects two closely related phytoplanktonic species, *Planktothrix agardhii* and *Planktothrix rubescens*, which have contrasting light requirements and inhabit different depths: (1) *Planktothrix rubescens* is rich in the specific red pigment phycoerythrin, efficiently utilises green light (495–570 nm) and occurs in the metalimnion of stratified lakes; (2) in *Planktothrix agardhii*, the blue pigment phycocyanin predominates over phycoerythrin, allowing maximum growth rates in red light (620–750 nm), and the alga inhabits the epilimnion of stratified lakes or shallow waterbodies without stratification. However, experiments have shown that for both species, low light can provide a refugium from chytrid epidemics, while increased light availability can lead to the potential development of chytrids (Tao et al. 2020). Cells of the freshwater green alga *Tetradesmus obliquus* (Syn. *Scenedesmus obliquus*) infected with *Chytridium* sp. lysed only when exposed to blue light (Abeliovich and Dikbuck 1977). In addition to spectral composition, light intensity may affect algal hosts and thus their AZP. Experiments with *Planktothrix rubescens* showed that, although it grows optimally at higher light intensity under laboratory conditions, it can also grow at lower light intensity to avoid chytrid epidemics. This may explain its frequent occurrence in deeper lake layers (Rohrlack 2018; Wierenga et al. 2022). However, increased light availability can induce stress in other algal hosts, such as marine and ice diatoms, leading to a higher incidence of disease (Hassett and Gradinger 2016; Scholz et al. 2017).

Light can influence AZP directly or through indirect host-mediated effects and, together with temperature, can potentially explain seasonality in some AZP (Van den Wyngaert et al. 2022). In addition to the light requirements of the host, the occurrence and success of AZP infections in aquatic habitats also depend on the light requirements and behaviour of their zoospores, which play key roles in the dispersal, biogeographical range, ecological function and infection dynamics of AZP (Galindo et al. 2024). Light largely impacts the activity of photosynthetic microalgal and macroalgal hosts in both freshwater and marine habitats and likely influences the production of photosynthetic cues that chytrids may rely on to find suitable hosts through chemotaxis (Bruning 1991c; Muehlstein et al. 1988; Van den Wyngaert et al. 2014; Tao et al. 2020). Therefore, high light conditions and increased photosynthetic production can raise the probability of AZP finding hosts, and vice versa, as first demonstrated

in Bruning's (1991a) experiments with the host–parasite system *Asterionella formosa*–*Rhizophyidium planktonicum*. Other studies have also shown that low light levels or complete darkness hinder the attachment of zoospores to their hosts (Bruning 1991b, d; Scholz et al. 2017). Experiments with zoospores of *Rhizophyidium planktonicum* showed that they are more active in bright light than in dim light (Canter and Jaworski 1980) and rarely adhere to the cells of *Asterionella formosa* under very low light conditions or in complete darkness, but once attached in the light, they remain alive and continue their development in the dark (Canter and Jaworski 1981). However, increased light penetration stresses ice algae and raises chytrid disease incidence under reduced snow cover in the Arctic (Hassett and Gradinger 2016). Not only the intensity, but also the duration of light exposure and spectral composition influence the development of AZP zoospores and the success of their infection, with species-specific responses. For example, day length has been shown to correlate positively with infection success (Scholz et al. 2017), and the spectral composition of light can influence parasite transmission, as shown for the zoospores of *Ulkenia* sp. (Labyrinthulomycota) which exhibited positive phototaxis to blue light (440–480 nm) produced by the marine bacterium *Vibrio fischeri*, but only under low light intensity conditions (Amon and French 2004).

Regarding UVR, there are limited and contradictory results on the sensitivity of AZP. According to Hite et al. (2016), high UVR negatively affects amphibian chytrids and increases their mortality, but may also harm their zooplankton grazers, potentially increasing the infection success of AZP. In the field, UVR led to a decrease of *Batrachochytrium dendrobatidis* infections in tadpoles (Ortiz-Santaliestra et al. 2011), while it had no effect in cultures (Johnson et al. 2003). Experimental exposure of amphibian larvae to high UVR resulted in higher infection rates by this chytrid, and the larvae carried a greater infection burden than those not exposed to elevated UVR (Cramp et al. 2022). Given the complexity of co-factors (e.g. presence of pathogens, low temperatures, species sensitivity to radiation) that may influence the response of both hosts and parasites to UVR, these factors need to be investigated in more detail (Haver et al. 2022).

With the growing human population, the role of anthropogenic light at night, which disrupts the normal photoperiod and is known as light pollution, is increasing (Poulin 2023). Although it affects aquatic systems, its impact on aquatic organisms, especially on AZP, is almost completely ignored (Poulin 2023).

Desiccation and precipitation Desiccation and drought can affect host–parasite systems both on land and in aquatic habitats. This is particularly true for amphibians and their chytrid parasites (especially *Batrachochytrium dendroba-*

tidis), which are highly susceptible to desiccation (Johnson et al. 2003; Piotrowski et al. 2004). Wetter regions and rainy seasons have been associated with more severe cases of chytridiomycosis in amphibians (Puschendorf et al. 2011; Valencia-Aquilar et al. 2016), and amphibians in constant contact with water are more susceptible to chytrid infections, whereas chytridiomycosis and related mortality remain low in dry environments (Haver et al. 2022). It remains unclear whether this is solely due to the drought sensitivity of the amphibian host, the protective skin microbiome, or pathogen characteristics. The occurrence of dry periods in some seasonal water bodies potentially provides a pathogen-free refugium for amphibians if the parasites cannot withstand drought (Scheele et al. 2015). This may also be hypothesised for other host–parasite systems and may explain a greater prevalence of AZP infections in larger water bodies.

Disturbances Disturbance in aquatic systems includes all processes that disrupt established conditions and is recognised as an important factor increasing heterogeneity at all levels (Naselli-Flores et al. 2003). This particularly affects biodiversity, as described by Connell's (1978) Intermediate Disturbance Hypothesis (IDH), a key ecological theory explaining patterns of species diversity in relation to the frequency and intensity of disturbances in ecosystems, and proposing that biodiversity is maximised at intermediate levels of disturbance. Natural and anthropogenic disturbances often co-occur, and their effects are frequently synergistically compounded (Archer and Stokes 2000). However, disturbances are more commonly associated with various anthropogenic activities (mining, livestock, fishing, wastewater discharges, agriculture, industrialisation and urbanisation) that lead to rapid transformation and pollution of aquatic ecosystems, severely affecting their biodiversity. Although data on the effects of pollutants and other human activities have previously been collected regarding different AZP and their hosts (e.g. Schmeller et al. 2018), the adverse effects of disturbances and disruption of steady states on host–parasite interactions have rarely been discussed (e.g. Jacinto-Maldonado et al. 2024). This contrasts with the broad acceptance of disturbances, such as extreme weather and floods, in altering seasonal events and causing shifts in phytoplankton assemblages (e.g. Gaedeke and Sommer 1986; Reynolds et al. 1993; Stoyneva 2003; Znachor et al. 2008; Shabarova et al. 2021).

Dispersal via air and water Many algae that host AZP have a large geographical range, although there are generally no obvious links between localities. It is assumed that they achieve their wide distribution passively through physical factors such as wind, water currents, rain, animal vec-

tors or human activities (Kristiansen 1996b, 2008). The same transport vectors have been demonstrated for AZP, although little research has been conducted on the subject. For example, rainfall enhanced by wind can directly and indirectly influence AZP transmission and thus infection success. The detection of *Batrachochytrium dendrobatidis* zoospores in rainwater (Kolby et al. 2015) and fog samples (Prado et al. 2023) suggests that aerial transport may be possible and that its vectors may affect AZP from other hosts. In vitro experiments, in which *Batrachochytrium dendrobatidis* survived for up to 3 months in moist but sterile river sand, suggest that this AZP can be translocated by the movement of river sand (Johnson and Speare 2005).

Culture experiments with *Paraphysoderma sedebokereense* and its green algal hosts *Haematococcus lacustris* (Hoffman et al. 2008) and *Tetradesmus dimorphus* (Syn. *Scenedesmus dimorphus* – Letcher et al. 2016) have shown that the parasite's resting sporangia can retain infectivity after 3 to 4 weeks on agar (Hoffman et al. 2008). This enables widespread dispersion by water or wind after separation from the host until it finds a new suitable host (Letcher et al. 2016).

For aquatic organisms, water transport is the most common and long-distance dispersal mechanism (Kristiansen 1996a, b; Haver et al. 2022). The role of stream and river networks as natural transport corridors for various organisms, including pathogens and their hosts, has been summarised (Kristiansen 1996a, b; Meentemeyer et al. 2012; Rinaldo et al. 2018), although all mechanisms of abiotic dispersal of AZP, especially in complex landscapes, remain largely unknown (Haver et al. 2022). However, a review of current studies on the translocation of *Batrachochytrium dendrobatidis* by water movement in lakes or by downstream transport suggests that water connectivity is important in pathogen dispersal, particularly when facilitated by short distances between freshwater habitats (Haver et al. 2022).

Currently, a new form of facilitated water connectivity between non-adjointing waterbodies across various spatial scales, through the use of raw water transfers (RWTs), is rapidly increasing globally (Waine et al. 2024). These complex networks of purpose-built infrastructure – pipelines, tunnels and water supply canals – translocate large volumes of untreated freshwater and effectively create artificial corridors for aquatic organisms (Waine et al. 2024). Although RWTs pose a significant risk for the spread of invasive species and pathogens, there is limited awareness of RWTs among the scientific and nature conservation communities (Foster et al. 2025; Waine et al. 2025). At present, there are no studies on the spread of AZP by RWTs; however, considering their potential significant influence on biodiversity, we highlight these poorly

understood, artificial, novel potential dispersal pathways of AZP.

Chemical drivers

In addition to physical properties, the chemical composition of water is also important for aquatic life (Reynolds 1997). The solute content of water originates from several sources: the precipitation load, the geological basis of the catchment area, its geomorphological relief, and anthropogenic contributions to acidity, particulate matter and dissolved content. There is greater variation in the composition of major ions in inland waters than in marine systems (Reynolds 1997). The variability in the effects of different chemicals highlights the complexity of understanding the influence of water chemistry on host–AZP relationships (Chew et al. 2024) and its role as a driver of AZP diversity and abundance through its different components (Fig. 3).

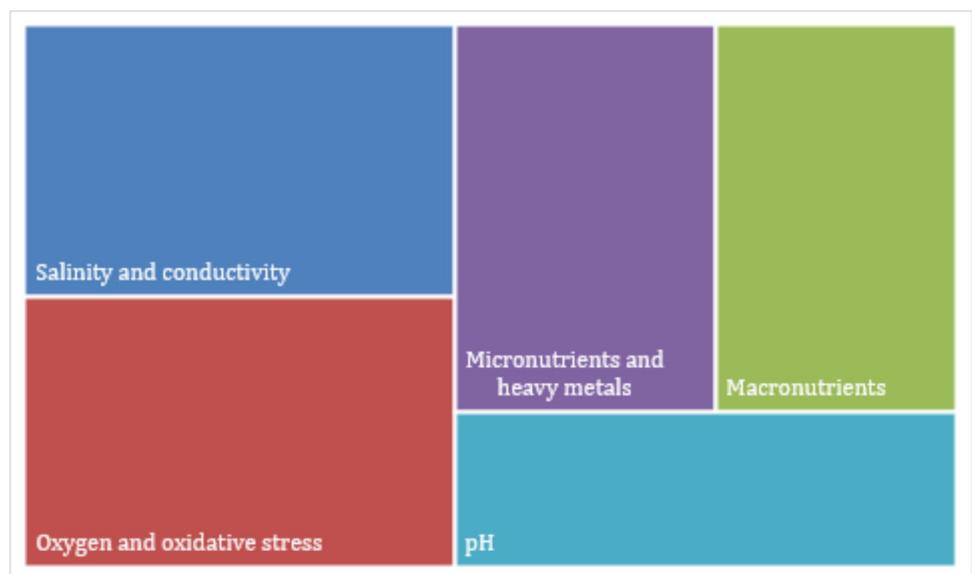
Salinity and conductivity Salinity, defined as the total ion content in the water column, is closely related to water conductivity, which measures the ability of a solution to conduct electricity (Wetzel 2001). Salinity is among the key environmental variables controlling aquatic mycota (Harrison and Jones 1971; Walker and White 2017; Rojas-Jimenez et al. 2019; Guo et al. 2023; Peng et al. 2024) because changes in water ion concentration can affect the physiology of resident organisms by causing osmotic stress or inducing variations in ambient electric fields. In general, fungi can survive the adverse effects of osmotic stress by altering their morphogenetic programme or making structural and metabolic changes (Duran et al. 2010). Most AZP, and chytrids in particular, are known to prefer habitats with low osmotic

potential, and very few live in marine or brackish waters (Sime-Ngando 2012; Rojas-Jimenez et al. 2019).

The distribution of fungi along salinity gradients depends on the physiological adaptations of different species. Recently, the application of molecular-genetic methods revealed that the overall diversity of marine micro-benthic chytrids is primarily influenced by salinity (Ilicic et al. 2024). The decreased salinity increased the diversity of chytrids on benthic diatoms (Ilicic et al. 2022), but on several occasions led to outbreaks of marine seagrass (*Zostera marina*) infections by *Labyrinthula zosterae* (Muehlstein 1998; Muehlstein et al. 1991). In general, marine species can tolerate a wider range of salinities compared with freshwater species (Jones 2022). For example, plasticity in adaptation to salinity is less pronounced in chytrids of freshwater phytoplankton hosts such as *Planktothrix rubescens* (McKindles et al. 2021b) and in amphibian chytrids such as *Batrachochytrium dendrobatidis* (Heard et al. 2015, 2018; Bolom-Huet et al. 2023; Chew et al. 2024). High salinity disrupts osmotic balance, decreases parasite infections and likely creates a salt niche for amphibian hosts to escape chytrid infections (Stockwell et al. 2012, 2015a, b). The mechanism of this beneficial effect of increased salinity for amphibians has been explained by the inhibition of zoospores and reduced disease transmission (Clulow et al. 2018). However, data from a few other studies show no influence of salinity on the infection dynamics of amphibian parasites (for details see Chew et al. 2024).

Oxygen and oxidative stress Oxygen is a widely recognised essential element for life and is important for the distribution of organisms in aquatic environments (Verberk et al. 2011; Deutsch et al. 2020; Lobo et al. 2024). Zoosporic parasites from freshwater, brackish and marine environments are con-

Fig. 3 Chemical drivers of AZP, organised according to the degree of knowledge based on the number of publications on the topic



sidered obligate aerobes whose growth is inhibited by low concentrations of dissolved oxygen (Gleason 1976; Barr 2001; Czczuga and Godlewska 2001; Gleason et al. 2007, 2008; Sime-Ngando 2012). In a single study, two chytrids, *Rhizophyidium sphaerotheca* and *Spizellomyces punctatus* (Syn. *Phlyctochytrium punctatum*), were shown to be facultative anaerobes, yet their growth was higher in oxygenated flasks (Goldstein 1960).

Low oxygen saturation levels and anaerobic conditions prevented the increase of *Chytridium* sp. infection and cell lysis of *Tetrademus obliquus*, while varying the oxygen concentration between 3.5 and 13 mg L⁻¹ had no effect on chytrid infections (Abeliovich and Dikbuck 1977). Laundon and Cunliffe (2021) expand on the role of oxygen for chytrids by highlighting several questions related to reactive oxygen species (ROS) in host and parasite biology, particularly in parasite development and virulence. The term ‘ROS’ refers to a large family of oxidants derived from molecular oxygen, often as byproducts of cellular metabolism (Sies et al. 2022). Their accumulation and release into cells increase in response to various unfavourable environmental factors (such as increased salinity, heat or cold stress and pathogen attack), but they can also act as toxic molecules with strong antioxidant activity (Camejo et al. 2016). Therefore, ROS were thought to accumulate in hosts from different taxonomic groups as a defence against fungal parasitism (Canter and Jaworski 1979; Santhosh and Neuhauser 2024). However, AZP can also utilise them, and the review by Camejo et al. (2016) summarises data on the major enzymatic systems involved in ROS generation and the role of ROS as cell signalling components and parasite developmental cues during plant–pathogen interactions.

By studying the interaction between *Haematococcus lacustris* and *Paraphysoderma sedebokerense*, Yan et al. (2022) identified oxidative degradation as a strategy for fungal infestation. Two metabolites, 3-hydroxyanthranilic acid and hordenine, were identified and shown to stimulate infection by inducing oxidative stress, causing gene expression disorder and pigment degradation in algal cells. These metabolites generated hydroxyl radicals that destroyed sub-cellular components of algal cells, such as the cell wall, membranes and pyrenoid, thus increasing the susceptibility of algal cells to infection (Yan et al. 2022). Although in many pathogenic fungi the strong antioxidant glutathione is a virulence factor that helps to neutralise oxidative stressors generated by host immune cells (Wangsanut and Pongpom 2022), its role in the oxidative stress tolerance of AZP has not been investigated.

Macronutrients The resource requirements of AZP, which obtain their nutrients directly from hosts, include essential components of cell protoplasts needed in large quantities for organism development and growth (Reynolds 2006). These

macronutrients include various forms of phosphorus (P) and nitrogen (N), which are constituents of nucleic acids and amino acids, respectively. Therefore, their availability in the aquatic environment is of great importance for the occurrence and maintenance of host–parasite systems. This availability, along with other environmental factors, plays an important role in the chytrid infection rate and thus in growth on algal hosts (Sime-Ngando 2012; Rasconi et al. 2012). For example, P limitation of *Asterionella formosa* led to reduced algal growth and thus facilitated the development of a *Rhizophyidium* epidemic despite reduced zoospore production and parasite growth rate (Bruning 1991a).

AZP are sensitive to changes in N supply (Frenken et al. 2021) and a relatively high availability of N compared with P may also stimulate chytrid infection rates on cyanobacteria (Frenken et al. 2017a). Experiments with cultures of *Haematococcus lacustris*–*Paraphysoderma sedebokerense* likely revealed an important role of N in the development of parasite infection (Asatryan et al. 2019). In this study, the vegetative flagellate cells of the alga remained uninfected, but infection of its two resting stages (green, palmelloid stage and red, astaxanthin-rich cyst stage) was observed, with epidemics developing faster in the red cells when N was present. Although *Paraphysoderma sedebokerense* has long been known as the ‘*Haematococcus* parasite’, it also has another green algal host, namely *Tetrademus dimorphus* (Letcher et al. 2016). This ‘new’ host was easily infected when grown in a medium with low N levels (Letcher et al. 2016).

Although the effect of nutrients on AZP has been less specifically studied, given the ecological requirements of the hosts, it can be hypothesised that nutrient enrichment increases algal host availability and thus AZP infectivity (Rasconi et al. 2012). Although parasitic fungi are found across the full range from poor in nutrients (oligotrophic) to nutrient-rich (eutrophic) waterbodies (Kagami et al. 2007a; Rasconi et al. 2012), in oligotrophic conditions, AZP associations (e.g. various species of chytrid parasites on green algae and diatoms) have been detected in small quantities and with a generally low prevalence of infection (Rasconi et al. 2012). Total N and P, together with dissolved carbon (C), were positively correlated with *Batrachochytrium dendrobatidis* concentrations on frog swabs (Battaglin et al. 2016). However, according to other authors, these nutrients do not impact pathogen growth and infection outcomes regarding amphibian AZP (Chew et al. 2024).

For decades, nutrient enrichment, particularly by N and P, has occurred in various aquatic environments in a process known as eutrophication, which affects species composition, productivity and elemental cycles of aquatic organisms, including AZP (Smith et al. 2006). The role of AZP in material transfer within aquatic food webs increases with nutrient availability, and their key role in eutrophic systems has been

demonstrated through both modelling and empirical results (Miki et al. 2011; Thongthaisong et al. 2022).

Micronutrients and heavy metals Many elements play a role in cell structures and their functions at relatively low concentrations (Reynolds 2006). Some of these micronutrients, such as sodium (Na), potassium (K), calcium (Ca), magnesium (Mg) and chlorine (Cl) are relatively abundant in water, while many others generally occur naturally in low concentrations (Reynolds 2006). The influence of these elements on AZP is poorly understood, although much knowledge has been accumulated about the hosts themselves. In sewage oxidation ponds, massive infections of *Tetrademus obliquus* by *Chytridium* sp. were triggered by low concentrations ($<10^{-2}$ M) of K^+ and Mg^{2+} , while Ca^{2+} and Na^+ had no effect (Abeliovich and Dikbuck 1977). The authors therefore hypothesised that it might be possible to prevent infection of mass cultures by replacing most Na^+ and Ca^{2+} cations with K^+ and Mg^{2+} .

In the aquatic environment, some metals such as manganese (Mn), iron (Fe), zinc (Zn), copper (Cu) and cobalt (Co), occur in quantities ranging from deficient to toxic concentrations (Reynolds 2006). Some of these metals are important for the homeostasis of fungi as they are required for various biochemical processes, usually as enzymatic cofactors. For example, Zn plays a crucial role as a large proportion of zinc-binding proteins are associated with critical functions including DNA binding, regulation of transcription, activity of transcription factors and response to chemical stimuli (Wilson et al. 2012; Staats et al. 2013), while Cu is necessary for the activation of metalloproteins (Festa and Thiele 2011). However, there is inconsistency in the reported impacts of both these metals on amphibian chytrids (e.g. Threlfall et al. 2008; Boisvert and Davidson 2011; Chew et al. 2024). At the same time, copper sulphate ($CuSO_4$) can act as fungicide against algal and amphibian chytrids (Fott 1967; Boisvert and Davidson 2011; Carney and Lane 2014). Fe is bound to many proteins and in this form is essential for metabolic pathways and physiological functions of living cells (Kontoghiorghes and Kontoghiorghes 2020).

Cadmium (Cd) is generally considered a toxic metal with no biological significance, although its catalytic role for carbonic anhydrase enzymes in marine diatoms has recently been discovered (Alterio et al. 2015). Cd tolerance in *Batrachochytrium dendrobatidis* appears to be controlled by the availability of glutathione, as since its depletion significantly affects the parasite's tolerance to Cd and silver (Ag) and moderately affects tolerance to Cu, Zn and lead (Pb) (Webb et al. 2024a, b).

Zoospores, which have a cell membrane rather than a cell wall, may be more sensitive than other AZP stages, such as the thick-walled sporangium (e.g. Berger et al. 2005). The metals mentioned above are transported in aquatic habitats

mainly by surface run-off, which also exposes waterbodies to various agrochemical pollutants such as pesticides, fertilisers and veterinary pharmaceuticals. This diffuse pollution from agriculture is considered one of the main pressures on European waters (EEA 2024). Many organic micropollutants, including pesticides, are harmful to various organisms, including amphibians and their parasites (e.g. Johnson et al. 2003; Haver et al. 2022). However, it must be considered that while a particular compound may affect the parasite, it may have even more detrimental effects on the host, increasing its susceptibility to infection (Chew et al. 2024).

In addition to all the inorganic elements mentioned above, micronutrients also include certain vitamins, such as thiamine, biotin and cyanocobalamin (Reynolds 2006). Growth of the chytrid *Rhizophyidium sphaerotheca* has been shown to depend on thiamine (Goldstein 1960). Some AZP may require vitamins, as demonstrated in a study on the oomycetes *Lagenidium callinectes* and *Haliphthoros milfordensis* in marine crustaceans (Bahnweg and Blang 1980).

pH The concentration of hydrogen ions (H^+) in a solution, traditionally measured as pH (Jensen 2004), is used to determine the acidity ($pH < 7$) or basicity ($pH > 7$) of aqueous solutions (Covington et al. 1985; Lim 2006). pH and closely related dissolved variables, such as dissolved organic matter, influence the biomass of aquatic microfungi (e.g. Jørgensen and Stepanauskas 2008). Recently, a substantial proportion of the carbon dioxide (CO_2) emitted by humans into the atmosphere has been absorbed by the oceans, progressively reducing their pH in a process known as 'ocean acidification' (Caldeira and Wickett 2003; Sabine et al. 2004; Ma et al. 2023). The decrease in pH is 0.1 units (from 8.2 to 8.1) compared with pre-industrial levels (Gruber et al. 2019), with some studies suggesting a further decrease to 7.8 by 2100 (Gattuso et al. 2015). Such changes are thought to favour the growth of marine fungi, some of which parasitise less resilient hosts, highlighting the need to assess the risk of increasing fungal infections in an ocean acidification scenario (Krause et al. 2013). In general, zoosporic fungi, and chytrids in particular, have been shown to tolerate low pH but not extremely high alkaline conditions (Gleason et al. 2008, 2010, 2011b). In cultures, growth of zoosporic strains was confirmed across the pH range 2.9–11.2; most isolates grow at pH between 4.7 and 8.9 but survived for 7 days at both pH 2.9 and pH 11.2 (Gleason et al. 2010). According to this study, many zoosporic fungi appear well adapted to a wider range of pH values than those found in the environments from which they were isolated, and, after brief exposure to pH extremes, they recover quickly.

An acidic cultivation strategy was developed to prevent contamination of *Haematococcus lacustris* cultures by *Paraphysoderma sedebokerensis* (Hwang et al. 2019). Instead of the commonly used neutral pH, an acidic condition ($pH = 4$)

was applied during cultivation, resulting in significant inhibition of fungal contamination. The authors claim that this may be due to acid-induced denaturation of a surface protein of the parasite, which plays an important role in the parasite's recognition of the host.

According to some studies, amphibian chytrids have a very low tolerance to pH values outside the range of 6–8 (Piotrowski et al. 2004; Johnson and Speare 2005). However, inconsistency exists regarding the impacts of pH on their infection dynamics: one study showed increased infection in more alkaline conditions (pH > 6.5) (Kärvemo et al. 2018), while others reported high infection prevalence and intensity at low pH values (Battaglin et al. 2016; Valencia-Aguilar et al. 2016) or found no impact of pH (Chew et al. 2024).

Biotic drivers

Although parasitism is one of the most common consumer strategies among organisms (Lafferty et al. 2008) and a common ecological interaction in most aquatic environments (e.g. Sime-Ngando 2012), AZP can be influenced by many biological factors and interactions. The main biological drivers (Fig. 4) are reviewed below.

Host abundance

The presence and development of AZP infections depend primarily on the distribution of their hosts (e.g. Sime-Ngando 2012; Ilicic and Grossart 2022). Many AZP infect algal species from different taxonomic groups (Table 1), and evidence of them has been collected since the late nineteenth century (e.g. Wildeman 1890). Most data collected concern phytoplankton (e.g. Karpov et al. 2014; Frenken et al. 2017b; Van den Wyngaert et al. 2022), where hosts in

freshwater are mainly representatives of green algae, diatoms and cyanobacteria, but also include desmids, zygnemalaeans, dinoflagellates, yellow-green algae and glaucocystophytes (Table 1). In marine environments, phytoplankton hosts are predominantly dinoflagellates and diatoms, while benthic communities include a wide range of macroalgal hosts such as green algae from the class Ulvophyceae and brown algae, commonly known as kelp (and their associated microalgae – Fernández-Valero et al. 2023), as well as microbenthic diatoms (Table 1). AZP also infect green algae that develop on snow and ice, such as *Ancylonema nordenskiöldii*, *Ancylonema alaskana*, *Sanguina nivaloides* and *Chloromonas* sp., as well as microbenthic diatoms (Table 1). AZP not only infect algae in natural waters but are also widespread in mass algal cultures, where they frequently infect chlorophytes such as *Chlamydomonas*, *Haematococcus* and *Scenedesmus* s.l. (Table 1). It must be emphasised that AZP infect not only the vegetative algal cells but also some of their reproductive structures (e.g. sporangia and receptacles of brown algae; Karling 1943; Müller et al. 1999; Gachon et al. 2017) or resting stages, which are formed in large quantities to survive unfavourable conditions. Among these, infections have been reported on the cysts of the dinoflagellate *Ceratium hirundinella* (Canter 1961; Canter and Heaney 1984) and on the akinetes of cyanobacteria *Sphaerospermopsis aphanizomenoides* (Skuja 1948; Stoyneva-Gärtner et al. 2023) and *Dolichospermum macrosporum* (Syn. *Anabaena macrospora* – Sime-Ngando 2012, Gerphagnon et al. 2013, 2017). Other cells, such as *Dolichospermum* sp. heterocysts (cyanobacterial specialised nitrogen fixing cells) may also be infected by AZP (chytrids of the order Lobulomycetales) causing disturbances in the nitrogen fixation process (Xu et al. 2025). Despite their different taxonomic affiliations, the presence of susceptible

Fig. 4 Biotic drivers of AZP organised according to the degree of knowledge based on the number of publications on the topic. AOW aquatic organisms and waterfowl as dispersal vectors of AZP

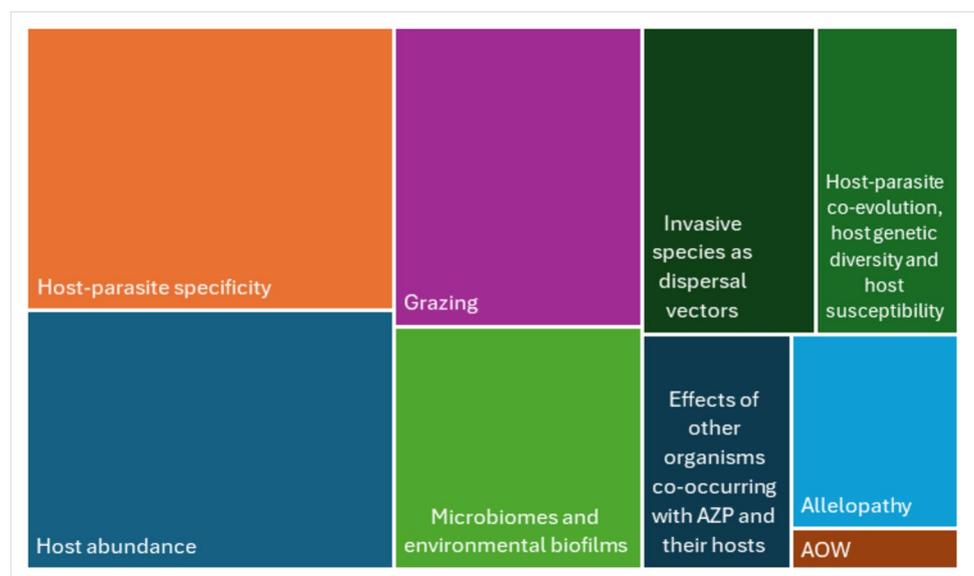


Table 1 Algal hosts of AZP from different aquatic communities

Aquatic community	Taxonomic group(s)	Refs.
Freshwater phytoplankton	Green algae, diatoms and cyanobacteria	Sparrow 1960; Canter 1951, 1967, 1972; Canter and Lund 1951, 1953; Batko 1975; Canter and Jaworski 1979, 1980, 1981, 1982, 1986; Sen 1988a, b; Canter and Beakes 1992; Canter-Lund and Lund 1995; Kagami et al. 2007a; Hanic et al. 2009; Sime-Ngando 2012; Gerphagnon et al. 2013, 2015; Gsell et al. 2013c; Voigt et al. 2013; Gleason et al. 2015; Takano et al. 2008; Van Wichelen et al. 2010, 2016; Sønstebo and Rohrlack 2011; Danz and Quandt 2023; Seto et al. 2020, 2023; Stoyneva-Gärtner et al. 2023; Wagner et al. 2023; Stoyneva-Gärtner and Uzunov 2024; Xu et al. 2025
	Desmids	Canter 1961; Cook 1963; Canter and Lund 1969; Canter-Lund and Lund 1995; Kagami and Urabe 2002; Rasconi et al. 2011; Van den Wyngaert et al. 2017, 2023; Seto et al. 2023; Stoyneva-Gärtner and Uzunov 2024
	Zygnematales	Canter-Lund and Lund 1995; Seto et al. 2023; Stoyneva-Gärtner and Uzunov 2024
	Dinoflagellates	Alster and Zohary 2007
	Yellow-green algae	Gromov et al. 1999; Karpov et al. 2014
	Glaucocystophytes	Seto et al. 2023
Freshwater phytoenthos	Green and yellow-green algae	Torruella et al. 2018; Stoyneva-Gärtner and Uzunov 2024
Marine phytoplankton	Dinoflagellates and diatoms	Johnson 1967; Hanic et al. 2009; Chambouvet et al. 2011; Lepelletier et al. 2014; Gleason et al. 2015; Scholz et al. 2017; Reñé et al. 2021a, b; Fernández-Valero et al. 2022
Marine macrobenthos	Green algae	Muehlstein et al. 1988; Ilicic et al. 2024
	Brown algae (kelp)	Karling 1943; Gleason et al. 2011a; Gachon et al. 2017; Murúa et al. 2020, 2023
Marine microbenthos	Diatoms	Muehlstein et al. 1988; Ilicic et al. 2024
Snow and ice algae	Green algae and diatoms	Hassett and Gradinger 2016; Fiołka et al. 2021; Kobayashi et al. 2023; Nakanishi et al. 2025
Mass freshwater algal cultures	Green algae	Fott 1967; Ilkov 1975; Shin et al. 2001; Hoffman et al. 2008; Han et al. 2013; Letcher et al. 2013, 2016; Carney and Lane 2014
Marine aquacultures	Red algae	Badis et al. 2019, 2020

hosts in the environment allows the parasites to develop, transmit and increase their abundance.

Hosts of AZP also include aquatic vascular plants, such as seagrasses (*Zostera marina*—Muehlstein 1998; Muehlstein et al. 1991) and their pollen (e.g. Sparrow 1960; Stoyneva

et al. 2013; Seto et al. 2023; Stoyneva-Gärtner and Uzunov 2024), while heterotrophic hosts include marine and freshwater amoebae and invertebrates (including rotifers, nematodes, mites, crustaceans and insects), amphibians and fish (Table 2). AZP of these groups infect not only adults, but

Table 2 Heterotrophic hosts of AZP from different taxonomic groups

Taxonomic group	Refs.
Protozoa (Amoebae)	Sparrow 1960; Corsaro et al. 2014a, b
Invertebrates (rotifers, nematodes, mites, crustaceans and insects)	Prowse 1954; Sparrow 1960; Green 1974; Batko 1975; Whisler et al. 1975; Bahnweg and Blang 1980; Burns 1980, 1985; Seymour et al. 1984; Martin 1987; Lord and Fukuda 1990; Glockling 1998; Gleason et al. 2008; Do Amaral et al. 2013; Valois and Burns 2016; Garcia et al. 2020; Masigol et al. 2023; Seto et al. 2023; Stoyneva-Gärtner and Uzunov 2024
Amphibians	Berger et al. 1998, 2005; Czczuga et al. 1998a; Longcore et al. 1999; Fisher and Garner 2007, 2020; Hyatt et al. 2007; Fisher et al. 2012
Fish	Stirling 1880; Shanor and Saslow 1944; Florinskaya 1971; Ferguson and Ride 1980; Chien 1981; Lartseva 1986; Frick and Reinhold 1987; Dudka et al. 1989; Czczuga and Muszynska 1998, 1999a, b; Lartseva and Dudka 1990; Hatai and Hoshiai 1992; Czczuga and Godlewska 2001; Chukanhom and Hatai 2004; Fisher et al. 2012; Masigol et al. 2023

also eggs and various developmental stages such as larvae and tadpoles (e.g. Chukanhom and Hatai 2004; Seto et al. 2023).

It is widely recognised that parasite fitness depends on successful transfer from one host to another (Shaw et al. 2020) which requires host densities exceeding a critical threshold necessary for effective transmission and parasite development (Holt et al. 2003). Peaks in the abundance of susceptible algal hosts are indeed associated with peaks in the abundance of related parasitic species (e.g. Holfeld 2000; Ibelings et al. 2011; Gutiérrez et al. 2016; Taylor and Cunliffe 2016). Such patterns have been observed in various environments, from lakes (Rasconi et al. 2012) to marine settings (Hanic et al. 2009; Fernández-Valero et al. 2022, 2023), and are typical not only for classical AZP such as chytrids, but also for Perkinsia parasitoids (i.e. parasitic protists; Reñé et al. 2021b). The same applies to artificial freshwater systems for mass algal cultivation, where the chytrid *Rhizophyidium scenedesmi* (Syn. *Phlyctidium scenedesmi*), *Amoebophilidium protococcarum* (Cryptomycota) and the blastoclad *Paraphysoderma sedebokerensis* cause severe production losses in green microalgae *Scenedesmus* s.l. (e.g. Fott 1967; Ilkov 1975), *Tetradesmus dimorphus* (Letcher et al. 2013) and *Haematococcus lacustris* (Han et al. 2013), respectively.

The high prevalence of AZP infections during algal blooms directly affects algal growth and leads to the termination or suppression of the bloom (e.g. Canter and Lund 1951; Van Wichelen et al. 2010; Gsell et al. 2013a; Sime-Ngando 2012), which, in turn, usually reduces the abundance and infection success of the causative AZP (e.g. Holfeld 2000; Sime-Ngando 2012). The presence of AZP and successful infections can influence the composition, succession, dominance and size structure of phytoplankton communities not only directly, but also by affecting interspecific competition between hosts and other algal species (e.g. Canter and Lund 1951; Van Donk and Ringelberg 1983; Van Donk 1989; Ibelings et al. 2004; Grami et al. 2011; Rasconi et al. 2012, 2014; Sime-Ngando 2012; Kagami et al. 2014). This occurs because, by parasitising one species and causing its population to decline, AZP alter competition among phytoplankton species, leading to an increase in the population of one or more competing algae. Several examples of such size- and composition-altering replacements confirm the general temporal shift from infection-inhibited large pennate diatoms (e.g. 50–70 µm in length: *Asterionella*, *Fragilaria*) to small non-infected centrics (e.g. 10 µm in diameter: *Stephanodiscus*) in deep lakes (e.g. Van Donk and Ringelberg 1983; Sommer 1987; Rasconi et al. 2011; Sime-Ngando 2012). Spatial shift in phytoplankton dominance was observed in the Danube River, from the heavily chytrid-infected small centric diatom *Stephanodiscus hantzschii* to the larger, uninfected pennate diatoms *Nitzschia acicularis*

and *Diatoma elongata* (Stoyneva 1998). In marine waters, temporal replacement of large centric diatoms *Thalassiosira* and *Skeletonema* infected by chytrids by non-infected centric *Chaetoceros* was recorded (Gutiérrez et al. 2016). The results of physiological experiments with a widespread marine thraustochytrid and *Chaetoceros* suggest that parasitic systems target weaker phytoplankton individuals (Leshem et al. 2016) and thus improve population health (Laundon et al. 2021). Increased infections of stressed ice algae have also been recorded during field studies in Arctic marine systems (Hassett and Gradinger 2016). In contrast, other researchers claim that parasites grow better on healthy, non-stressed individuals, as has been shown for desmids and chytrids (Van den Wyngaert et al. 2014; Schampera et al. 2021).

It is worth noting that, in addition to free-swimming and chemotactic zoospores, some AZP can also reproduce by non-motile spores with thick cell walls (i.e. resting stages), which are not chemotactic and, after germination, usually need a high host density to infect new hosts (Gleason et al. 2015; Weldon et al. 2025).

A dependence of parasites on host density has been documented, though rarely, in zooplankton. For example, Valkanov (1931a, b) highlighted the more intensive development of *Hydatinophagus apsteinii* (Syn. *Aphanomyces hydatinae*) in the freshwater rotifer *Hydatina senta* during its mass growth following the snowmelt in February–March. The strong dependence of the zoospore–zooplankton interaction on high parasite efficiency has been indicated (Miki et al. 2011; Thongthaisong et al. 2022, 2025). Several studies show the selective pressure exerted by AZP from Blastocladiomycota (*Aphanomyces*, *Leptolegnia*, *Saprolegnia* and *Scoliolegnia*) on zooplankton populations of the genera *Boeckella*, *Daphnia* and *Parabroteas*, as well as on mosquito larvae (Whisler et al. 1974, 1975; Johnson et al. 2006; Gleason et al. 2008; Wolinska et al. 2008, 2009).

In other hosts, such as amphibians, density-dependent dynamics of infections has also been demonstrated. For example, experimental assessments of AZP disease transmission in *Rana mucosa* tadpoles have shown that the proportion of newly infected tadpoles increased with the number of previously infected individuals (Rachowicz and Briggs 2007). Similarly, a field experiment with tadpoles of *Alytes obstetricans* showed a rapid spread of infection once a single tadpole in a group was infected (Courtois et al. 2017). As these two species also produce overwintering larvae, they could act as super-spreaders, as their infected tadpoles usually do not die from the infection. This gives the pathogen time to multiply throughout the year and increase zoospore loads during winter in the ice-free, deeper parts of a lake, with potentially higher host densities during winter and early spring (Clare et al. 2016; Fernández-Beaskoetxea et al. 2016).

Fish mass mortality, particularly in salmonids, caused by excessive development of AZP has been known since the late nineteenth century (Stirling 1880). The AZP species most frequently responsible for losses in aquaculture and the wild are coregonids and acipenserids, mainly *Saprolegnia ferax*, *Saprolegnia monoica* and *Saprolegnia parasitica*, but also *Achlya americana*, *Achlya dubia*, *Achlya jlagellata*, *Achlya polyandra*, *Achlya proliferoides*, *Aphanomyces laevis*, *Dictyuchus sterilis*, *Leptolegnia caudata*, *Pythium proliferum*, and *Aphanomyces astaci* (see references in Table 2).

Host–parasite specificity

The extent to which parasites can exploit different host species, known as host specificity, is considered an important trait of parasites (Poulin et al. 2011; Frenken et al. 2017b). Some AZP are host-specific and infect only a single host or a few related host taxa, while others are generalists and infect multiple species (Glockling et al. 2014). Field studies have often identified single chytrid species infecting one host species or its specific parts (e.g. Canter 1949, 1953; Holfeld 1998; Ibelings et al. 2004; Rasconi et al. 2009), which is most likely due to limited knowledge on this topic and the infrequent observation of host–parasite pairs in nature. However, in some cases, such as for the hosts *Asterionella formosa* and *Planktothrix* sp., chytrid infection was specific to genotypes or chemotypes of the same species (De Bruin et al. 2004; Kyle et al. 2015). For the chytrid *Rhizophyidium algavorum*, a wide range of hosts from different taxonomic groups of algae (20 species from five green freshwater genera and one yellow–green alga) has been experimentally proved (Gromov et al. 1999), but some generalists infect hosts only within the same taxonomic group. For example, in marine environments, the parasitoid *Maranthos nigrum* (Perkinsea, Parvuluciferaceae) and the chytrid *Dinomyces arenysensis* can infect different genera of dinoflagellates (Lepelletier et al. 2014; Reñé et al. 2021a; Fernández-Valero et al. 2022, 2023). A broad host spectrum encompassing several orders of brown algae has also been demonstrated for the chytrid *Chytridium polysiphoniae*, oomycete *Eurychasma dicksonii* and plasmodiophorid *Maullinia ectocarpii* (Müller et al. 1999; Maier et al. 2000). Similarly, in freshwaters, *Chytridium versatile* can infect ten different diatom species (Sparrow 1960). Another known generalist is the chytrid *Batrachochytrium dendrobatidis*, which can infect hundreds of amphibian species (Scheele et al. 2019) and is considered a pan-zootic chytrid (Torres-Sánchez et al. 2022). The fish parasite *Sphaerothecum destruens* has been found in a wide range of fishes (Charrier et al. 2016; Gunn et al. 2024), and many fish AZP can also infect planktonic crustaceans (Czeczuga et al. 2000), benthic crustaceans (Czeczuga et al. 1999) and shellfish (Czeczuga

2000). *Aphanomyces astaci* also infects a broad spectrum of crustaceans and fish (Czeczuga and Godlewska 2001; Masigol et al. 2023). In addition to field observations, cross-infection experiments (Gutman et al. 2009; Van den Wyngaert et al. 2018) and molecular studies (Kagami et al. 2021) confirm the coexistence of both specialist and generalist parasitic chytrids.

Regarding the hosts, a similar occurrence of specialists and generalists can be observed. Some hosts are infested by a single species, although specificity and virulence may vary widely, as demonstrated by experiments in which strains of *Rhizophyidium* sp. infect only *Planktothrix* (McKindles et al. 2021a). In contrast, other hosts can harbour several parasitic species (e.g. Canter and Jaworski 1982; Letcher et al. 2013; Kagami et al. 2021). For example, *Asterionella formosa* has been found to be infected by more than two morphologically distinct chytrids (*Zygorhizidium* sp. and *Rhizophyidium* sp.) (Canter 1969; Van Donk and Ringelberg 1983), and several species of AZP of the toxic marine diatom *Pseudonitzschia pungens* have been identified (Hanic et al. 2009). Recently, three different parasitic chytrids, *Algomyces stechlinensis*, *Endocoenobium eudorinae* and *Dangeardia mamillata*, were isolated from a common volvocacean species, *Yamagishiella unicocca* (Van den Wyngaert et al. 2018).

Many studies have emphasised the importance of host species composition and size for parasite infectivity. In freshwater chytrids, many infected phytoplankton hosts are large, such as species of *Asterionella*, *Fragilaria*, *Staurastrum*, *Staurodesmus*, *Dolichospermum*, *Planktothrix* and *Sphaerospermopsis* (e.g. Canter 1969; Sen 1988a, b; Kagami et al. 2007a, b; Rasconi et al. 2009, 2012; Ibelings et al. 2011; Weisbrod et al. 2020; Stoyneva-Gärtner et al. 2023). Large hosts are easier to target because the contact rate between host and parasite is inherently higher (Canter and Jaworski 1981; Sime-Ngando 2012). Larger cells contain more resources (e.g. greater amounts of organic carbon) for parasite nutrition (Lund 1957; Holfeld 1998; Kagami et al. 2007a). In addition, large hosts are less susceptible to grazing (e.g. Stoyneva et al. 2007; Charalampous et al. 2021), making them more attractive for successful parasite development. Thus, it is not surprising that large algae can be heavily infected even at low population densities (e.g. Lund 1957; Holfeld 1998; Sime-Ngando 2012; Rasconi et al. 2012). In contrast, for smaller hosts, such as the centric diatom species of *Melosira* or *Cyclotella*, infection was generally below 3% (Canter and Lund 1969; Sen 1988a, b; Holfeld 1998; Rasconi et al. 2012). Smaller hosts represent the so-called r-type strategy organisms which have high turnover and growth rates and produce many offspring, which could be problematic for parasites with slower growth rates (Rasconi et al. 2012).

Host–parasite co-evolution, host genetic diversity and host susceptibility

Host–parasite interactions offer great potential for studying co-evolution mechanisms, as these interactions are generally very close and associated with high fitness costs (Thompson 2002). Co-evolution in host–parasite systems can occur in two ways, leading to different outcomes: (1) selective sweeps (i.e. successive fixation of beneficial mutations), which result in rapid gene evolution but low levels of standing genotypic variation, or (2) ‘Red Queen dynamics’, in which species must constantly evolve to avoid extinction and these sustained genotype frequency oscillations lead to long-term maintenance of genotypic diversity (Woolhouse et al. 2002; Frenken et al. 2017b). At the same time, many terrestrial studies indicate that high genetic diversity in host populations results in decreased parasite burdens (e.g. Gibson and Nguyen 2021). In aquatic environments, similar effects in preventing or at least slowing parasitic adaptation due to host genetic diversity have already been observed in the host–parasite system of *Asterionella formosa* and *Zygorhizidium planktonicum* (De Bruin et al. 2008).

The experimental study of *Rhizophyidium megarrhizum* and various cyanobacterial strains, such as *Planktothrix agardhii* and *Planktothrix rubescens*, demonstrated that host genetic diversity increases resistance to disease, but can also be considered an important driver leading to rapid, sometimes novel and diverse, adaptations of the parasite to efficiently overcome host defences (Agha et al. 2018). Genetic diversification of natural host populations, promoted by high prevalence of parasitism, was demonstrated by studying genotypes of *Asterionella formosa* populations in three spring blooms under high and low parasite pressure, respectively (Gsell et al. 2013b).

Host–parasite co-evolution was demonstrated in experiments with two chytrid strains, Chy-Lys2009 and Chy-Kol2008, and their *Planktothrix* hosts (Sønstebo and Rohrlack 2011). Results from this study showed the strong selection pressure of parasites on host population, leading to its subdivision into coexisting chemotypes with periodic shifts in their composition, which in turn acts as a driving force resulting in AZP polymorphism. Further experiments with mono- and polycultures of different *Planktothrix agardhii* strains exposed to infection by several chytrid isolates demonstrated the importance of host specificity (genotypic or phenotypic determination of susceptibility) for parasite infection success (McKindles et al. 2023). In addition, the same study highlighted specific interactions between AZP and their hosts when involving diverse organismic communities allowing for co-occurrence, co-infection and competition between parasites.

Specificity and differential susceptibility of algal hosts to AZP were also demonstrated by in situ experiments with

four marine diatoms, which showed a positive correlation between the abundance of chytrids (*Chytridium* sp. and *Rhizophyidium* sp.) and that of their hosts (*Chaetoceros* sp., *Navicula* sp. and *Rhizosolenia* sp.), while there was no significant correlation with the abundance of *Nitzschia* sp. (Scholz et al. 2017). Experiments with infections of two bloom-forming colonial freshwater cyanobacteria (*Microcystis aeruginosa* and *Microcystis viridis*) showed a generally higher susceptibility of *Microcystis viridis* (85% infection prevalence) to *Chytridium microcystoides* (Van Wichelen et al. 2010). Recent studies by Van den Wyngaert et al. (2023) have also shown differences in susceptibility to infection between the original desmid host *Staurastrum* sp. and the experimentally infected desmid *Staurodesmus* sp. by the chytrid *Staurastromyces oculus*. This may reflect the co-evolutionary history and adaptation of this chytrid and its sympatric host *Staurastrum* sp. as an important driving force.

The susceptibility of the algal host population to parasite infection may vary depending on environmental factors (e.g. temperature—Gsell et al. 2022), as well as various other factors, ranging from compatibility between specific strains to the production of physical barriers and defences or the transition to resistant life cycle stages or morphologies. The specific infection relationships of host species depend on parasite genotype and important morphological characteristics of both hosts and parasites, which influence the infection process and shape host–parasite specificity (Sun et al. 2023). Laboratory experiments with the dinoflagellate *Alexandrium minutum* and its parasite *Parvilucifera sinerae* (Perkinsea) suggested that genes involved in infectivity and subsequent regulation of infection intensities could potentially be involved in co-evolution (Räberg et al. 2014).

Genetic studies on *Batrachochytrium dendrobatidis* from 14 amphibian hosts identified groups of differentially expressed genes among host environments with similar infection outcomes, demonstrating that this parasite exhibits plastic infection strategies when challenged by hosts with varying disease susceptibility (Torres-Sánchez et al. 2022). This study shows that host environments can exert different selection pressures, and that the success of multihost pathogens is promoted by an evolutionary factor such as plasticity of gene expression. According to available evidence for the co-evolution of *Batrachochytrium dendrobatidis* and its hosts, host susceptibility depends on the parasite’s geographical distribution, with some hosts able to recover from parasite outbreaks. Parasite genotypes also vary in their geographical distribution and virulence, leading to geographical patterns of host genotype distribution (O’Hanlon et al. 2018; Carvalho et al. 2024).

The functional role of mucilage as a physical barrier against AZP has been demonstrated as one of the defence features of phytoplankton. For example, with its mucilage

envelope, *Staurodesmus* is protected from infection by *Staurastromyces oculus*, but without it, it becomes susceptible (Van den Wyngaert et al. 2023). In contrast, other studies have shown the ability of some chytrid AZP, such as *Dangardia mamillata*, *Algomyces stechlinensis*, *Endocoenobium eudorinae*, *Phlyctochytrium planicorne* and *Rhizophyidium ubiqetum*, to penetrate the mucilage layer of coenobial flagellate chlorophyte algae (Canter 1950; Van den Wyngaert et al. 2018) or desmids (Canter 1961) with their rhizoids, probably using carbohydrate-active enzymes that can degrade complex mucilage polymers (Lange et al. 2019; Van den Wyngaert et al. 2023).

The role of autophagy as a defence pathway during infection, considered a key player in host–pathogen interactions (Evans et al. 2018), has been demonstrated in studies on marine brown algae (Murúa et al. 2020), as well as on freshwater diatoms (Canter and Jaworski 1979) and chlorophytes (Seto et al. 2020).

Allelopathy

In aquatic ecosystems, chemical interactions such as allelopathy play key roles in regulating various physiological processes and biological interactions that strongly influence population structure, community organisation and ecosystem function (Hay 2009, 2014). Allelopathy is a biological phenomenon in which one organism produces one or more chemical compounds, called allelochemicals, that can influence the germination, growth, reproduction and even survival of other organisms (Stamp 2003; Sentenac et al. 2022). The term was originally created for direct biochemical interactions between neighbouring plants, but was later broadened to include all possible chemical interactions between organisms (Whittaker and Feeny 1971; Rice 1984; Stamp 2003). In this broader sense, organisms can not only be harmed but also benefit from allelochemicals (Whittaker and Feeny 1971; Stamp 2003). Thus, allelopathy can have both positive and negative effects, either promoting symbiotic relationships or eliminating competitors, thereby influencing species interactions and ecological balance. Although allelochemicals are primarily studied in terrestrial plants, they are also a recurrent focus of interest in aquatic ecosystems, particularly in interactions with planktonic parasites (Vallet et al. 2023; Vallet 2024). Aquatic parasites can alter host chemical cues, affecting community dynamics and impacting organisms at all trophic levels (e.g. Friesen and Detwiler 2021).

Research has shown that certain metabolic products of the host act as specific chemical signals—chemoattractants—that attract parasites. The zoospores of AZP (e.g. chytrids, blastoclads) are chemotactic and can quickly locate uninfected host cells (Sparrow 1960; Canter and Lund 1951; Gleason and Lilje 2009; Voyles et al. 2011; Van

den Wyngaert et al. 2014; Wang et al. 2021). Zoospores of chytrid parasites can be attracted to cell extracts of stressed hosts and to certain dissolved carbohydrates (Scholz et al. 2017), or to small amino acids and carbohydrates that are by-products of photosynthesis (Muehlstein et al. 1988). In addition, the volatile compounds released by infected and uninfected hosts display different metabolic profiles, such as a reduction in antifungal compounds in chytrid-infected cells (Yoneya et al. 2021).

The excretion of primary or secondary metabolites as a host defence mechanism has been widely discussed in relation to algae and their production of toxins (phycotoxins), with all accumulated contradictory data arguing against the primacy and predominance of this protective function (Gärtner et al. 2021). A targeted study on *Planktothrix rubescens* has shown that its toxins (microcystins, microviridins and/or anabaenopeptins) can reduce susceptibility to chytrid infections and thus increase the host's chances of survival (Rohrlack et al. 2013). In contrast, observations on the microcystin-producing *Microcystis aeruginosa* and *Microcystis viridis* indicate that the toxins are not involved in host resistance to *Chytridium microcystoides* (Van Wichelen et al. 2010). Similar conclusions can be drawn from experiments on the variation of microcystin production in *Planktothrix agardhii* and *Planktothrix rubescens* which show that chytrid infection leads to limited or no release of this toxin (Agha et al. 2022). Weisbrod et al. (2020) reported selective parasitism of the chytrid *Rhizosiphon crassum* on a straight morphotype of the filamentous, bloom-forming cyanobacterium *Dolichospermum* sp. with increased toxin production (microcystins and anabaenopeptins), while the coiled morphotype of this alga was not infected despite the constant presence of chytrids in the environment. The authors interpreted this as a defence reaction to parasitism at the expense of lower competitiveness of the straight form compared with the co-occurring coiled morphotype. Abscisic acid is a chemical produced by *Tetradesmus obliquus* to protect the host from infection by the chytrid *Rhizophyidium scenedesmi* (Puneva and Christov 2004; Puneva 2006).

Grazing

In both freshwater and marine environments, phytoplankters are exposed to protozoans and metazoans that exert temporally and spatially variable feeding pressure (Reynolds 2006; Lüring 2021). Thus, phytoplankton must withstand grazing (Reynolds 2012), which is considered one of the largest loss processes and strongest selective forces on phytoplankton (Smetacek 2001; Lüring 2021). The altered structures of phytoplankton communities due to grazing are well documented. Such changes in the occurrence and availability of hosts inevitably lead to shifts in their AZP and infection success. For example, *Asterionella formosa* is less affected by

Rhizophyidium planctonicum in summer, when zooplankton grazers are more active, than in winter (Rasconi et al. 2011). The presence of the generalist zooplankton grazer *Daphnia* can significantly reduce the intensity of chytrid infection on phytoplankton (Kagami et al. 2004; Lürling 2021).

In addition to these indirect effects of grazing on parasites through changes in host availability, evidence shows that grazing can also have direct effects on AZP by consuming their free-living zoospores (Thongthaisong et al. 2022), which are highly susceptible owing to their lack of cell walls (Gleason and Lilje 2009). Although AZP, and chytrids in particular, are generally thought to produce too many spores to be fully controlled by zooplankton grazing alone (Frenken et al. 2019), several studies have shown that crustaceans, ciliates and rotifers actively ingest zoospores in freshwaters (Kagami et al. 2007a, b, 2014; Gleason and Lilje 2009; Buck et al. 2011; Hamilton et al. 2012; Searle et al. 2013; Schmeller et al. 2014; Agha et al. 2016; Frenken et al. 2018; Thongthaisong et al. 2025). In marine systems, grazing on zoospores can also lead to fluctuations in the relative abundance of chytrids (Johansson and Coats 2002). Parasitic zoospores are a valuable food source with suitable dimensions for both types of zooplankton: selective or specialised grazers (e.g. rotifers) and non-selective or generalist grazers (e.g. filter-feeding cladocerans) (Gleason et al. 2009; Sime-Ngando 2012; Frenken et al. 2016; Lürling 2021; Abonyi et al. 2024). It is estimated that chytrid zoospores can supplement the zooplankton diet in freshwaters by up to 60% (Rasconi et al. 2014). Results also suggest that small zooplankton benefit nutritionally from chytrid zoospores during blooms of large diatoms, as they prevent losses from downstream transport and retain organic matter within the system (Maier et al. 2016). In the marine environment, next generation sequencing has shown that chytrids from diatoms account for a large proportion of sequence reads in the gut of the Arctic copepod *Calanus glacialis* (Cleary et al. 2017). Zoospores are not only an important food source in terms of quantity, but also contribute to food quality, as they transfer all the nutrients they have absorbed from large inedible algae directly to zooplankton (Gerphagnon et al. 2015). In addition, zoospores enhance trophic value by converting short-chain polyunsaturated fatty acids (PUFAs) from inedible hosts into long-chain PUFAs (Gerphagnon et al. 2019; Taube et al. 2019; Rasconi et al. 2020) or by enriching the biochemical composition of their hosts through de novo synthesis of sterols (Gerphagnon et al. 2019). Therefore, zoospores are considered a valuable food source for zooplankton and form an alternative trophic link between large inedible phytoplankton and zooplankton, known as the mycoloop (Kagami et al. 2007a, 2014; Miki et al. 2011; Frenken et al. 2020; Chen et al. 2024; Thongthaisong et al. 2022, 2025). Other notable trophic links involve bacteria and carbon transfer through the so-called fungal shunt (Klawonn

et al. 2021), including pollen, from which organic carbon, normally unavailable to zooplankton, is efficiently transferred via chytrid fungi to higher trophic levels (Gerphagnon et al. 2019). In this way, AZP can also control the population dynamics of many invertebrate grazers such as crustaceans, rotifers, nematodes, arachnids (e.g. mites) and insects (Gleason et al. 2008), particularly in benthos (benthic shortcut) (Attermeyer et al. 2013). Thus, AZP themselves may serve as a driving force that alters the abundance, distribution and availability of their hosts, with profound consequences for their own evolution.

Herbivory can indirectly affect phytoplankton infections, for example through fragmentation of hosts caused by zooplankton, which provides more new infection sites for chytrid parasites to colonise (Frenken et al. 2020). Grazers may also reduce epidemics of virulent parasites by selectively grazing on and consuming infected cells or organisms, thereby maintaining population health (Packer et al. 2003). However, this effect has not yet been demonstrated for AZP.

Microbiomes and environmental biofilms

The interactions between globally distributed communities enclosed in a gelatinous matrix, commonly referred to as biofilms (Sentenac et al. 2022), and AZP–host systems, as well as interactions between microbial communities coexisting with different AZP–host systems, may influence the occurrence and success of parasitic infections.

Interactions between algae and microbial communities have been studied mainly in the context of algal production systems, where algae coexist with various associated microorganisms (Dittami et al. 2014) that promote their growth (Hom et al. 2015) or support faster and more efficient removal of organic and inorganic wastes and hazardous substances (e.g. Su et al. 2012; Luo et al. 2014; Cavaliere et al. 2017). However, whether these associated or symbiotic microorganisms also help algae to ward off chytrid infections remains unclear. Significant changes in the native bacterial microbiome during AZP infection of algae were recorded in experiments with the freshwater green alga *Coelastrrella vacuolata* (Syn. *Scenedesmus vacuolatus*) and its endoparasite *Amoebophilidium protococcarum* (Hoeger et al. 2022). The observed shifts in the functional and metabolic profiles of the bacteria during infection were driven by the release of nutrients and metabolites from decaying algal cells, resulting in increased bacterial metabolism of amino acids, lipids, nucleotides and carbohydrates, suggesting that the microbiome can adapt its metabolism to algal host die-off.

Research on the interactions between *Batrachochytrium dendrobatidis* and microorganisms is particularly advanced, with growing evidence highlighting the important role of amphibian skin-associated microbial communities and those from biofilms in infection dynamics

and the likelihood of AZP occurrence (e.g. Woodhams et al. 2011; Bletz et al. 2013; Garner et al. 2016; Torres-Sánchez and Longo 2022; Loyau et al. 2024). Although in some cases the bacterial composition on the skin reflects the host and environment rather than any chytrid impact (Kruger 2020), it is broadly accepted that the severity of infection depends on the composition and/or richness of the skin microbiome (e.g. Brucker et al. 2008; Bates et al. 2018; Schmeller et al. 2022; Loyau et al. 2024). In addition, infection success is influenced by the abundance of chytrid-inhibitory taxa on amphibian skin (Bernardo-Cravo et al. 2020; Loyau et al. 2024) and by antifungal metabolites secreted by bacteria to inhibit parasite growth (e.g. Harris et al. 2006, 2009; Bell et al. 2013; Loudon et al. 2014; Woodhams et al. 2015; Niederle et al. 2019; Torres-Sánchez and Longo 2022; Ujszegi et al. 2023).

The effects of the microbiota on *Batrachochytrium dendrobatidis*, which prevent pathogen-induced morbidity and mortality, led to the formulation of the 'Adaptive Microbiome Hypothesis'. This hypothesis proposes that exposure to a pathogen can trigger the rapid proliferation of anti-pathogenic microbes, thereby reducing microbial diversity but increasing anti-pathogen function (Woodhams et al. 2023). This understanding of the individual host response may also apply to other AZP. However, our knowledge remains limited, and the recruitment process of protective microbes is largely unknown, though it may be related to host physiology and/or environmental factors (Davis et al. 2017; Flechas et al. 2019). Thus, the concept of the Disease Pyramid, which includes the host, pathogen, environmental factors and the microbiome, provides a comprehensive framework for future research into AZP (Bernardo-Cravo et al. 2020).

For *Batrachochytrium dendrobatidis*, it was further shown that the different compositions of biofilms in mountain lakes affect infection dynamics, and experiments confirmed that parasite zoospores are inactivated even by simple biofilms within a few hours (Sentenac et al. 2024). Besides biofilms associated with *Batrachochytrium dendrobatidis*, many other biofilms on amphibians may entrap zoospores, which are then consumed or displaced by the biofilm inhabitants or inactivated by biofilm metabolites (Chabaud et al. 2006; Rendueles and Ghigo 2012). Biofilm inhabitants may also include sessile or semi-sessile rotifers and ciliates, which have been shown to feed on AZP and chytrids in particular (Weitere et al. 2018; Mialet et al. 2013). Depending on their composition, biofilms can therefore be considered natural biological barriers that prevent AZP infections (Sentenac et al. 2024) and, in this respect, are considered biological drivers affecting AZP.

Effects of other organisms co-occurring with AZP and their hosts, co-infections and hyper-parasitism

In aquatic environments, as in other parts of the world, host-parasite systems are surrounded by numerous other organisms that influence them through various physical, chemical and biotic interactions, extending beyond grazing, symbiotic microbiomes or environmental biofilms. For example, high densities of planktonic green algae negatively affected the abundance of zoospores of *Batrachochytrium dendrobatidis*, suggesting physical interference or allelopathy (Searle et al. 2013).

Besides the interactions between host and AZP, other parasites can also develop on, or in, the host and thus influence the host's susceptibility to infection and potential mortality. This is the case with pathogens such as *Hannemania* mites, which also persist and develop within the spongy epidermis and cause various detrimental effects on amphibians, such as deformities, reduced foraging ability, loss of chemosensory function and reduced reproduction and survival (Jacinto-Maldonado et al. 2024). In addition, mites can transport toxic cerium oxide nanoparticles into the bodies of amphibians during the larval period, increasing mortality, causing behavioural changes and inhibiting amphibian growth (Jacinto-Maldonado et al. 2022). Currently, a significant correlation has been observed between the intensity of chytrid infection and the abundance of mites and amphibian species (especially in *Lithobates yavapaiensis*), particularly in anthropogenically disturbed waters (Jacinto-Maldonado et al. 2024). Similar modulations may occur with co-infections of *Batrachochytrium dendrobatidis* and Ranavirus, particularly Frog Virus 3 (FV3), which, although not common (Bosch et al. 2020), has been documented in various amphibian populations worldwide (Olori et al. 2018; Thumsová et al. 2024; Yoldas et al. 2024). Some studies suggest that the chytrid and the virus may interact synergistically, potentially increasing infection intensity and disease progression in co-infected hosts (De Jesús Andino et al. 2024; Thumsová et al. 2024).

Despite their small size, individual AZP cells are complex environments and can be hyper-parasitised by other parasites (Seto et al. 2023). Thus, hyper-parasitism may be another biological driver of AZP, as suggested after the discovery of chytrid or rozellid hyper-parasites that infect other parasitic chytrids or oomycetes on phytoplankton. Examples include *Rhizophyidium carpophilum* on endoparasitic *Olpidiopsis* infecting *Achlya*, *Chytridium parasiticum* or *Septosperma anomalum* on *Chytrium tabellariae* (Canter 1949) and on *Zygophlyctis asterionellae*, *Rozella parva* on *Zygorhizidium affluens* and *Rozella* spp. on chytrids or oomycetes (Sparrow 1932; Canter 1969; Canter and Jaworski 1986; Gleason et al. 2014; Frenken et al. 2017b; Letcher and Powell 2018), as well as additional chytrid and rozellid

strains on algal AZP currently identified and sequenced by Seto et al. (2023). The ecological importance of hyper-parasitism has been demonstrated in phytoplankton studies, where hyper-parasitism of the primary parasite may reduce its impact by decreasing or suppressing the release of spores (Canter-Lund and Lund 1995; Frenken et al. 2017b).

Aquatic organisms and waterfowl as dispersal vectors of AZP and their hosts

Dispersal is one of the most important drivers of biodiversity at both local and global levels and concerns both AZP and their hosts. In addition to physical dispersal vectors, aquatic organisms and waterfowl are important for the spatial distribution of AZP hosts such as algae (Kristiansen 1996a, b; Padišák et al. 2016). Following the classic studies by Proctor (1966) and Atkinson (1970, 1971, 1972, 1980), there has been little research demonstrating the importance of algal transport by birds over long distances along their migratory routes (Coesel et al. 1988; Cellamare 2010, 2013; Stoyneva 2016). Birds can also act as sentinels for potential disease outbreaks (Green and Elmberg 2014). In vitro studies of *Batrachochytrium dendrobatidis* have shown that it can survive 1–3 h of drying between transfers from sterile feathers to new culture media, supporting the idea of its possible translocation by birds (Johnson and Speare 2005). A few research papers have experimentally demonstrated the role of birds in the transport of aquatic invertebrates (for details see Charalambidou and Santamaría 2002; Figuerola et al. 2004; Green et al. 2023). Here, we would like to highlight the important findings of Brown (1933), which demonstrate that invertebrate reproduction can still occur after 26 h in the gut of birds and up to 14 days in the gut of amphibians and reptiles. This supports the concept of long-distance transport of aquatic invertebrates as AZP hosts. Aquatic insects (32 species of Odonata, Hemiptera, Diptera and Trichoptera in both larval and adult stages) were identified as dispersal vectors of 28 fish and crayfish AZP in six waterbodies with different trophic statuses (Czeczuga et al. 1998b, 1999b, 2000; Czeczuga and Godlewska 2001). The highest number of fish parasites developed on the water scorpion *Nepa cinerea* (16), while the lowest numbers (four species on each) were found on the larvae of the hawk dragonfly dragonfly *Anax imperator*, the phantom midge *Chaoborus cristallinus*, the fly *Paramormia decipiens* (Syn. *Pericoma decipiens*) and adult ground beetle *Hydroporus palustris*.

Invasive species as dispersal vectors that drive AZP

Non-native (also known as alien, non-indigenous or allochthonous) species are defined as those capable of surviving and reproducing when introduced outside their former or current range (Kokociński et al. 2017; Soto et al. 2024).

An invasive species is a widespread alien species that has an adverse effect on the invaded habitat, local ecosystems and species, thereby threatening local biodiversity (Colautti and MacIsaac 2004; Wilk-Woźniak and Najberek 2013; Kokociński et al. 2017). In the context of increasing globalisation and the associated exchange of goods (e.g. Stoyneva-Gärtner and Uzunov 2015), it is often difficult to distinguish between human-mediated movements, whether intentional or unintentional, and natural migration processes. Nevertheless, it appears likely that most current invasions are facilitated by global climate change (Walther et al. 2009).

The emergence of non-indigenous pathogenic AZP is an increasing threat to global health, biodiversity, food safety and the global economy (Fisher et al. 2012; Combe et al. 2022). When discussing AZP drivers, we consider invasive host species for their role as vectors in spreading their specific parasites. For example, *Aphanomyces astaci*, which causes crayfish plague in noble crayfish and many fish species, arrived in Europe from America via non-native crayfish imported for aquaculture and farming (Holdich et al. 2009; Svoboda et al. 2017; Martín-Torrijos et al. 2021). Owing to the high host mortality, *Aphanomyces* itself, as well as its non-native crayfish vectors, are listed by the IUCN as among the 100 worst invasive alien species in the world (Lowe et al. 2000).

Another prominent example of an AZP whose global presence has been driven by an invasive species is *Batrachochytrium dendrobatidis* (Voyles et al. 2009). It is currently distributed on all continents except Antarctica, where its amphibian hosts do not occur (Fisher et al. 2009). Vectors of this AZP and the related disease include the invasive African clawed frog *Xenopus laevis* and the American bullfrog *Lithobates catesbeianus* (Fisher et al. 2009; Scheele et al. 2019; Nie and Feng 2022; Carvalho et al. 2024). Parasite transport routes involve both indirect transmission via free-swimming zoospores and direct transmission of zoospores through host-to-host contact (Carvalho et al. 2024). Its sister species, *Batrachochytrium salamandrivorans*, is also considered an emerging fungal pathogen of salamanders and newts (Urodela) (Martel et al. 2014; EFSA 2018). Its origin is thought to be in East Asia, where it infects native salamanders without causing significant disease (Martel et al. 2014). Recently, it has been introduced into European amphibian populations, where it has caused severe declines, and in at least one case its spread has been linked to the presence of an invasive species (Lötters et al. 2020; Martel et al. 2020).

Within the non-fungal AZP, an emerging pathogen is *Sphaerothecum destruens*, an agent of a disease with high mortality among populations of cyprinids and salmonids, posing a threat to native European fishes and causing significant economic damage to their aquaculture (Charrier et al. 2016; Combe et al. 2022; Gozlan and Combe 2023). The most common biological driver of the pathogen worldwide

is the highly invasive Asian gudgeon, the small cyprinid *Pseudorasbora parva* (Charrier et al. 2016; Combe et al. 2022), which appeared in Europe from China (e.g. Sana et al. 2017) through multiple invasions (Brazier et al. 2021).

Discussion

During the past two decades, AZP have received increasing attention in the scientific community; however, our understanding of their physiology and ecology remains limited (Grossart and Rojas-Jimenez 2016; Grossart et al. 2019; Rasconi et al. 2022). This is hindered by the fact that we are far from knowing the true AZP biodiversity in all communities across various types of aquatic habitats and different geographical regions. Currently, research is dominated by studies in large lakes from temperate regions, while marine environments, deep-sea habitats and polar regions are gaining increasing attention. There is still little data from small water bodies, rivers and streams. Most AZP studies focus on chytrids and their phytoplankton or amphibian hosts, while much more observation and experimentation are needed for other host–parasite systems.

Analysis of the available literature reveals diversity of factors influencing the AZP success and that no single factor has been equally well studied for AZP across all taxonomic groups in different habitats and communities. Therefore, assessing the role of individual influencing factors and determining general patterns of AZP responses to environmental changes remains a challenge. The variability of the results obtained indicates that the effects of individual factors are often species-specific and may influence infection dynamics differently in various host–pathogen systems. For example, data collected to date on the effects of pH, salinity and heavy metals on different AZP are inconsistent and vary in a case-specific manner. All factors can either directly influence the diversity, distribution, growth and virulence of AZP or act in complex and often complicated ways in environments where host–parasite systems coexist with a variety of organisms. An additional complication in identifying major AZP drivers arises from the fact that, during their life cycles, AZP produce zoospores and non-motile resting stages that may have environmental requirements different from those of the adult stages. For example, AZP zoospores are naked and therefore more sensitive to factors such as drought, light, heavy metals and grazing (e.g. Gleason and Lilje 2009; Thongthaisong et al. 2022, 2025; Galindo et al. 2024). Regarding individual factors, most studies report on temperature effects on AZP. Although higher temperatures generally favour more infections (Gehman et al. 2018), many chytrids and some other AZP cannot grow at water temperatures above 23–27 °C (Boyle et al. 2003; Johnson et al. 2003; Piotrowski et al. 2004; McKindles et al. 2021b). Therefore,

future global warming, with an increasing frequency of heat extremes, is expected to have a significant impact on the population sizes and diversity of AZP in aquatic environments. However, most empirical data on parasite fitness and disease progression have been measured immediately after sudden temperature increases, ruling out the possibility of parasite adaptation to continuously increasing temperatures. This suggests that disease progression under global warming is largely determined by the thermal tolerance range of both host and parasite (Schampera et al. 2020). Their development is strongly affected by shifts in the timing of seasonal events caused by global warming, leading to the so-called phenological mismatch, with differences in the resource requirements of hosts and parasites (e.g. Stenseth and Mysterud 2002; Gsell et al. 2023; Gronchi et al. 2023).

Trends in the future development of different AZP and relevant epidemics are likely to differ if only water temperature is considered. However, temperature never acts alone, and the AZP response can be strongly influenced by other natural environmental factors or anthropogenic impacts. In projected future scenarios resulting from global change, environmental conditions of various aquatic habitats and their biodiversity, including hosts and AZP, are expected to change. Among these changes is the anticipated warming of large parts of the world, especially during the winter season and at high latitudes (IPCC 2014; Sharma et al. 2019). Reduced glacier areas and shifts in the timing and intensity of snow and ice melt will decrease habitats for cryophilic AZP–host systems, leading to reduced diversity. Conversely, such events can also affect these host–parasite systems by altering the light regime and reducing salinity (e.g. Hassett and Gradinger 2016; Scholz et al. 2017; Ilicic et al. 2022). Thus, the influencing factors are manifold, especially in the context of global climate change, and the same factor can affect different AZP not only directly and indirectly, but often in opposing ways. The difficulty in predicting specific responses of various AZP also arises from the fact that gradual and pulse warming effects differ in magnitude and seasonality depending on the study area (e.g. lakes versus open ocean), region (e.g. polar versus tropical) and water depth (e.g. surface versus deep water layers).

There is virtual consensus that rising temperatures due to global climate change are leading to the spread of algal blooms (Paerl and Huisman 2009; Griffith and Gobler 2020) and floating plant cover (Kleinschroth et al. 2021), resulting in higher host density and a corresponding increase in AZP infection success (e.g. Holfeld 2000; Reñé et al. 2021b; Fernández-Valero et al. 2022, 2023). Conversely, mass development of algae and plants reduces light availability in the water column, which can hinder zoospore attachment to hosts, leading to reduced infection success (Canter and Jaworski 1980; Bruning 1991b, c, d; Scholz et al. 2017). Temperature fluctuations can also lead to changes in water

stratification and the relevant spectral composition of available light: if the mixed surface layers become shallower during warming, AZP and their hosts will be exposed not only to more light, but also to more red or green light instead of blue light, which may favour parasite transmission and infection outcomes (e.g. Tao et al. 2020).

Climate change is expected to alter precipitation patterns both spatially and temporally, and increase the likelihood of extreme weather events, including heavy rainfall and droughts (IPCC 2014). Excessive water run-off from land to water can dilute ion concentrations, while seawater intrusion or a reduction in volume due to droughts can increase dissolved ion concentrations in freshwater (Kaushal et al. 2025). In both cases, such events may severely affect the physiology of both freshwater and marine AZP. If lower salinity promotes the development of AZP with lower salinity tolerance, there may also be a concomitant loss of halophilic AZP (e.g. Muehlstein 1998; Muehlstein et al. 1991). When discussing the impact of global warming and the expected drier conditions on AZP, it is important to emphasise their high susceptibility to desiccation, especially in amphibian parasites (e.g. Johnson et al. 2003; Haver et al. 2022). There is no doubt that increasing drought periods will affect waterbodies, especially smaller and temporary ones, resulting in a highly probable decline in host diversity and, subsequently, of AZP.

Another major problem for aquatic environments is eutrophication. Extensive literature already exists on its occurrence and consequences in freshwaters and coastal seas, and various approaches have been proposed to mitigate this generally undesirable but mainly human-induced phenomenon. Both the extent and impact of eutrophication, as well as the effectiveness of measures to prevent it, can vary greatly both spatially and temporally, but in all cases they affect aquatic life, including the distribution and abundance of AZP and their hosts. It should be noted that eutrophication increases pH to extreme levels owing to the depletion of dissolved inorganic carbon, exposing aquatic diversity, including AZP diversity, to the risk of species shifts or declines (Lobo et al. 2024). Elevated pH can impair the chemosensory abilities of organisms that rely on the perception of dissolved chemical cues for survival, as has been shown for predatory snails (Turner and Chislock 2010). Such ‘blinding’ may also be expected to affect the chemotactic, host-seeking zoospores of AZP (e.g. Sparrow 1960; Canter and Lund 1951; Muehlstein et al. 1988; Gleason and Lilje 2009; Wang et al. 2021). In addition, microbial decomposition of the organic matter from dense algal blooms caused by eutrophication leads to a strong reduction in dissolved oxygen, creating hypoxic or anoxic ‘dead zones’. The lack of oxygen negatively affects almost all aquatic organisms, including AZP, most of which are typical aerobes (e.g. Gleason 1976; Sime-Ngando 2012).

Eutrophication is also accompanied by significant changes in the structure of aquatic communities. As AZP use their hosts as the main source of nutrients, the effects of nutrient enrichment or depletion are usually more pronounced in the hosts and typically affect the AZP indirectly. Thus, increased P and N concentrations generally stimulate algal growth and promote their parasites (Sime-Ngando 2012; Rasconi et al. 2012). However, greater epidemic success can also occur following a reduction in host growth in cases of nutrient limitation (Bruning 1991a).

From all collected data, we conclude that the diversity, dominance, infection rates and success of AZP depend largely on the presence and abundance of their hosts and, therefore, on the drivers of host abundance and distribution. In some cases, factors affect both AZP and their hosts in similar ways but with different intensities. This has been shown in experiments where fungicides harmful to the parasites may also affect the host and increase infection outcomes (Chew et al. 2024). Evidence has also been collected, albeit less frequently, that a particular ecological factor can affect the parasites and their hosts in opposing ways and, when it has a negative effect on the host, it can increase AZP infection success. This has been exemplified by the different effects of UVR on amphibians and their chytrids (Cramp et al. 2022; Haver et al. 2022). Similarly, the overall effect of global warming is predicted to be rather negative for amphibians, but may benefit their pathogens (e.g. Cohen et al. 2017; Haver et al. 2022).

Conversely, certain environmental conditions may limit the performance of parasites more than that of their hosts, thereby providing the hosts with environmental refugia from disease, as shown for chytrids in amphibians during drought (Scheele et al. 2015) and for many phytoplankters and their parasites with different thermal tolerance ranges (e.g. Van Donk and Ringelberg 1983; Ibelings et al. 2011; Wierenga et al. 2022; Wagner et al. 2023). However, whether parasites have higher or lower thermal adaptability to global warming than their hosts remains unclear for most host–parasite pairs. Most authors currently assume that AZP can adapt faster than their large-bodied hosts owing to their smaller size and shorter generation times (e.g. Cohen et al. 2017; Gsell et al. 2023), although it is also recognised that larger hosts have the advantage of a wider temperature tolerance range and greater acclimatisation capacity (Rohr et al. 2018; Gsell et al. 2023). Larger hosts appear to be favoured by AZP, especially in phytoplankton (e.g. Rasconi et al. 2012), indicating the importance of host size for algal AZP occurrence and success. Whether size is important in other host–AZP systems remains to be determined.

While abiotic factors, which strongly influence AZP and their hosts, are better understood, knowledge of biotic factors, which also play an important role in AZP diversity and abundance, remains limited. Very little research has

addressed the potential for co-evolution, and little is known about the specific interactions between AZP and their hosts involving multiple organisms, including co-occurrence, co-infection and competition (e.g. De Bruin et al. 2008; Sønsteibø and Rohrlack 2011; Gsell et al. 2013b; Lepelletier et al. 2014; Råberg et al. 2014; Agha et al. 2018; O'Hanlon et al. 2018; Torres-Sánchez et al. 2022; McKindles et al. 2023; Sun et al. 2023; Carvalho et al. 2024). Regarding biotic factors, we always consider that AZP are not only affected by biological relationships, but can also influence the composition, succession, dominance and size structure of host communities, which in turn can alter their own diversity, abundance and infection success. Such influence by AZP can occur not only directly through detrimental effects, but also indirectly by affecting interspecific competition between hosts and other species, as has been demonstrated in phytoplankton (e.g. Canter and Lund 1951; Van Donk and Ringelberg 1983; Rasconi et al. 2012, 2014; Kagami et al. 2014).

Our synthesis identifies grazing as one of the most important biotic factors for AZP. Grazing directly reduces host availability (Kagami et al. 2004; Rasconi et al. 2011; Lürling 2021) or acts by consuming zoospores via a specific trophic link, the mycoloop (e.g. Kagami et al. 2007a). As zooplankton may benefit nutritionally from zoospores, which are abundant, valuable food of suitable size (e.g. Rasconi et al. 2014, 2020; Gerphagnon et al. 2015, 2019; Lürling 2021; Abonyi et al. 2024), this can increase populations of certain grazers, affecting the abundance, distribution and availability of algal hosts, which in turn influences the development of AZP. However, grazing can also have the opposite effect and increase infection through host fragmentation, leaving more sites available for parasite colonisation (Frenken et al. 2020).

The interactions between AZP and microbiomes from different biofilms appear to be another important factor in AZP infection dynamics. This is particularly true for amphibian chytrids and the microorganisms associated with amphibian skin (e.g. Woodhams et al. 2011; Schmeller et al. 2022; Loyau et al. 2024). As the antifungal metabolites excreted by these organisms can inhibit parasite growth (e.g. Harris et al. 2006, 2009; Ujszegi et al. 2023), both the Adaptive Microbiome Hypothesis (Woodhams et al. 2023) and the Disease Pyramid Concept (Bernardo-Cravo et al. 2020) provide a comprehensive framework for future research into AZP and their biological drivers. In addition to excreting antipathogenic metabolites, microbiomes from amphibian skin and other biofilms can influence infection dynamics by trapping zoospores and inactivating or consuming them through biofilm inhabitants (e.g. Sentenac et al. 2024). These findings open new avenues for future research on this topic. As with the interactions between amphibian biofilms and AZP host systems, current findings on the relationships between other aquatic communities co-existing with AZP host

systems highlight the complexity of these interactions and the need for further investigation of these biological drivers. We still know little about the effects of co-infection and hyper-parasitism, as well as the interactions between parasites, although it is well established that they can strongly modulate relationships with their hosts (e.g. Sparrow 1932; Canter 1949, 1969; De Jesús Andino et al. 2024; Thumsová et al. 2024; Yoldas et al. 2024). Consequently, targeted co-infection studies are needed in the future.

All biological factors are important, but given the increase in harmful algal blooms due to climate change (Wilk-Woźniak et al. 2024), understanding the allelopathic events and dependence of toxin production in hosts on their parasites will be useful for future management and conservation efforts related to aquatic habitats and their diversity. This protective mechanism experienced by hosts during parasite infection, as well as the release of volatile compounds or other organic metabolites, may increase their chances of fighting off AZP infections (Puneva and Christov 2004; Puneva 2006; Rohrlack et al. 2013; Yoneya et al. 2021; Agha et al. 2022), despite some conflicting results on this topic (Van Wichelen et al. 2010; Weisbrod et al. 2020).

Many questions remain unanswered, and further research is needed on all possible transport vectors of AZP, particularly concerning the increasing biological invasions. In this context, invasive host species, acting as transporters for the spread of their specific parasites, represent important AZP drivers. This is illustrated by the spread of emerging pathogens such as the noble crayfish plague (*Aphanomyces astaci*), the two causative agents of amphibian chytridiomycosis, *Batrachochytrium dendrobatidis* and *Batrachochytrium salamandrivorans*, as well as the fish-infecting *Sphaerothecum destruens* (Lowe et al. 2000; Martel et al. 2020; Gozlan and Combe 2023; Carvalho et al. 2024, etc.).

Our results emphasise the importance of complex physical, chemical and biological processes in the control of AZP and highlight the need for further research on this topic, as data on most factors remain limited and additional studies with broader scope are required. The following areas are poorly documented and lack sufficient evidence in the literature: (1) effects of oxygen and oxidative stress on AZP ecology, including the role of antioxidants such as glutathione; (2) macro- and micronutrient requirements of AZP; (3) physiological responses to osmotic stress and salinity tolerances of AZP; (4) effects of UVR on AZP; (5) role of disturbances in the diversity and abundance of AZP; (6) defensive role of autophagy during infection; (7) genomic data on AZP strains and their hosts, which are necessary for a better understanding of co-evolution and other biological drivers.

This review, based on an analysis of more than 500 papers, identifies significant research gaps and further open questions in various aspects of AZP biology and their role in the functioning of aquatic ecosystems: (1) the effect of

transient climatic events (such as heatwaves) on host–parasite interactions; (2) the effects of light pollution on AZP; (3) potential of RWTs as novel transporting vectors of AZP; (4) the ability of AZP to acclimatise or adapt, and the speed of this process; (5) the selective shedding of infected cells or organisms via grazing, which maintains population health; (6) the role of associated or symbiotic microorganisms in the algal mucus and other mucilage layers that help hosts resist AZP infections; (7) viral infection of AZP and its consequences for disease outbreaks. How expected changes in environmental factors, particularly those caused by climate warming, eutrophication and biological invasions, will affect AZP biodiversity and functionality, as well as AZP strategies—especially whether generalists will be more successful than specialists—remain important open questions.

Conclusions

There is already a substantial body of literature on the role of AZP and their functional diversity in natural and artificial aquatic habitats. Nevertheless, there is a significant need for a deeper understanding of the key factors promoting or constraining AZP diversity, distribution and infection success. The accumulated knowledge indicates that these are shaped by a complex interaction of environmental factors. Although the review raises further questions, it demonstrates the importance of identifying the abiotic and biotic drivers of AZP, especially in the context of climate change, eutrophication, water and light pollution, as well as biological invasions, to better predict environmental controls of disease dynamics. Despite the growing recognition of climate change as a dominant driver across various ecosystems, the specific impacts on AZP remain underexplored, and further research is needed to unravel the nuanced relationships between climate factors, parasite life cycles, host–parasite interactions and transmission patterns. This gap in understanding presents a significant challenge to managing and predicting disease outbreaks in aquatic environments, particularly in the context of global biodiversity conservation and aquaculture sustainability. Given the potential consequences for both aquatic biodiversity and human industries, it is essential to prioritise interdisciplinary research that integrates climate modelling, parasitology and ecological forecasting. Such research will enable more accurate predictions of disease risks and guide the development of adaptive management strategies. Ultimately, understanding the drivers of AZP dynamics in a changing climate will be crucial for mitigating future disease outbreaks, preserving aquatic ecosystems and ensuring the sustainability of aquatic resources. We hope that this review, which synthesises results on potential factors affecting AZP and outlines current knowledge gaps,

will provide useful information for the design of further interdisciplinary research on AZP ecology.

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Declarations

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