Impacts of human-induced pollution on wild fish welfare

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Abstract

The natural environment has been altered by anthropogenic actions for several centuries. For example, land clearing, water diversion and abstraction for agriculture have changed aquatic ecosystems, as have inputs from various diffuse and point-source pollution sources. Alteration of natural waterbodies leads to water quality and habitat changes that ultimately impact the welfare of resident fishes and may compromise their existence. In this chapter, we review different classes of pollutants and provide key examples of impacts observed in wild fish populations from freshwater and marine environments worldwide. This includes case studies on major pollution events and key pollution sources. Impacts ranging from direct toxicity and physiological perturbations through to behavioural changes and alterations in species compositions have all been documented and highlight the need for on-going management of anthropogenic inputs to aquatic environments.

Introduction

Anthropogenic disturbances are a major consideration for the welfare of wild fish. Disturbances include any external forces that alter ecosystem structure, such as toxic chemical pollutants that may cause direct and/or indirect mortality or habitat alteration that compromises living spaces and resource availability. Pulse disturbances are short-term perturbations (e.g. accidental chemical spills or flood events), while press disturbances are longer term perturbations that remain in an altered state after the initial disturbance (e.g. barriers caused by dams or elevated sediment contaminant concentrations). Ramp disturbance occurs when the disturbance increases over time with or without an upper boundary or asymptote (e.g. increasing sedimentation in a wetland) (Lake, 2000). The biotic response to each of these disturbance types can equally be categorised as pulse (short-term with return to baseline), press (longer term with new baseline/altered steady state) or ramp (increasing/decreasing response over time), and responses to the same disturbance will be different for different types of organisms. Fish inhabit diverse environments, and may be exposed to press, pulse and ramp disturbances throughout their lifetime. The ability to live and reproduce in such environments is based on differences in physiology and ecology which contribute to species' resilience and ability to recover from disturbance, such as that caused by pollution.

Some fishes are capable of living in extreme conditions and exhibit evolutionary adaptations to hostile environments. For example, annual killifish (*Austrofundulus limnaeus*) inhabit ephemeral tropical ponds in South America and produce embryos that are tolerant to UV radiation, salinity changes, anoxia and desiccation (Wagner et al., 2018). The embryos can depress their metabolism in a state of diapause, and accordingly have become an important vertebrate model for investigating the genetic mechanisms that enable these unique biological adaptations (Wagner et al., 2018). Arctic charr (*Salvelinus alpinus*) are the most northerly distributed freshwater fish, which inhabit polar regions with thermal extremes and up to 24 hr daylight during summer and 24 hr darkness in winter. This species can alter activity rhythms based on photoperiod, and forages in low light conditions at temperatures below 1^{o}C (Hawley et al., 2017).

The above examples illustrate the diversity in the physical and behavioural needs of fish, and the likely consequences of disturbances to welfare across fishes. Furthermore, what is considered acceptable in some circumstances (e.g. fish production for aquaculture) may not be acceptable in others (e.g. laboratory research). Subsequently, defining 'good fish welfare' is complex and necessarily species specific. A good general definition should incorporate good health, meeting behavioural (and social) needs and being free from pain or fear (Huntingford and Kadri, 2008).

The most effective way to determine fish health and overall welfare is through regular monitoring. While this is possible for captive fish, in wild fish it is much more difficult and requires both stock assessment (to determine estimates of abundance and size/age), and biomonitoring to establish health status. This makes it particularly difficult to link specific disturbances to effects in wild fish, which is reflected in the low number of published scientific studies (Henry, 2015; Hamilton et al., 2016).

In this chapter, we review the effects of human induced pollution in aquatic environments on the welfare of wild fish, first via summarising known effects for specific pollutant classes, and then through case studies that summarise known effects within an ecosystem through time.

Part 1: Types of Pollution

Point source pollution originates from a known location; often a deliberate site of waste removal, such as a sewage outfall. Conversely, non-point source or diffuse pollution may be an unintentional waste spill comprising diverse inputs that are difficult to identify and trace back to a source. Diffuse pollution is often associated with specific land uses, such as stormwater pollution with urban land use, or pesticide and nutrient pollution with agricultural use. Aquatic pollution, regardless of the source, is a major anthropogenic disturbance with negative impacts on fish welfare globally.

1. Nutrients

Human-induced increases in nutrient loadings have led to eutrophication of numerous areas around the globe, especially in developed countries. Eutrophication is now considered one of the greatest threats to surface waters worldwide (Xu et al., 2014). Eutrophication occurs when waterways experience excessive plant growth due to nitrogen (N) and phosphorus (P) inputs, derived from sources such as fertilisers, land clearing, animal production and discharge of human and animal wastes (Cloern, 2001). Direct impacts of eutrophication include selective mortality leading to changes in community structure (selected for nutrient tolerant species), while indirect effects include causing increased turbidity and low dissolved oxygen, leading to food web alterations and eventually changes in community structure (Smith and Schindler, 2009; Budria, 2017).

Reduced water clarity as a result of eutrophication can affect sexual selection in species that use visual-based selection cues, such as sticklebacks, sand gobies, pipefish and cichlids (reviewed by (Alexander et al., 2017). Eutrophication also influences host-parasite interactions, resulting in increases in opportunistic parasite infections in fish (Budria, 2017), while Warry et al. (2018) reported negative associations between demersal species richness in juvenile fish assemblages in estuaries with the degree of catchment fertilisation.

During the 20th century, nutrient transport to surface waters has increased dramatically, especially from agriculture. Global inputs are estimated to be 67 million ton/yr N and 9 million ton/yr P and contribute >50 % of total input rates (Beusen et al., 2016). This is predicted to be further exacerbated with future climate change, as N loadings into river catchments from agricultural soils increase during heavy rainfall events (Jeppesen et al., 2011).

Nitrogen, in the form of ammonia (NH_4), nitrate (NO_3) and nitrite (NO_2) are all very toxic to fish at high concentrations (Handy and Poxton, 1993), however such levels greatly exceed values that are normally observed in natural environments.

2. Hypoxia

A consequence of eutrophication is increased algal blooms and associated oxygen demand when the algae decomposes (Rabalais et al., 2010). Reduced dissolved oxygen concentrations in water lead to the development of hypoxia when levels reach <2.0 ml O_2/l (~35%S; 2.9 mg/l), and severe hypoxia when levels reach <0.5 ml O_2/l (Diaz and Rosenberg, 2008). Based on mean lethal concentrations for a range of fish and invertebrate species, Vacquer-Sunyer and Duarte (2008) suggested a threshold of 4.6 mg/l (~62.5 % S; 3.2 ml O_2/l) should be used to define hypoxia. Regardless of which threshold is used, it remains that severely hypoxic waters limit where species can live, and globally, more than 400 systems have been identified as adversely affected (Diaz and Rosenberg, 2008). The number of coastal sites affected by hypoxia is rapidly increasing at a rate of 5.5% per year, based on nearly 100 years of data (Vaquer-Sunyer and Duarte, 2008). Moreover, in temperate regions there are usually seasonal patterns of hypoxia (Gobler and Baumann, 2016).

There are large variations in oxygen thresholds between species and a range of behavioural and physiological responses occur in response to hypoxia (Wu, 2002; Vaquer-Sunyer and Duarte, 2008; Rabalais et al., 2010). For example, some benthic species actively avoid low dissolved oxygen areas, moving into shallower, more oxygenated waters to avoid hypoxia. Physiological responses in fish range from increased ventilation rates and oxygen binding capacity of haemoglobin, to downregulation of several metabolic pathways to conserve energy, before eventually shifting to anaerobic respiration (Wu, 2002). There are several welfare concerns associated with these responses, including increased risk of predation (by moving into shallower water), reduced growth and fitness, as well as the potential for transgenerational and epigenetic effects on offspring (Wu, 2002; Vaquer-Sunyer and Duarte, 2008; Wang et al., 2016).

3. Ocean acidification

Predictions have been made that oceanic carbon dioxide (CO_2) concentration oscillations may increase up to 10-fold by 2100 if atmospheric CO_2 emissions continue to rise (Meinshausen et al., 2011; McNeil and Sasse, 2016). This would result in CO_2 levels in surface waters increasing from current estimates of around 390 µatm to values >1000 µatm (Melzner et al., 2013; McNeil and Sasse, 2016). Additionally, the same processes responsible for driving hypoxia in marine systems are also implicated in ocean acidification, and are likely to exacerbate effects since low dissolved oxygen and associated low pH contribute to the formation of carbonic acid, and resultant acidification of waters (Gobler and Baumann, 2016).

Acidification has four main effects on fish: 1) altered otolith formation at high levels of atmospheric CO_2 (Checkley et al. 2009) ; 2) altered extra- and intracellular acid-base status, affecting physiological processes (Melzner et al., 2009); 3) altered acid-base status, which affects energy budgets (Melzner et al., 2013); and 4) altered behaviours (e.g. preferences for salinity and temperature; Pistevos et al., 2017). Fish can compensate by adjusting their extra-cellular acid-base equilibrium, which can lead to hyper-calcification and behavioural changes (Melzner et al., 2009; McNeil and Sasse, 2016).

Increased CO_2 levels in water may interfere with oxygen transport and uptake (Hannan and Rummer, 2018) as well as impairing physiological functions and causing the diversion of energy away from important fitness needs (i.e. locomotion, reproduction and avoidance of predation and environmental stressors) (Brewer and Peltzer, 2009). The combination of low DO and low pH causes variable but generally adverse effects on growth and survival of the early life stages of estuarine forage fishes (*Menidia beryllina*, *M. menidia*, *Cyprinodon variegatus*). Both additive and synergistic negative effects on fitness and survival may occur (DePasquale et al., 2015). In orange clownfish (*Amphiprion perula*), increased pCO_2 in seawater affected the ability of larvae to detect adult cues, and the larvae were attracted to cues that they would normally avoid (Munday et al., 2009). Acidification caused impaired sensory ability in fish larvae, which negatively affected their ability to locate suitable settlement sites. In adult fish, chronic exposure to elevated CO_2 affected reproductive output in two Australian reef fishes (Welch and Munday, 2016): in *Amphiprion perula*, an increase in the number of clutches and number of eggs per clutch were observed, while in *Acanthochromis polyacanthus*, the opposite occurred, with a decrease in the number and size of the egg clutches produced.

Clearly ocean acidification has immense potential to negatively impact welfare in wild fish populations, and given that responses are species-specific, accurate predictions of likely impacts are difficult. Furthermore, even with improved management of all anthropogenic inputs that influence eutrophication, hypoxia and acidification, future climate change will intensify the amplitude of these stressors and therefore the potential for widespread impacts on fish welfare.

4. Heavy metals

Metal pollution is a global problem resulting from mining and industry, as well as natural geological processes. Metals enter aquatic systems predominantly through point sources such as industrial effluents, as well as diffuse sources like stormwater and urban runoff. Some metals (and metalloids) are essential, in that organisms require trace concentrations of them for normal physiological function (e.g. iron in haemoglobin to facilitate oxygen binding), while presence of non-essential metals is often associated with stress and detoxification responses. The toxicity of metals depends on bioavailability, which is in turn influenced by the organic and inorganic complexes formed by the metals in the natural environment (Wang and Rainbow, 2008). The key metals that negatively affect fish welfare are lead, mercury, cadmium, chromium, nickel, copper, zinc, tin, aluminium and arsenic.

In the widespread freshwater fish, *Galaxias maculatus*, metal pollution causes a range of impacts, including increased oxidative stress and ionoregulatory disturbances (McRae et al., 2018), alterations in behavioural responses to alarm cues (Thomas et al., 2016) and delayed embryo development and poor quality larvae with reduced phototactic responses (Barbee et al., 2014). Physiological responses to metal pollution may also be increased by additional stressors such as salinity or handling stress (Harley and Glover, 2014; Glover et al., 2016). In fish early life stages, metal toxicity can delay growth and cause developmental retardation, reduced survival and increased rates of deformities. Skeletal deformities are of particular concern, since they affect essential characteristics such as the ability to swim, with subsequent effects on other behaviours such as predator avoidance and migration (Kennedy, 2011; Sfakianakis et al., 2015).

Metals can accumulate in specific fish tissues, which over time can affect metabolism and cause oxidative stress, lead to cellular alterations, chromosomal damage, and potentially cause multi-generational effects (Giguere et al., 2005; Mieiro et al., 2011; Mohmood et al., 2012; Pereira et al., 2016; Defo et al., 2018), Metals may also interfere with physiological processes that can affect dominance behaviours and social interactions in fishes (Sloman, 2007).

To assess food web interactions and their influence on metal bioaccumulation, 26 different fish species were sampled from the upper Yangtze River in China and the highest concentrations of metals (As, Cr, Cd, Hg, Cu, Zn, Pb, Fe) were measured in large predatory fish and benthic feeding species (Yi et al., 2017). Similarly, in perch (*Perca fluviatilis*) and roach (*Rutilis rutilis*) sampled from a Polish lake, some species-specific differences in tissue accumulation were observed and the concentration of Cu in perch livers was positively correlated with the hepatosomatic index (HSI), whilst the concentration of Hg in perch gonads was negatively correlated with the gonadosomatic index (GSI) (Luczynska et al., 2018).

In sand flathead (*Platycephalus bassensis*) sampled from a metal contaminated estuary in Australia, Fu et al. (2017) reported significant up-regulation of genes associated with metal homeostasis, detoxification and oxidative stress, relative to fish sampled from a reference site. They also noted a higher prevalence of certain gill histopathologies in the fish from polluted sites, yet, the prevalence of gill parasites was lower in fish from polluted sites than the reference site (Fu et al., 2017).

Chronic exposure to metals may lead to adaptation in local populations, resulting in higher metal tolerances and altered physiological responses and bioaccumulation kinetics (Durrant et al., 2011; Hamilton et al., 2016; Abril et al., 2018). For example, populations of brown trout (*Salmo trutta*) from a British river contaminated by metals from historic mining activities were found to be genetically distinct (forming separate populations) across a very short distance (1 kilometre) due to a 'chemical barrier' formed by the heavily polluted section of the river (Durrant, et al., 2011).

5. Persistent organic pollutants (POPs)

Persistent organic pollutants (POPs) are derived from a range of industrial and agricultural processes and are a globally important pollution class. The most well studied and environmentally important POPs include dioxins and furans, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), polybrominated diphenyl ethers (PBDEs) and some per- and poly-fluoroalkyl substances (PFASs). POPs persist in the environment, often bioaccumulating in exposed organisms, and can function as reproductive toxicants. Many POPs have been banned internationally under the Stockholm Convention, and several other chemicals have tight restrictions on use or phase-out strategies (UNEP, 2001). Toxic responses to POPs are mostly mediated through the aryl hydrocarbon receptor (AhR) pathway (Zhou et al., 2010) and there are several well documented cases of POPs affecting fish welfare (Cook et al., 2003; Letcher et al., 2010; Henry, 2015; Akortia et al., 2016).

The most potent dioxin, 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) is extremely toxic to the early life stages of fish and causes a distinctive and predictable spectrum of toxic responses, including yolk sac and pericardial oedema, haemorrhaging and vascular damage, craniofacial malformations and hyperpigmentation (Zabel et al., 1995; Walker et al., 1996; Cook et al., 2003). Other AhR agonist chemicals can cause similar effects in fish larvae, and high loadings of POPs entering Lake Ontario during the 20th century were associated with the collapse of a Lake trout (*Salvelinus namaycush*) population (Cook et al., 2003).

Some POPs are also endocrine disruptors. For example, Baldigo et al.(2006) reported that ratios of sex steroids (17β-estradiol and 11-ketotestosterone) and vitellogenin protein were correlated with lipid-based PCB residues in tissue of male fish of four species (carp - *Cyprinus carpio*; bass - *Micropterus salmoides* and *Micropterus dolomieui*; and bullhead - *Ameiurus nebulosus*) from the Hudson River, New York.

6. Endocrine disrupting chemicals (EDCs)

Environmental pollutants that can interfere with the normal functioning of the endocrine system are known as endocrine disrupting chemicals (EDCs). This grouping includes many different types of chemicals, such as pharmaceuticals, some metals, POPs, pesticides and plasticisers. These chemicals are capable of binding or blocking hormone receptors, which leads to upregulation or downregulation of hormone production and cascading effects within specific endocrine pathways (Colborn et al., 1993). The most well studied systems in fish are the hypothalamic-pituitary-gonadal (HPG) axis, which regulates all aspects of reproduction, and those involved with glucose regulation and thyroid metabolism pathways (Trudeau and Tyler, 2007). Fish exposed to EDCs may exhibit behavioural changes that affect sexual selection and reproductive outcomes (copulation success), as well as physiological and morphological changes.

Exposure to EDCs can have transgenerational and epigenetic impacts, leading to poor reproductive outcomes, reduced survival of offspring and lower fecundity in subsequent generations (Guerrero-Bosagna et al., 2007). Several anthropogenic pollutants are classified as EDCs, including natural and synthetic estrogens from sewage discharges, as well as pesticides, pharmaceuticals and personal care products from point source and diffuse sources (Tijani et al., 2016). A review of chemicals known to elicit endocrine-mediated adverse effects on wildlife concluded that legacy compounds such as tri-organic tins and POPs caused greater impacts than most current use compounds, with the exception of 17α-ethinylestradiol (EE2) and other sewage-associated estrogenic compounds that have been widely studied and demonstrated to cause adverse effects on fish populations (Matthiessen et al., 2018). A whole-lake experiment conducted in the Canadian Experimental Lakes Area demonstrated adverse impacts in fish chronically exposed to the potent synthetic estrogen, 17α-ethinylestradiol (EE2) (Kidd et al., 2007; Palace et al., 2009): fathead minnows (*Pimephales promelas*) displayed endocrine disruption impacts including feminisation of male gonads, increased production of the egg yolk precursor vitellogenin, and altered oogenesis in females. Over multiple years, decreased catch rates were observed, and reproductive failure led to near complete collapse of the minnow population within two years of the initial exposure (Kidd et al., 2007). However, other fish species in the lake were not affected as severely (Palace et al., 2009), indicating that life history characteristics (i.e. length of life cycle) are important considerations for predicting the impacts of endocrine disruptors on wild fish populations. Subsequent studies on the same populations have indicated that recovery (of fish abundance and distribution) occurred following cessation of EE2 exposure (Blanchfield et al., 2015).

In the United Kingdom, several years of research has also demonstrated that environmental estrogens cause reproductive alterations in fish (Tyler and Routledge, 1998; Sumpter and Jobling, 2013). The development of intersex gonads, where male testis develop oocytes, has been linked with decreased fertility (Harris et al., 2011), and in severe cases can result in complete sex reversal and feminisation. However, despite a large body of scientific literature from both lab and field studies, establishing population-level impacts in English rivers due to endocrine disruption has been difficult, and is still not completely resolved (Sumpter and Jobling, 2013).

7. Pesticides

Pesticides are an essential pest management tool for intensive agriculture globally. Several different classes of pesticides, with different modes of action (and therefore toxicity) have been developed to target specific organisms, such as insecticides for invertebrate pests and herbicides for weed species. Effects on fish welfare (in laboratory settings) are well documented, especially for older pesticides, many of which are now banned due to their known persistent, bioaccumulative and toxic properties (Jorgenson, 2001). General effects of pesticides on fish are direct toxicity, sublethal stress responses (i.e. upregulation of detoxification enzymes and protective proteins) and reproductive toxicity. Furthermore, several pesticides bioaccumulate (Lazartigues et al., 2013) and some are also classified as EDCs due to demonstrated interference with hormonal systems (McKinlay et al., 2008; Brander et al., 2016).

The synthetic pyrethroids are a class of insecticide used widely in agricultural, veterinarian and domestic/household uses. Pyrethroids bioaccumulate in wild fish (Corcellas et al., 2015) and following a chemical spill from an industrial area in North Eastern Italy, a large fish kill involving multiple species was attributed to pyrethroid pollution (Bille et al., 2017). In fishes sampled from a river in Eastern Spain, Belenguer et al. (2014) observed a significant relationship between tissue concentrations of the organophosphate insecticide, diazinon, and Fulton's condition factor, suggesting growth may be affected by pesticide exposure. Whilst in Brazil, common carp *(Cyprinus carpio*) reared in an irrigated rice-farming system that uses multiple pesticides, were shown to bioaccumulate certain synthetic pyrethroids and fungicides, display altered enzymatic activity in brain, liver, gills and muscle, as well as increased lipid peroxidation and protein oxidation (Clasen et al., 2018).

8. Emerging pollutants and problems

As populations grow and new technologies are developed globally, emerging pollutants continue to increase in the environment. These emerging pollutants are likely to bring welfare concerns for wild fish, but in many cases there is insufficient data on their effects.

Plastic pollution is a direct product of anthropogenic development. It is a manmade product that has been widely used for all kinds of applications since the 1940s. Since plastic does not degrade, once it enters the environment it remains, often breaking into smaller fragments (Li et al., 2016). Marine plastic pollution has been linked with welfare issues in fish, most notably mortality due to starvation and obstruction of the digestive system by ingested plastic, as well as exposure to contaminants sorbed to ingested plastic particles (Gall and Thompson, 2015; Rummel et al., 2016; Wardrop et al., 2016). In a comparative study of anthropogenic debris in fish sold for human consumption, Rochman et al. (2015) reported that 55% of all species sampled from Indonesian fish markets, and 67% of all species sampled from US fish markets, contained debris (plastic or fibres) in their digestive tracts, which included fish from various trophic levels and habitat types, including small foraging species through to large predators.

Per- and poly-fluoroalkyl substances (PFASs) are another emerging group of pollutants that have impacts on fish welfare. PFASs are widely used in industrial applications from firefighting foams and stain resistant fabrics, through to non-stick frypans and water repellent clothing. They persist in the environment and bioaccumulate. Two widely used PFASs, perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) are harmful pollutants, with phase-outs and future bans in place under the Stockholm Convention. Detectable concentrations of PFOA and PFOS in blood samples from wild eels (*Anguilla anguilla*) from two Italian waterways were associated with liver macrophage aggregates and lipid vacuolation (Giari et al., 2015), while in carp (*Cyprinus carpio*) and eels (*A.*

anguilla) sampled from multiple Belgian rivers, hepatic PFOS concentrations were significantly correlated with changes in serum alanine aminotransferase activity, protein content and electrolyte levels (Hoff et al., 2005).

Each year several new chemicals are registered for use in industrial and agricultural processes, and often the properties that make them useful (i.e. long-lasting and hard-wearing) contribute to their persistence once they enter aquatic ecosystems. Therefore, prevention or minimisation of any chemicals entering waterways is the most effective way of ensuring there will be no welfare impacts on wild fish.

Part 2: Case studies – pollutants in the environment and their impacts on wild fish

1. Aquaculture pollution and its effects on wild fish populations

Aquaculture, especially sea cage fish farming, introduces a range of environmental pollutants that affect wild fish populations. Foremost, high stocking densities produce large quantities of waste in the form of faeces and spilled feed. This localised nutrient input can lead to altered benthic communities and low dissolved oxygen conditions where water exchange is insufficient and nutrient thresholds exceed those that receiving environments can biologically assimilate (Wu et al., 1994). However, this waste also provides an attractive trophic subsidy (Dempster et al., 2002; Dempster et al., 2009; Dempster et al., 2011; Sanchez-Jerez et al., 2011). In both tropical and temperate systems, fish farms thus act as ‘hot spots’ of wild fish aggregation, with substantial increase in abundance and diversity in the near vicinity of farms. By feeding preferentially at farms, wild fish ameliorate and disperse nutrient loading (Vita et al., 2004), but also undergo dietary changes, including a shift from marine-derived long chain polyunsaturated fatty acids to terrestrial short chain fatty acids (Fernandez-Jover et al., 2011; Arechavala-Lopez et al., 2015). Likely effects of this dietary shift are poorly understood in wild populations (Salze et al., 2005; Bogevik et al., 2012). In addition to nutritional changes, fish that are attracted to farms by waste feed face increased risk of disease transmission from farmed to wild fish (Zlotkin et al., 1998; Diamant et al., 2000; Colorni et al., 2002; Glover et al., 2013), and are vulnerable to increased fishing pressure and predation (Bagdonas et al., 2012; Callier et al., 2017).

Contaminants from feed, antibiotics, parasiticides and anti-fouling are also possible near aquaculture sites (Burridge et al., 2010; Taranger et al., 2015). Elevated levels of mercury (2.0-2.1x: DeBruyn et al., 2006; Bustnes et al., 2011) and organohalogens (Bustnes et al., 2010) have been reported in the tissues of farm-associated wild fish, although it is unclear whether this effect is driven by contaminated feed or biomagnification due to the elevated trophic level of farm-associated fish assemblages. The effects of these levels on wild fish remain untested.

Where antimicrobials are used at fish farms, residues can appear in the tissues of wild fish (oxytetracycline: Björklund et al., 1990; oxolinic acid: Samuelsen et al., 1992; flumequine: Ervik et al., 1994), leading to selection for resistant genes that over time can increase the rate of antimicrobial resistance. Selective microbial growth can also alter biodiversity of normal skin and gut flora, which may compromise fish immunity and reduce resilience (Cabello et al., 2013). Antimicrobial resistance is a major threat to the aquaculture industry since disease outbreaks can rapidly spread in intensive holding conditions, and without effective treatment options, infection can cause widespread mortality that can easily decimate populations (Cabello et al., 2013; Watts et al., 2017).The development of vaccines has allowed fish farmers in some countries (e.g. Norway, Scotland) to largely cease antimicrobial use, but use remains high elsewhere, such as Chile (Watts et al., 2017). The aquaculture industry is developing a range of new techniques to minimise effects of pollution, including new feed delivery technologies that reduce spillage, moving farms offshore where waste disperses better, protecting farm-associated fish from fishing, and disease control measures. Successful improvements along these fronts will minimise negative welfare impacts on wild fish.

2. Acid rain

Freshwater fish populations in many regions are vulnerable to acidic deposition or 'acid rain'. Industrial emissions of sulfur dioxide and nitrogen oxide react with atmospheric moisture to form sulfuric and nitric acid, resulting in atmospheric moisture with a pH range of 3.5-5.0. Acid is then deposited by precipitation (wet deposition) or contact between the atmosphere and the Earth's surface (dry deposition), before accumulating in aquatic environments via surface runoff. Long term exposure can erode the buffering capacity of water bodies, with pH in highly vulnerable lakes falling below 5.0.

Acid rain caused fish kills in thousands of Scandinavian lakes and rivers from the 1950s until the 1990s (Hesthagen et al., 1999; Leivestad and Muniz, 1976), and large-scale fish population declines were also documented throughout western Europe and North America during that period (Menz and Seip, 2004). Declines in fish populations may initially be an indirect result of acidification, as invertebrates are often the first affected by changes to pH and resulting declines in prey abundance can lead to starvation for predatory fishes before direct effects occur (Schindler, 1988). Early fish life stages are also more vulnerable than adults, causing recruitment failure (Alabaster and Lloyd, 1982). For example, pH <5.1-5.9 was sufficient to reduce egg and larval survival in several Canadian freshwater fishes (Holtze and Hutchinson, 1989). Fatal pH levels for adult fish vary widely across species and are strongly dependent on interacting factors such as adaptation, exposure duration, water hardness, free carbon dioxide concentration and the presence of other pollutants such as aluminium, which is mobilised by low pH (Alabaster and Lloyd, 1982; Schindler, 1988). Adults in acid-adapted populations may survive at least temporary exposure to pH 3.7, but more often pH <5.0 is lethal (Alabaster and Lloyd, 1982). The most productive fish populations are found at pH >6.3, with lower productivity in mildly acidic waterways perhaps reflecting low food availability and sublethal effects such as physiological stress, sensory disruption and behavioural changes that are likely to reduce reproductive output. For example, reproductive behaviours in salmonids are suppressed at pH <6.4 (Ikuta et al., 2003).

Fish populations can recover once acid deposition is reduced, but there may be a lag of several years or even decades depending on the rate of return to natural pH and recovery of lower trophic levels, especially where recolonization or restocking is necessary after local extinction (Hesthagen et al., 2011; Menz and Seip, 2004; Mills et al., 2000).

3. Oil spill impacts on wild fish

Marine oil spills have become an inevitable consequence of fossil fuel extraction and refinement. The various pollutants that enter the environment as a result cause a range of toxic effects on wild fish populations. Oils are complex mixtures of components including linear hydrocarbons, polycyclic aromatic hydrocarbons (PAHs), pentacyclic hopanes, and benzene, toluene, ethylbenzene and xylene (BTEX). BTEX tend to degrade quickly in seawater, whereas PAHs are most persistent and can accumulate in sediments (Murawski et al., 2016).

In 2010, the BP Deepwater Horizon oil rig was damaged due to fire, resulting in the release of > 3 million barrels of crude oil into the Gulf of Mexico (Beyer et al., 2016). The estimated extent of the contamination of coastal and continental shelf areas with oils was 144 192 km^2 and 88% of these areas had concentrations of PAHs greater than those known to cause toxic impacts in marine life (Murawski et al., 2016). Hydrocarbon concentrations in the water column were 160-fold higher than levels measured prior to 2010 and reached mean (± 95% CI) levels of 104 ± 17 ppb (total hydrocarbons) and 43 ± 17 ppb (PAHs) near the surface (Murawski et al., 2016).

Measuring PAH metabolites in fish bile is a sensitive indicator of exposure to hydrocarbons, and while very high concentrations of naphthalene equivalents were measured in some fish from the Gulf of Mexico in 2011 (470 000 ng/ g bile), levels have been gradually decreasing since then (Beyer et al., 2016). Similarly, reductions in the activity of several hepatic biomarkers known to be upregulated by exposure to PAHs (ethoxyresorufin-O-deethylase (EROD), glutathione transferase (GST) and glutathione peroxidase (GPx)) were observed in red snapper (*Lutjanus campechanus*) and gray triggerfish (*Balistes capriscus*) over a three year period following the Deepwater Horizon spill (Smeltz et al., 2017). Contrary to expectation, fish abundances in the Gulf of Mexico increased after the event due to fisheries closures and reduced predation (due to mortality in seabirds and other large predators). Therefore, it has been difficult to accurately gauge the impacts of the oil spill on fish assemblages, but changes in species compositions and patterns of fish recruitment in the Gulf of Mexico are beginning to emerge (Schaefer et al., 2016). For example, exceptionally high recruitment has been observed in Gulf menhaden (*Brevoortia patronus)* for multiple years following the oil spill due to a loss of predators (Short et al., 2017).

Developmental and subsequent physiological impacts are also beginning to emerge, such as malformations of the hearts of large predatory species, with associated reductions in fitness and swimming capacity (Incardona et al., 2014). Such changes are likely to have profound effects on shaping the fish communities and overall ecosystem integrity of the Gulf of Mexico in the future.

4. Radionuclide impacts on wild fish

Energy production using nuclear technologies is widely utilised, and accidental releases of radioactive materials into marine ecosystems can have serious impacts on wild fish populations. In 2011, an earthquake in Japan triggered a large tsunami that flooded the Fukushima Daiichi nuclear power plant, leading to a catastrophic explosion and subsequent release of large quantities of radioactive cesium (^{134}Cs, ^{137}Cs) and other isotopes into the Pacific Ocean. The two major sources of radionuclides to the environment following the Fukushima incident were atmospheric fallout and discharge of contaminated seawater from the power plant. Groundwater and river runoff contribute additional and ongoing sources of contamination (Buesseler et al., 2017).

Radiocesium isotopes have long half-lives (^{134}Cs- 2.06 years; ^{137}Cs – 30.2 years) and were detected widely in surface seawater and marine biota immediately following the accident (Wada et al., 2016; Buesseler et al., 2017). Other radionuclides that were released include ^{90}Sr, ^{239,240}Pu and ^{129}I, all of which carry health concerns due to their long (>1 year) half-lives. Radionuclides can damage cells and chromosomal DNA, causing a range of adverse welfare effects. Reduced fitness, due to changes in blood composition and immunosuppression, as well as reduced reproductive output due to gonad abnormalities, reduced fertility and increases in mortality and abnormalities in fish early life stages have all been observed following chronic radiation exposure (Sazykina and Kryshev, 2003; Kong et al., 2016; Hurem et al., 2018). Radioactive pollution is measured in Becquerel (Bq) units, which represent the quantity of radioactive material per unit time. Large quantities are reported as penta Becquerels (PBq, 10^{15} Bq). Estimates of the total nuclear fallout from the Fukushima accident are in the range of 8.8-50 PBq (Buesseler et al., 2017).

Cesium uptake into fish occurs through water as well as food ingestion and Cs has moderate bioconcentration (uptake from water) and biomagnification (increase with trophic level) factors (Madigan et al., 2017). In 2011, about half of all fish sampled in coastal areas surrounding Fukushima exceeded the Japanese regulatory limit for Cs of 100 Bq/kg, and the levels tended to be higher in demersal than pelagic fishes. Within 4 years, less than 1% of samples exceeded the regulatory limits (Buesseler et al., 2017). In heavily contaminated areas within Fukushima harbour, some fishes still exceed guideline values, and as such, netting barriers have been installed to prevent these fish leaving the contaminated harbour. Reductions in fish Cs levels are occurring more slowly in demersal than pelagic species, due to ongoing exposure through feeding on contaminated benthic infauna. Continued monitoring of wild fish populations in the Fukushima region are needed, because whilst to date there has been no evidence of adverse effects, chronic radiation exposure is known to reduce fitness and reproductive output and therefore could affect fish welfare in the future.

Conclusion

Pollution and other anthropogenic disturbances affect all aspects of animal welfare, from direct toxicity due to chemicals such as heavy metals and hydrocarbons, through to avoidance behaviours and species alterations due to eutrophication and associated hypoxia and ocean acidification. Quantifying stress and welfare is especially difficult in wild fish and requires a versatile range of endpoints that can be measured from individual fish through to entire fish communities.

To ensure fish populations are sustainable, and that adequate welfare requirements are achieved, consideration needs to be given not only to existing pollution levels and suitable abatement/reduction methods, but also of the probable effects that future climate change with have on fish, in particular the altered dynamics of processes such as the nitrogen cycle and acid chemistry in both freshwaters and oceanic ecosystems.

Animal ethics legislation already assists in managing captive fish and direct human interactions with fish (i.e. fishing regulations), but establishing and maintaining appropriate welfare standards in wild fish requires close, adaptive management of not only physical disturbances (i.e. habitat loss), but of chemical pollutants and wastes from agriculture and aquaculture. These are important issues that require a global response to protect and preserve our diverse global fish populations.

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