



Harbours as unique environmental sites of multiple anthropogenic stressors on fish hormonal systems

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ABSTRACT

Fish development and acclimation to environmental conditions are strongly mediated by the hormonal endocrine system. In environments contaminated by anthropogenic stressors, hormonal pathway alterations can be detrimental for growth, survival, fitness, and at a larger scale for population maintenance. In the context of increasingly contaminated marine environments worldwide, numerous laboratory studies have confirmed the effect of one or a combination of pollutants on fish hormonal systems. However, this has not been confirmed *in situ*. In this review, we explore the body of knowledge related to the influence of anthropogenic stressors disrupting fish endocrine systems, recent advances (focusing on thyroid hormones and stress hormones such as cortisol), and potential research perspectives. Through this review, we highlight how harbours can be used as "*in situ* laboratories" given the variety of anthropogenic stressors (such as plastic, chemical, sound, light pollution, and invasive species) that can be simultaneously investigated in harbours over long periods of time.

1. Introduction

Coastlines – the interface between the land and the sea – have been increasingly subjected to anthropogenic pressures such as fishing, boating, pollution, coastal construction, and tourism since the end of the 19th century (e.g., Crain et al., 2009; Dugan et al., 2011). Marinas, harbours, and ports are among the most common man-made structures along coastlines worldwide (e.g., Hardaway and Duhring, 2010; Dafforn et al., 2015). Harbours are often hotspots of human-mediated environmental pressures which significantly affect the quality of surrounding coastal and marine waters, as well as the native vs. non-native species assemblages (Mayer-Pinto et al., 2015). For example, Harik et al. (2017) assessed anthropogenic stressors along the Mediterranean coastline

using a multi-criteria index. The study showed that harbours along the Mediterranean coastline are the third largest cause of stress to marine ecosystems, behind landfills and industrial sites (Harik et al., 2017). The construction and presence of harbours not only directly destroy natural coastal habitats, but also indirectly damage neighbouring ecosystems through changes in currents and sediment loads which dramatically impact the composition of benthic communities (Meinesz et al., 1991; Martin et al., 2005; Li et al., 2010), as well as fish larval dispersal and recruitment (Roberts, 1997). Within harbours, direct stressors include pollutants from maritime activities (e.g., diesel spills during vessel refuelling, hydrocarbon pollution from boat traffic - Steen et al., 2004 in Danish harbours; Bergen et al., 2005 in the USA; Ingole et al., 2009 in India; Li et al., 2010 in China; Mayer-Pinto et al., 2015 in Australia;

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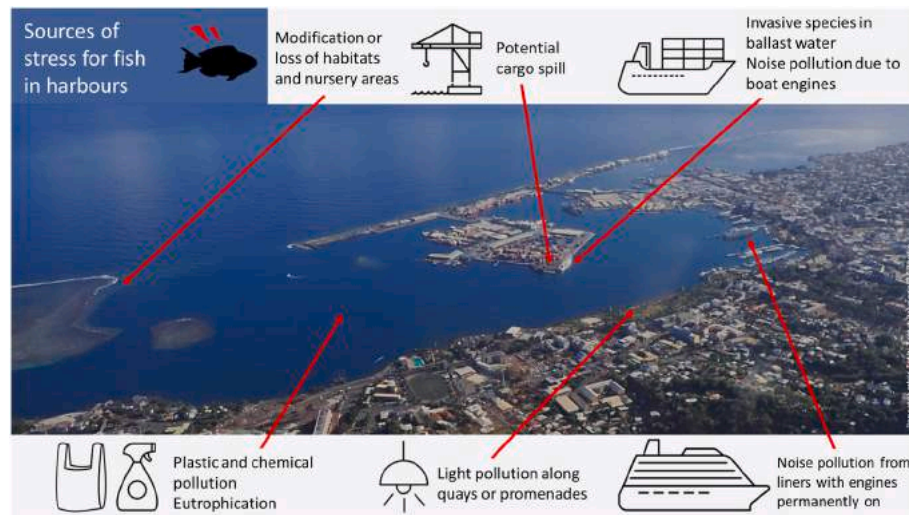


Fig. 1. Illustration of the different anthropogenic stressors present in the port of Papeete, Tahiti, French Polynesia (illustration by C. Berthe).

Harik et al., 2017 in Mediterranean harbours), and pollution generated by neighbouring cities and agricultural activities. These can lead to high levels of heavy metals and organic pollutants (e.g., high polychlorinated biphenyl levels such as in New Bedford Harbour - Bergen et al., 2005; Nogales et al., 2011; Mali et al., 2017), affecting the composition and function of microbial (Nogales et al., 2011) and macro-faunal communities (Martínez-Lladó et al., 2007) in the water and sediments in harbours and marinas. Overall, although the legislative framework concerning marine pollution control and prevention has been increasingly enforced in harbours since the early 2000s (AAPA, 2003; ESPO/EcoPorts, 2009; US-EPA, 2009), harbours remain man-made structures in which intense human usage impacts coastline environments. Nonetheless, harbours and marinas contain marine communities and therefore constitute sites of interest for ecologists and endocrinologists working on fish populations.

In their review of 200 environmental studies on Sydney harbour, Mayer-Pinto et al. (2015) found that 109 focused on contamination, 58 on habitat modification, 11 on invasions by non-indigenous species, and only 8 on fish and fisheries. This relative gap in studies on fisheries from Sydney harbour mirrors the lack in knowledge of the impact of harbours on fish worldwide. Elsewhere, although many studies have been conducted on intertidal sessile organisms living in harbours (e.g., Yung et al., 1999; MacFarlane and Burchett 2003), only a few have monitored mobile species such as fish (e.g., Wooldridge et al., 1999; Mercader et al., 2017a; Mercader, 2018). The low number of studies looking at fish species may be due to practical aspects such as the danger of boat traffic when swimming, as well as fishing restrictions in harbours (Ferreira et al., 2020). In addition, harbours are seldom considered of interest for fish studies as they are rarely viewed as key habitats in terms of ecological function along coastlines and islands and are often only referred to as polluted and artificial habitats (Mayer-Pinto et al., 2015). These multiple reasons may explain the paucity of scientific papers focusing on the ecology and physiology of fish living in harbours. However, harbours could be considered as unique habitats of key scientific interest. Indeed, they concentrate uncommonly high and varied levels of pollutants. These urbanized centres, previously absent from coastlines, completely differ from the ecosystems they replaced as well as from the surrounding natural areas in terms of abiotic composition (e.g., substrate type, water quality, hydrodynamics). They represent a novel ecosystem (Hobbs et al., 2013) with their own functional groups of living organisms. These make them unique areas in which to study the synergistic effects of multiple anthropogenic stressors on the ecology, biology, and physiology of fish. In addition, from an adaptive and evolutionary point of view, they provide a setting for long-term field

experiments impossible in mesocosm conditions (e.g., Viard et al., 2020; Le Moan et al., 2021). The aim of our review is to encourage future research on the biology (with a focus on endocrinology) and ecology of fish affected by multiple stressors encountered in harbours. To do so, we (i) identify the different stressors present in harbours and their potential impacts on marine fauna, (ii) present baseline knowledge on the effect of multiple anthropogenic stressors on two endocrine pathways in marine fish (stress and thyroid hormones), and (iii) propose a framework to conduct integrated research that could provide crucial insights for ecosystem management, as well as a better understanding of hormonal responses in fish under stressful conditions.

2. Harbours as fish habitats and potential nurseries

Harbours are semi-closed coastal marine areas with limited water mixing with the open sea, in which marine organisms are exposed to multiple stressors (e.g., chemical contamination, elevated nutrients, turbidity, marine debris, microplastics, invasive species, habitat modification, boat traffic and noise and artificial light at night - Li et al., 2010; Mayer-Pinto et al., 2015, Fig. 1). In this review, we focus on six major anthropogenic stressors that can affect the hormonal system of fish: plastic pollution, chemical contamination, invasive species, boat noise, artificial light at night, temperature variation, as well as their combined effects.

Despite the numerous anthropogenic stressors present in harbours, studies have found that harbours can be marine biodiversity hotspots, but these have mostly focused on macro-invertebrates (Johnston et al., 2015a,b; Mayer-Pinto et al., 2015). The existing quantitative data on fish populations does not paint a similar picture for fish biodiversity (Edgar and Shaw, 1995; Clynick, 2006; Hutchings et al., 2013). However, recent studies have highlighted that some harbours with artificial habitats (mainly in temperate regions) could be nursery areas for fish (Bouchoucha et al., 2016; Dufour et al., 2009; Mercader et al., 2017b; Patranello et al., 2017; Mercader, 2018). Breakwaters host high densities of juvenile fish (Ruitton et al., 2000; Pizzolon et al., 2008; Dufour et al., 2009) and in some areas (e.g., in Sydney harbour; Clynick 2008) adult fish species richness and abundance inside marinas are close to those found on natural rocky habitats. Mercader (2018) surveyed juvenile fish populations on various artificial infrastructures and natural sites along a 100-km section of the French Mediterranean shoreline. Juvenile fish densities were highly variable between harbours depending on the type of small-scale habitats, with juvenile densities on ripraps or jetties higher or equivalent to those on natural sites. Port and marina jetties might in some cases provide suitable nursery grounds for juvenile

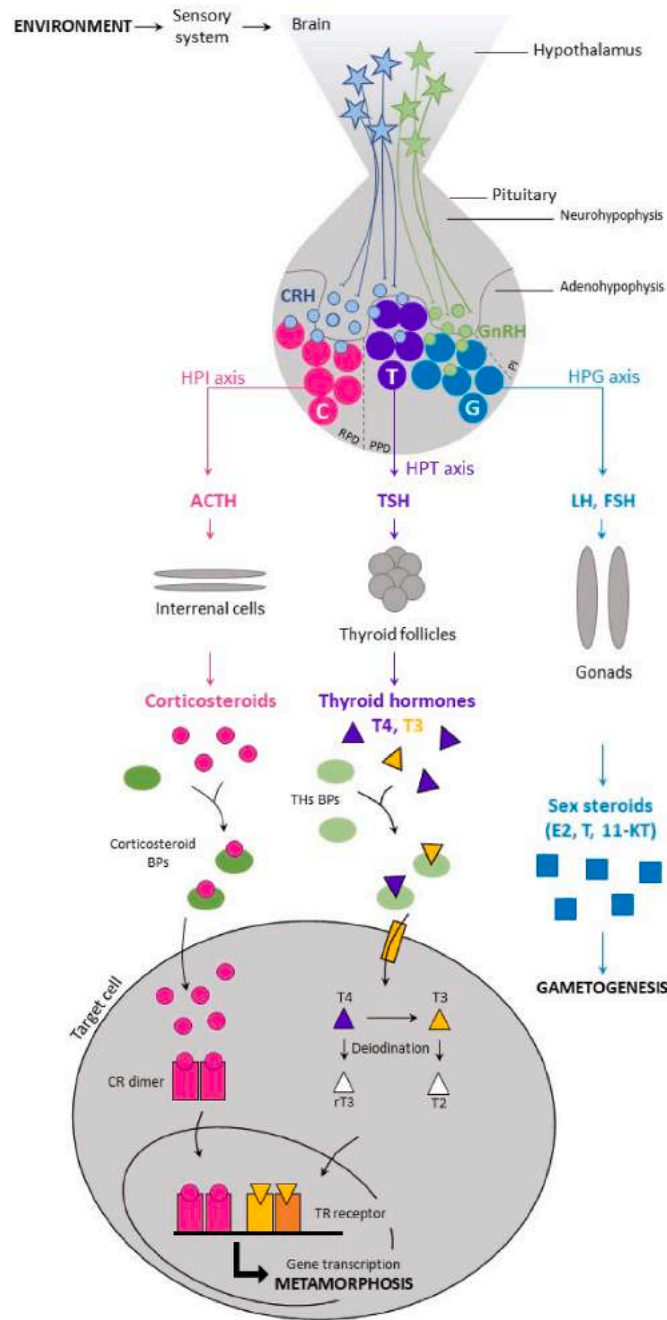


Fig. 2. Modified from [Dussenne et al. 2022](#). Illustration representing the hypothalamo-pituitary-interrenal (HPI), hypothalamo-pituitary-thyroid (HPT), and hypothalamo-pituitary-gonadal (HPG) axes in non-mammalian species (pink, purple and blue, respectively). Environmental cues are perceived by sensory organs and processed in the brain, where corticotropin-releasing hormone (CRH) and gonadotropin-releasing hormone (GnRH) are secreted. In non-mammalian species, CRH is thought to control both the HPI and HPT axes. CRH acts on the pituitary cells to stimulate the synthesis and release of i) adrenocorticotropic hormone (ACTH) from corticotrophic cells (C, pink circles) and ii) thyroid-stimulating hormone (TSH) from thyrotrophic cells (T, purple circles). HPI axis: ACTH stimulates the secretion of glucocorticoids (GCs) by the interrenal cells. Once secreted, GCs are transported in the blood by corticosteroid binding proteins (BPs). GCs enter target cells where they bind to corticosteroid receptor dimers which then translocate into the nucleus where they bind to specific DNA regions to induce gene expression. HPT axis: TSH stimulates the production of thyroid hormones (THs, T4 and T3) by thyroid follicles. THs are transported to target cells via their specific BPs. Inside target cells, THs are substrates of deiodinase enzymes which can both activate (i.e., forming T3) or inactivate (i.e., rT3 and T2) THs. THs are transported to the cell nucleus where they bind to TR receptors, which activate gene transcription, in fine resulting in metamorphosis. HPG axis: GnRH reaches gonadotrophic cells to induce synthesis and release of gonadotropins (LH and FSH). LH and FSH both reach the gonads where they stimulate gonadal maturation and function (i.e., synthesis of sex steroids: E2, T and 11-KT). The balance between these sex steroids, in turn, regulates gametogenesis. RPD: rostral pars distalis, PPD: proximal pars distalis, PI: pars intermedia of the adenohypophysis.

fish, at least by comparison with highly urbanized neighbouring sections of the coastline (Dufour et al., 2009; Bouchoucha et al., 2016; Mercader et al., 2017b; Patranella et al., 2017; Mercader, 2018). Therefore, marinas, harbours, and ports may play a surprisingly vital role in the life cycle of some fish species. This is important when it comes to emblematic protected species, for instance the dusky grouper along the French Mediterranean coastline. This species, which almost disappeared from the French coasts, is almost exclusively found in marine protected areas, and there are very few juveniles. Projects to restore the nursery function in several French and Moroccan Mediterranean ports have led to an increase in observed juveniles (Mercader et al., 2017a; Selfati et al., 2018).

Many marine fish species have structured life histories with two distinct stages: first, a pelagic larval stage capable of long-distance dispersal, followed by a relatively sedentary benthic stage (usually juveniles and adults in reefs or along coastlines; Leis and McCormick, 2002). After the oceanic dispersal phase, larvae settle in a suitable recruitment site. Recruitment is a critical phase in the life cycle of fish as its success is decisive for population maintenance. Fish must undergo an abrupt transformation from larvae to juvenile during this transition phase (Laudet, 2011). This transformation, called metamorphosis, is usually quite spectacular and includes changes in behaviour, morphology, and physiology, including biochemical and histological remodelling that affect several tissues at different levels (McCormick et al., 2002; Parmentier et al., 2004; Frédéricich et al., 2012; Holzer and Laudet, 2015; Besson et al., 2020). Metamorphosis is usually triggered by physiological and environmental cues (Holzer et al., 2017). Indeed, thyroid hormones (TH, notably thyroxine (T4) and triiodothyronine (T3)); Holzer et al. (2017) and cortisol (Jesus et al., 1991) act in concert to control and modulate metamorphosis. TH levels increase during post-embryonic development to reach a peak which induces metamorphosis; TH then coordinate ontogenetic transformations from the larval to the juvenile stages (Gilbert et al., 1996; Wada, 2008; Isorna et al., 2009; McMenamin and Parichy, 2013; Campinho, 2019; Klann et al., 2021). As TH signalling is critical for neurogenesis and the development of sensory organs, it may play a role in determining the “quality” of future juveniles, *i.e.*, their capacity to select suitable habitats, feed efficiently, as well as detect predators and escape from them, which are all crucial for fish to successfully recruit into adult populations (Besson et al., 2020).

3. Effects of multiple anthropogenic stressors on the hormonal system of fish living in harbours

In this review, we discuss on the effect of pollutants and stressors on fish endocrinology, with a focus on three major hormonal axes: the hypothalamo-pituitary-interrenal (HPI), hypothalamo-pituitary-thyroid (HPT), and hypothalamo-pituitary-gonadal (HPG) axes, as described by Fig. 2.

3.1. Plastic pollution

Plastic pollution in the marine environment has become a global environmental concern (Coyle et al., 2020). Plastics are derived from raw natural substances such as petroleum, coal, and gas (Gabbott et al., 2020) and can absorb multiple pollutants from the aquatic environment. The mechanical or chemical impacts of plastics on marine organisms are well documented (Jacob et al., 2020; Welden, 2020). Multiple mechanical impacts have been directly or indirectly linked to plastics including entanglement in macro-plastics (impeding mobility, leading to starvation or drowning), and the ingestion or inhalation of smaller plastic debris, such as micro-plastics (MPs) and nano-plastics (NPs; Welden, 2020). In fish, most ingested MPs and NPs are eventually excreted, but a small fraction can accumulate in organs and tissues (mainly in the intestine and liver) causing severe tissue damage and inflammation (*e.g.*, Jovanović 2017; Wang et al., 2019; Araújo et al.,

2019; Jacob et al., 2021; Marana et al., 2022).

Beyond mechanical risks, marine plastic debris can also act as a reservoir of chemicals that pose potential threats to wildlife. A wide array of contaminants may be contained in plastics: additives such as flame retardants and plasticisers that can leach out from the plastic, or contaminants that get absorbed into plastics from seawater. More than 250 organic compounds have been identified on plastic debris (Campanale et al., 2020). Bisphenol A (BPA), used in the manufacturing of plastics, is found in aquatic environments worldwide, notably harbours, and its concentration is positively correlated with the amount of plastic waste generated in the region (Wu and Seebacher, 2020). BPA is a xenestrogen, *i.e.*, an endocrine disruptor capable of mimicking natural estrogens (Matozzo et al., 2008; Rubin, 2011). BPA and its congeners can alter testis maturation and decreased sperm quality in fish (Chen et al., 2017; Yang et al., 2017) and can induce gonad feminisation (Drastichová et al., 2005; Gonzalez et al., 2021). Males exposed to BPA show increased plasma vitellogenin (VTG) levels or increased *vtg* gene expression in the liver (Sohoni et al., 2001; Van den Belt et al., 2003; Yang et al., 2017; Frenzilli et al., 2021; Gonzalez et al., 2021). VTG is a precursor of the egg-yolk protein; in females, VTG is synthesised in response to estrogens and stored in oocytes. In males, the gene coding for VTG is normally not expressed; its induction is thus used as a marker of estrogen exposure (Matozzo et al., 2008). Together with increased estrogen levels, exposure to BPA is also associated with decreased androgen levels (T and/or 11-KT; Yang et al., 2017; Gonzalez et al., 2021). Bisphenol also acts on the HPG axis, a hormonal cascade starting with the release of gonadotropin-releasing-hormone (GnRH) from GnRH neurons in the pituitary, stimulating the synthesis of gonadotropins, namely luteinising hormones (LH) and follicle-stimulating hormone (FSH). Both hormones together stimulate gonad maturation and the production of sex steroid hormones (in fish, mainly estradiol (E2), testosterone (T) and 11-ketotestosterone (11-KT); Mills et al., 2010, 2018; Zohar et al., 2010). Moreover, in fish, kisspeptin (Kiss) neurons are thought to control the synthesis of gonadotropins, acting either directly on the pituitary cells or indirectly on GnRH neurons (Somoza et al., 2020). Following bisphenol exposure (BPA, bisphenol S, bisphenol F), the expression of *Kiss1*, *Kiss receptor*, *GnRH2*, *GnRH3*, *GnRH receptors*, *fshb*, *lhb*, and *aromatase* genes increases, whereas the expression of genes involved in steroidogenesis decreases (Ji et al., 2013; Qiu et al., 2016; Yang et al., 2017; Gonzalez et al., 2021). BPA and their congeners can have profound effects on fish reproductive systems, raising questions about population persistence in the future.

Other studies have evaluated the effects of BPA on the thyroid system and found disruption of the HPT axis (Zhang et al., 2017; Wei et al., 2018; Lee et al., 2019) initiated by thyrotropin-releasing hormone (TRH) in mammals and corticotropin-releasing hormone (CRH) in non-mammalian species (see Laudet, 2011; Denver, 2017). In fish, when the HPT axis is stimulated, hypothalamic CRH neurons release CRH in the anterior pituitary, which synthesises and releases thyroid-stimulating hormone (TSH) into the global circulation. In turn, TSH reaches thyroid follicles and stimulates the synthesis of thyroxine (T4), which can be converted into its active form triiodothyronine (T3) in peripheral tissues under the action of deiodinase enzymes (Laudet, 2011; Denver, 2017). Several studies have found altered circulating levels of T4 and T3 in individuals exposed to bisphenols compared to control individuals, suggesting a disruption to the HPT axis (Zhang et al., 2017; Wei et al., 2018; Lee et al., 2019). In addition, BPA causes regulatory changes in genes involved in all steps of the HPT axis, for which gene transcription occurs mainly in the brain, pituitary, thyroid, peripheral tissues, and liver (Lee et al., 2019). Interestingly, BPA on one hand and BPS – BPZ on the other hand may have different modes of action on the HPT axis. For example, *in vivo* (zebrafish embryos) and *in vitro* experiments have shown that exposure to BPA upregulated the expression of several genes involved in TH synthesis as well as genes involved in thyroid development (Gentilcore et al., 2013). These results were further confirmed in zebrafish embryos exposed to BPA (*i.e.*,

upregulation of genes directly implicated in thyroid development and function; Lee et al., 2019). On the contrary, BPS and BPZ may deregulate gene expression in the brain and pituitary, notably causing the upregulation of *crh* and *tsh β* gene transcription (at levels above 0.68 mg/L; Lee et al., 2019, higher than those observed in harbours, which are in the few ng/L range; Robinson et al., 2009; Emmet et al., 2020).

A transgenerational experiment showed that the thyroid hormone disruption – lower T4 and higher T3 circulating levels– seen in F0 zebrafish females exposed to BPS from 2 h to 120 days post-fertilisation was transferred to eggs, leading to higher T3 circulating levels in F1 embryos, causing adverse effects (delayed hatching, decreased swimming speed and escape behaviour, and reduced pigmentation; Wei et al., 2018).

Overall, studies have highlighted numerous significant negative consequences of plastics on fish, either mechanically or through chemical impacts on several hormones. BPA is one of many plastic components that cause endocrine disruptions in marine organisms. However, there has been no study on fish living in harbours, although these zones are often exposed to high concentrations of plastics and associated chemicals (Claessens et al., 2011; Naidoo et al., 2015; Romeo et al., 2015; Chen et al., 2021); characterising the effects of plastics on fish in harbours would be an informative research avenue.

3.2. Chemical contamination

In many coastal areas, including marinas, harbours, and ports, the legacy of past industrialisation and the past absence of regulations on emissions is still present. Coastal regions are part of the catchment-coast continuum, where catchment areas are often dominated by intense agriculture, may contain dump sites or landfills, and wastewater discharge is carried across drainage basins into the marine environment. It is therefore unsurprising that chemical contamination is increasing in many harbours and ports across the world (e.g., USA: Dauer et al., 2000; Hong Kong: Nicholson et al., 2011; Brazil: Hatje and Barros 2012; Australia: Mayer-Pinto et al., 2015). Chemical contamination is one of the greatest threats to marine species, leading to impairments in development and reproduction (Rochman et al., 2014), the emergence of diseases (Kiesecker, 2002), declines in biodiversity and ecosystem function (Johnston and Roberts, 2009; Johnston et al., 2015a), and numerous chemical pollutants are thyroid hormone disruptors (Jarque and Piña, 2014).

3.2.1. Persistent organic pollutants

Among the multiple pollutants recorded in harbours, persistent organic pollutants, such as polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs), are the most commonly detected (Casado-Martínez et al., 2009; Ghosh et al., 2003; Mali et al., 2016). Persistent organic pollutants are organic compounds that are resistant to environmental degradation through chemical, biological, or photolytic processes (Jones and De Voogt 1999). These persistent contaminants are usually found only in trace amounts in water but tend to adsorb onto inorganic and organic material settled in sediments (Jones and De Voogt 1999). Persistent organic pollutants are remobilised by human activities such as dredging, rendering them bioavailable (Martins et al., 2012). Consequences on aquatic organisms are numerous: they are immunotoxic, genotoxic, mutagenic, and carcinogenic, and have the potential to disrupt hormonal systems (Tanabe, 2002; Vega-López et al., 2007; Luch and Baird, 2010).

Polychlorinated biphenyls (PCBs), along with other organic pollutants (dioxins, and polycyclic aromatic hydrocarbons PAHs, a diverse group of organic molecules produced through the incomplete combustion of fossil fuels, petroleum, or petrochemical materials), have been strictly regulated since 2001, yet remain present in harbour sediments worldwide (Connell et al., 1998; Adami et al., 2000; Barakat et al., 2002; Ghosh et al., 2003). In Sydney Harbour, 13 of 19 seafood species studied presented tissue concentrations of organic pollutants above standard

thresholds for human consumption (Manning et al., 2017).

Leatherland and Sonstegard (1978) reported changes in thyroid histological appearance and decreased plasma TH levels in coho salmon (*Oncorhynchus kisutch*) and chinook salmon (*Oncorhynchus tshawytscha*) after PCB exposure. Since then, the thyroidal responses of fish to PCBs have shown variable results in different studies, depending on the species and the type of PCB congeners (Brown et al., 2004). A 30-day laboratory exposure to PCB in the diet of juvenile rainbow trout *O. mykiss* reported several detrimental effects, such as oxidative stress, thyroid system dysfunction – higher T4 levels and more active deiodination (Buckman et al., 2007). A similar exposure in young-of-the-year Atlantic croaker *Micropogonias undulatus* decreased both T4 and T3 levels (LeRoy et al., 2006). Altered reproductive physiology has been documented in numerous species (review by Henry, 2015): for instance, female fish fed with PCB mixtures at environmentally realistic concentrations showed impaired ovarian maturation, delayed reproduction, and produced fewer viable eggs (Daouk et al., 2011; Horri et al., 2018).

Life history transitions, such as metamorphosis, are hugely sensitive to endocrine disruption. Larval and juvenile Japanese flounder (*Paralichthys olivaceus*) exposed to PCBs show declines in T3 and T4 levels as well as changes in deiodinase gene expression levels, which were associated with slower growth and delay metamorphosis (Dong et al., 2014, 2017). The effects of PAHs on the thyroid function in fish remain poorly studied (Brown et al., 2004; Kim et al., 2016; Price and Mager, 2020). Nevertheless, a few studies have reported adverse effects on TH. For example, in juvenile common carp (*Cyprinus carpio*), levels of plasma T3 and T4 were reduced in response to PAHs exposure (Shirdel et al., 2016), and similar results were found in zebrafish larvae (Kim et al., 2016). In addition, short-term exposures of water-soluble fractions from oil (PAH compounds) to larval and juvenile turbot (*Scophthalmus maximus*) increased whole-body concentrations of T4, but not T3 (Stephens et al., 1997). Overall, the effects of persistent organic pollutants on the thyroid system are varied and depend on the compound and species. Changes in TH levels in both larvae and juveniles may reflect a complex disruption of biological processes, both in terms of TH synthesis and of their biotransformation in peripheral tissues, which may result in developmental deformities of adults.

3.2.2. Metals

Developing fish (embryos, larvae, and juveniles) are particularly sensitive to heavy metal exposure (Sfakianakis et al., 2015), and certain metals such as Cadmium (Cd) or Copper (Cu) are often present in harbours and ports (Broman et al., 1994; Jupp et al., 2017). Fish embryos exposed to Cd or Cu have lower hatching rates and survival, while young individuals (larvae and juveniles) notably suffer from severe skeletal deformities, impairments of the lateral line, and cardiac oedema (Sfakianakis et al., 2015). Moreover, Cd exposure negatively impacts the function of both the HPT and the HPI axes (HPI ends with the synthesis and release of corticosteroids, mainly cortisol). Indeed, fish larvae exposed to Cd show complete disturbance of gene expression in the HPT axis and reduced circulating plasma levels of T4 (Jancic and Stosic, 2014; Li et al., 2014). Cd can also disturb cortisol synthesis by suppressing the expression of genes coding for proteins which are essential for corticosteroid production (i.e. StAR (steroidogenic acute regulatory protein) and MC2R (Melanocortin 2 receptor, which triggers steroid biosynthesis following activation by ACTH); Sandhu and Vijayan, 2011). Cadmium could potentially have detrimental effects on the metamorphosis and development of young fish and alter the reproductive system of adults. Persistent organic pollutants, such as PCBs, and metals, such as Cd, are examples of a large number of contaminants with varied and potentially cumulative effects.

3.2.3. Pesticides

A commonly used insecticide, chlorpyrifos (CPF), significantly decreases thyroid hormone levels of exposed fish juveniles, thus altering their metamorphosis, leading to decreased intestine lengthening and

impaired grazing behaviour (Holzer et al., 2017). Besson et al. (2020) also showed that CPF and increased temperature, independently as well as synergistically, affect sensory development and predator avoidance behaviour in a coral reef fish (Besson et al., 2020). Their results suggest that TH signalling might be the underlying physiological process impacted by these different stressors, however, each stressor might affect physiological process in unique ways. Increased temperature impacts T4 levels, suggesting an alteration at the neuroendocrine level – before T4 is converted to other compounds such as T3. In contrast, CPF is thought to have a more downstream effect, possibly by acting on T3 metabolism (Besson et al., 2020; Holzer et al., 2017). Fish exposed to these stressors had impaired anti-predator behaviour, similar to that of pre-metamorphosed larvae, and experienced higher mortality from predation. The disruption of TH signalling during fish metamorphosis can cause neurological defects, with possible community-level consequences that may even threaten the maintenance of a species (Crane et al., 2006; Noyes et al., 2009; Laudet, 2011; Holzer et al., 2017; Pinsky et al., 2019). Indeed, changes in survival rates during this transition and post-settlement can have drastic effects on population replenishment. This is of particular concern as nurseries are mainly located in shallow coastal waters and some harbours are even qualified as nurseries (Bouchoucha et al., 2016; Mercader et al., 2017b; Patranello et al., 2017; Mercader, 2018) despite containing multiple stressors that can affect fish metamorphosis and subsequent juvenile quality. As such, harbours are ideal sites to study the impact of multiple stressors on fish endocrine pathways “in situ”, in an ecologically relevant context. Understanding how harbours affect fish hormonal systems during metamorphosis is of great importance from a conservation perspective.

3.2.4. Wastewater discharge

In addition to pesticides, many other contaminants can reach the coast and harbours (endocrine disruptors, pharmaceuticals, etc.) through wastewater discharge (Mossa, 2006; Hamdhani et al., 2020). Among these, triclosan (TCS) is a synthetic chlorinated bactericide used in a vast number of personal care (soaps, toothpastes, deodorants, etc.) and textile products (Adolfsson-Erici et al., 2002). TCS has been monitored in US streams (Kolpin et al., 2002) and coastal waters (i.e., in the Narragansett Bay: Katz et al., 2013; the San Francisco Bay: Jackson and Sutton, 2008; Kerrigan et al., 2015). TCS has structural similarities with THs, suggesting it could disturb thyroid function (Adolfsson-Erici et al., 2002). Studies have investigated the toxicity of TCS on various organisms and reported high mortality rates, deformities, perturbations of behaviour, and reproductive failure (Orvos et al., 2002; Oliveira et al., 2009; Nassef et al., 2010; review by Dann and Hontela 2011). However, few studies have investigated the effects of TCS on fish metamorphosis. In the sheephead minnow *Cyprinodon variegatus*, the T3 peak observed in control fish (indicative of metamorphosis climax) is absent in fish exposed to TCS, resulting in delayed metamorphosis (Schnitzler et al., 2016). Zebrafish larvae exposed to TCS also displayed delayed metamorphosis (Stenzel et al., 2019). In the sole *Solea senegalensis*, TCS had the opposite effect and accelerated metamorphosis (Araújo et al., 2019). The mechanisms of action of TCS remain unclear but are thought to modulate the expression of several genes involved in the HPT axis, notably affecting the expression of TH receptors α and β in the North American bullfrog *Rana catesbeiana* (Veldhoen et al., 2006), and up-regulating of the expression of the *tsh* gene in adult zebrafish (Pinto et al., 2013). TCS is detected in the environment worldwide, and has been recorded in various organisms, including humans (Dann and Hontela, 2011). A better understanding of the risks associated with its use is urgent.

3.2.5. Antifouling compounds

Antifouling compounds are used on many submerged structures (e.g., ship hulls, buoys) to prevent the recruitment of marine organisms, but have many unintended impacts. Tributyltin (TBT) used to be a widespread antifouling agent. Its use became regulated in the 1980s and was

banned in 2008 (IMO, 2005) after it was linked to a global decline in marine molluscs, likely causing reproductive failure (Gibbs and Bryan, 1996; Antizar-Ladislao, 2008). TBT has been shown to delay tadpole metamorphosis and growth in amphibians (Shi et al., 2014) and studies reported altered T4 and T3 levels and HPT gene expression in the zebrafish *Danio rerio*, goldfish *Carassius auratus* as well as in marine rockfish *Sebastes marmoratus* (Zhang et al., 2013, 2016; Li and Li 2021). Recently, medetomidine, a veterinary anaesthetic, has emerged as a new antifouling agent (Dahlström et al., 2000). It is widely used (Wendt et al., 2016) even though its ecotoxicological effects are poorly understood. Recent studies on amphibians showed that tadpoles exposed to medetomidine experienced delayed metamorphosis (Barr et al., 2018; Fong et al., 2018). Medetomidine induces paleness in fish (Bellas et al., 2005; Hilvarsson et al., 2007; Lennquist et al., 2010), which may be due to altered thyroid function, as TH regulates fish pigmentation (McMenamin et al., 2014; Saunders et al., 2019; Salis et al., 2021).

3.2.6. Case study of chemical contaminants: the San Francisco Bay

The San Francisco Bay features on the “303(d)” list of impaired water bodies in the US from the federal Clean Water Act. Chemical contaminants found in the San Francisco Bay include pesticides (diazinon, chlordane, dieldrin, dichlorodiphenyltrichloroethane), polyaromatic hydrocarbons, and chlorinated compounds. Several of the compounds monitored in the bay as well as in marine organisms are sometimes found in concentrations higher than the screening values established by the US EPA (Fairey et al., 1997; Davis et al., 2007; Greenfield and Allen, 2013). To our knowledge, only one study investigated fish TH levels in the vicinity of a harbour in the area (Brar et al., 2010). In this work, young individuals of two fish species, the shiner surfperch (*Cymatogaster aggregata*) and the Pacific staghorn sculpin (*Leptocottus armatus*), were sampled in several locations across the San Francisco Bay and over two consecutive years. Over the 2-year sampling period, both species had reduced T4 concentrations in locations with concentrated human activities (among which the Oakland inner harbour and the San Leandro Bay) compared to other locations (Brar et al., 2010). Fish liver analyses revealed the presence of over 85% of existing PCB congeners, with concentrations inversely correlated to T4 levels. Fishes also had significant differences in T3 concentrations and T3/T4 ratios across the study locations, which might reveal an alteration of deiodinase activity in peripheral tissues. The sampled sculpins may have been undergoing metamorphosis at the time of sampling (as indicated by their body size; Goto 1990). The altered TH levels due to contaminant exposure in the long term and notably during metamorphosis may lead to thyroid dysfunction, alter metamorphosis, and threaten population persistence.

3.3. Invasive species

Harbours, in addition to being subjected to anthropogenic stressors, are very susceptible to invasions of aquatic species. Boets et al. (2012) showed that one fourth of all crustacean species across four Belgian harbours were non-native. Invasive species in harbours mainly arrive in ballast water or from hull fouling (Carlton, 1985; Ruiz et al., 2000; Boets et al., 2012). They can also originate from aquaculture or the aquarium trade (Naylor et al., 2001). The arrival of new species in a harbour can cause interspecific competition with native species, leading to increased predation and stress. In fish, stress is mediated by glucocorticoids, among which cortisol is often measured as an indicator of stress (e.g., Beldade et al., 2017). During a stressful situation, the HPI axis is stimulated, leading to the release of cortisol into the organism (Wendelaar Bonga, 1997). During acute stress, cortisol release is considered adaptive because it stimulates the catabolism of lipids, mobilising energy to cope with the stressful event (Van Weerd and Komen, 1998). However, prolonged stress can have detrimental effects on organisms, including impaired disease resistance, loss of appetite, reduced growth, and altered reproduction (Wendelaar Bonga, 1997; Van Weerd and Komen,

1998). The presence of invasive species which could induce stress in native species has never been explored in fish living in harbours or elsewhere. The only insight available is from a mammal, the Eurasian red squirrel (*Sciurus vulgaris*), faced with the invasion of grey squirrels (*S. carolinensis*; Santicchia et al., 2018). The presence of the invader induced high physiological stress in the native species: faecal glucocorticoid levels of native individuals were related to the abundance of invasive individuals. Whether alien species can induce physiological stress in fish has not yet been studied. Given the highly mobile nature of fish, and because cortisol levels fluctuate extremely rapidly (within a few minutes; Pankhurst 2011), it will be particularly challenging to assess stress levels in response to invasive species from blood samples in harbours.

3.4. Boat noise

Sound travels faster and further underwater compared with in air, and the frequency and intensity of anthropogenic underwater sounds overlap with the ranges of natural biological sounds (Hastings and Popper, 2005; Slabbekoorn et al., 2010). Anthropogenic underwater sounds have therefore been defined as pollutants. Within the EU Marine Strategy Framework Directive (MSFD) (2010/477/EU European Commission Decision), two indicators have been proposed to monitor this pollution: impulsive noise and continuous noise. The anthropogenic contribution to the soundscapes of many harbours worldwide, notably in the northern hemisphere, is dominated by continuous noise caused by commercial shipping (Hildebrand, 2009; McKenna et al., 2013). In the northeast Pacific, noise from commercial shipping in inshore waters is also the most persistent type of anthropogenic noise, notably in large ports such as Vancouver and Seattle (Erbe et al., 2014). In addition, tourism and recreational boating also add a substantial amount of noise to coastal underwater soundscapes (e.g., McDonald et al., 2006; Hermanssen et al., 2019; McCormick et al., 2019). During a lockdown put in place by governments worldwide to slow the spread of COVID-19 (April–May 2020), the vocal activity of a range of coral reef fish living in the marina of Pointe-à-Pitre (Guadeloupe, French West Indies), located next to a major maritime harbour, was reduced as a response to the lower levels of anthropogenic noise they experienced (Bertucci et al., 2021).

Several studies have highlighted the effect of noise on fish physiology, although none have directly tested this on fish living in harbours. The potential stressors caused by anthropogenic underwater noise include behavioural responses, which are mediated through androgen and glucocorticoid pathways (Mills et al., 2020) and may ultimately lead to hearing impairment or habitat abandonment. Noise can affect physiology as early as the embryonic stage in fish, with heart rates shown to increase in staghorn damselfish embryos (*Amblyglyphidodon curacao*) exposed to noise from boats with 2- and 4-stroke engines (Jain-Schlaepfer et al., 2018). This study also demonstrated that heart rate doubled with playbacks of 2-stroke engines compared to 4-stroke engine playbacks. Similarly, heart rate increased by approximately 10% when sounds of small boats powered by 2-stroke engines were played back to embryos of damselfishes *Amphiprion melanopus* and *Acanthochromis polyacanthus* (Fakan and McCormick, 2019). Similar exposures to anthropogenic noise in starved green Chromis (*Chromis viridis*) and spiny damselfish (*A. polyacanthus*) caused a heightened corticosteroid stress response (Armstrong-Smith, 2016). Besides the Pomacentridae family, which has received most attention, boat noise playbacks also increased whole body cortisol levels of a Labridae species, the slippery dick (*Halichoeres bivittatus*; Staaterman et al., 2020). Should this exposure be prolonged, elevated cortisol levels might ultimately impact immune responses and weaken responses to diseases and parasites (Fast et al., 2008). While elevated cortisol levels drove fish to hide in their shelter during anthropogenic noise, circulating androgens such as testosterone, correlated with aggression in orange-fin anemonefish (*Amphiprion chrysopterus*) when exposed to both 30 min and 48 h of boat noise (Mills

et al., 2020). In freshwater environments, playbacks of underwater boat noise recorded from the Danube River and in two Austrian lakes led to increased cortisol secretion in the common carp (*Cyprinus carpio*), the gudgeon (*Gobio gobio*), and the European perch (*Perca fluviatilis*; Wysocki et al., 2006). However, no increase was observed when fish were exposed to continuous random noise. This indicated that irregular noises such as those caused by ship engines, with fluctuations in amplitude and frequency, constitute important stressors. This study also did not find differences in response between species possessing excellent (common carp and gudgeon) or poor (perch) hearing abilities. Nevertheless, species-specific sensitivities and ranges of hearing capacities may alter the extent to which anthropogenic noise impacts them. Motorboat noise also affects interspecies interactions. Bluestreak cleaner wrasses (*Labroides dimidiatus*) inspected their fish clients for longer and were significantly less cooperative during exposure to boat noise (Nedelec et al., 2017). Furthermore, clients did not retaliate as expected (i.e., by chasing) in response to increased cheating by cleaners, suggestive of cognitive impairments due to distraction by both parties. The effects of noise may also depend on multiple acoustic factors, such as sound frequency, intensity, and duration (Slabbekoorn et al., 2010; Hawkins and Popper, 2016).

While some fish species appear to somewhat acclimate to noise in terms of behaviour – with behaviours returning to control levels after continuous noise exposure (Nedelec et al., 2016; Holmes et al., 2017; Staaterman et al., 2020) –, hormonal effects could be long-lived. This has been documented through the increase in aggressiveness and hiding behaviour of *A. chrysopterus*, linked to heightened levels of testosterone and cortisol respectively, after short 30-min boat noise playbacks, but also over longer two-day playbacks, throughout which *A. chrysopterus* did not show any habituation or reduction in response (Mills et al., 2020). It illustrates that changes in the physiological state in response to noise could result in prolonged behavioural effects in reef fish species, which might ultimately lead to alterations in individual fitness and behavioural changes at the population level (Radford et al., 2016; Popper and Hawkins, 2019).

3.5. Artificial light pollution at night

Harbours are one of the largest sources of permanent light on coastlines with light emitted from homes, resorts, and streetlights (Davies et al., 2014). Shallow waters in harbours experience light intensities up to 150–200 lux (Bolton et al., 2017), compared to intensities of 0.03–2.5 lux due to skyglow in other zones (when scattered light from streetlights is reflected by clouds; Perkin et al., 2014), 0.1–0.3 lux during full moonlight on a clear night, and only 0.00003–0.0001 lux on a cloudy night (Rich and Longcore, 2013). The consequences on living organisms are multiple as artificial-light-at-night (ALAN) could affect functions related to orientation in space (phototropism, phototaxis) and in time (circadian rhythms; Falcón et al., 2020).

Circadian clocks are systems that translate environmental information, mainly the alternation of light and darkness (or photoperiod), into the hormonal signal melatonin to orchestrate a myriad of downstream biochemical, physiological, and behavioural events so that the right process occurs at the right time (Falcón et al., 2009; Falcón et al., 2020). In fish, the main source of information on photoperiod and light is from the photosensitive pineal organ, situated at the surface of the brain underneath a translucent window in the skull. The pineal photoreceptors are responsible for the nocturnal production of the time-keeping hormone melatonin, the duration of this nocturnal signal reflecting the duration of the night, while the amplitude varies with temperature in a species-specific manner. Consequently, daily and annual variations in melatonin production provide internal information of daily and annual time, for the orchestration and synchronization of numerous physiological and behavioural processes (Falcón and Meissl, 1981; Underwood, 1989; Falcón et al., 1992, 2009; Grubisic et al., 2019). Other tissues that produce hormonal signals include deep brain

Table 1

Examples of the impacts on endocrine systems of fish of different sources of stress that can be found in harbours (non-exhaustive list).

Category	Source of stress		Impact on endocrine systems	Consequences	Species	Developmental stage	Reference
Plastic pollution	Plastic chemical compounds	Bisphenol A and congeners	<i>HPT axis</i>	Altered circulating T4 and T3 levels, expression of HPT-related genes, hormonal disruption transferred to offspring	Zebrafish <i>Danio rerio</i>	Embryonic, larval, metamorphosis, adult	Gentilcore et al. (2013) Zhang et al. (2017) Wei et al. (2018) Lee et al. (2019)
Plastic pollution	Plastic-sorbed chemical compounds	Bisphenol A	<i>Reproductive system</i>	Altered testis maturation, gonad feminisation, decreased sperm quality, increased plasma vitellogenin levels, altered expression of genes related to the reproductive system	Fathead minnow <i>Pimephales promelas</i> Zebrafish <i>D. rerio</i>	Adult Adult	Sohoni et al. (2001) Van den Belt et al. (2003) Chen et al. (2017) Yang et al. (2017)
					Rainbow trout <i>Oncorhynchus mykiss</i>	Juvenile, adult	Van den Belt et al. (2003)
					Brown trout <i>Salmo trutta</i>	Juvenile	Frenzilli et al. (2021)
					Anemonefish <i>Amphiprion ocellaris</i>	Adult	Gonzalez et al. (2021)
Chemical pollution	Persistent organic pollutants	Polychlorinated Biphenyl	<i>HPT axis</i>	Altered HPT function, decreased TH circulating levels, change in thyroid gland structure In young individuals: Decreased circulating TH levels, slower growth, and delayed metamorphosis	Coho salmon <i>O. kisutch</i> Flounder <i>Paralichthys olivaceus</i>	Juvenile Metamorphosis, juvenile	Leatherland and Sonstegard (1978) Dong et al. (2014) Dong et al. (2017)
Chemical pollution	Persistent organic pollutants	Polychlorinated Biphenyl	<i>Reproductive system</i>	Impaired ovarian maturation, delayed reproduction, fewer viable eggs	Zebrafish <i>D. rerio</i>	Embryonic, larval, metamorphosis, juvenile, adult	Daouk et al. (2011) Horri et al. (2018)
Chemical pollution	Persistent organic pollutants	Bactericide – Triclosan	<i>HPT axis</i>	Disruption of thyroid function, deformities, modified metamorphosis timing	Zebrafish <i>D. rerio</i> Medaka <i>Oryzias latipes</i> Sheepshead minnow <i>Cyprinodon variegatus</i> Sole <i>Solea senegalensis</i>	Embryonic, metamorphosis, adult Embryonic Metamorphosis Metamorphosis	Oliveira et al. (2009) Stenzel et al. (2019) Nassef et al. (2010) Schnitzler et al. (2016) Araújo et al. (2019)
Chemical pollution	Persistent organic pollutants	Polycyclic Aromatic Hydrocarbon	<i>HPT axis</i>	Altered T3, T4, and TSH levels	Turbot <i>Scophthalmus maximus</i> Caspian brown trout <i>Salmo trutta caspius</i>	Metamorphosis Juvenile	Stephens et al. (1997) Shirdel et al. (2016)
Chemical pollution	Heavy metals	Cadmium	<i>HPT axis</i>	Altered gene expression, reduced circulating T4 levels	Chinese rare minnow <i>Gobiocypris rarus</i>	Larval, adult	Li et al. (2014)
Chemical pollution	Heavy metals	Cadmium	<i>HPI axis</i>	Reduced expression of genes linked to corticosteroid production	Rainbow trout <i>O. mykiss</i>	Juvenile	Sandhu and Vijayan (2011)
Chemical pollution	Antifouling	Tributyltin	<i>HPT axis</i>	Altered T3 and T4 circulating levels and gene expression	Rockfish <i>Sebastes marmoratus</i> Goldfish <i>Carassius auratus</i> Zebrafish <i>D. rerio</i>	Adult Adult Adult	Zhang et al. (2013) Zhang et al. (2016) Li and Li (2021)
Chemical pollution	Antifouling	Medetomidine	<i>HPT axis</i>	Paleness in fish – pigmentation linked to TH (McMenamin et al., 2014; Saunders et al., 2019; Salis et al., 2021)	Lumpfish <i>Cyclopterus lumpus</i> Atlantic cod <i>Gadus morhua</i> Turbot <i>Psetta maxima</i> L.	Larval Larval Juvenile	Bellas et al. (2005) Bellas et al. (2005) Hilvarsson et al. (2007)
Chemical pollution	Pesticides	Chlorpyrifos	<i>HPT axis</i>	Decreased circulating TH levels, impact on T3 metabolism during		Metamorphosis	Holzer et al. (2017)

(continued on next page)

Table 1 (continued)

Category	Source of stress		Impact on endocrine systems	Consequences	Species	Developmental stage	Reference
Light pollution	Artificial light at night	Disruption of the circadian cycle	Reproductive system	metamorphosis. Impaired anti-predator behaviour post-metamorphosis.	Surgeonfish <i>Acanthurus triostegus</i>		Besson et al. (2020)
				Suppression of LH and FSH synthesis	European perch <i>Perca fluviatilis</i>	Juvenile	Brüning et al. (2016, 2018)
Light pollution	Artificial light at night	Disruption of the circadian cycle	Reproductive system	Reduction in circulating levels of male and female sex hormones (11-ketotestosterone for males, 17 β -estradiol for females)	Roach <i>Rutilus rutilus</i>	Juvenile	Brüning et al. (2016, 2018)
					European perch <i>P. fluviatilis</i>	Juvenile	Migaud et al. (2004)
					Seabass <i>Dicentrarchus labrax</i>	Juvenile	Rodríguez et al. (2005)
					Sole <i>S. senegalensis</i>	Juvenile	Felip et al. (2008)
					Roach <i>R. rutilus</i>	Juvenile	García-López et al. (2006)
Light pollution	Artificial light at night	Disruption of the circadian cycle	Reproductive system	Altered gonad maturation	Atlantic cod <i>Gadus morhua</i>	Juvenile	Brüning et al. (2018)
					Turbot <i>Scophthalmus maximus</i>	Juvenile	Begtashi et al. (2004)
					Nile tilapia <i>Oreochromis niloticus</i>	Juvenile	Taranger et al. (2006)
					Seabass <i>D. labrax</i>	Juvenile	Immsland et al. (2003)
					Atlantic cod <i>Gadus morhua</i>	Juvenile	Rad et al. (2006)
					Turbot <i>Scophthalmus maximus</i>	Juvenile	
					Nile tilapia <i>Oreochromis niloticus</i>	Juvenile	
Temperature	Uncommon temperature conditions	Temperature changes in semi-enclosed harbours	All hormones	Modification of hormone synthesis and tissue sensitivity, impact on numerous processes	Numerous fish species	Embryonic, larval, metamorphosis, juvenile, adult	Review by Little (2021)
Temperature	Uncommon temperature conditions	Increase in average temperature	HPT axis	Modified T4 levels. Impaired anti-predator behaviour post-metamorphosis)	<i>A. triostegus</i>	Juvenile	Besson et al. (2020)
Invasive species	Competitors with native species	Long term competition	HPI axis	Increased stress – release of cortisol, mobilisation of energy. In the long-term, impaired disease resistance, loss of appetite, reduced growth and reproduction, more hiding behaviour	No fish study yet. Squirrel <i>Sciurus vulgaris</i>	Adult	Santicchia et al. (2018)
Sound pollution	Boat noise	Long term engine noise exposure	HPI axis	Increased stress – release of cortisol, mobilisation of energy. In the long-term, impaired disease resistance, loss of appetite, reduced growth and reproduction, more hiding behaviour	Anemonefish <i>Amphiprion chrysopterus</i>	Adult	Mills et al. (2020)
					Wrasse <i>Halichoeres bivittatus</i>	Adult	Staaterman et al. (2020)
					Anemonefish <i>A. chrysopterus</i>	Adult	Mills et al. (2020)

photoreceptors, the saccus vasculosus, or photoreceptors in the retina (Kojima et al., 2000; Philp et al., 2000; Peirson et al., 2009; Falcón et al., 2010; Nakane et al., 2013). The main hormone controlling circadian activities, pineal melatonin, is produced during the night and released in the cerebrospinal fluid and blood to control locomotor activity, food intake, shoaling, and diel vertical migration (Ekström and Meissl, 1997, 2004; Ryer and Olla, 1998; Mehner, 2012; Vowles et al., 2014). Pineal melatonin also regulates seasonal rhythms including reproduction, growth and development, sleep rhythms, rhythmicity of locomotor activity, and immune responsiveness (Duston and Bromage, 1986; Fairey et al., 1997; Boeuf and Le Bail, 1999; Downing and Litvak, 2002; Wiechmann and Sherry, 2013). However, more and more studies warn about the potentially disruptive effects of ALAN on the synchronization of biological functions and life cycle of species, questioning their sustainability on impacted areas, such as harbours.

Almost all studies in fish have shown that broad-spectrum white light, even as little as 1, 0.1 and 0.01 lux, reduces the nocturnal production of plasma melatonin across fish species. ALAN reduced melatonin in temperate freshwater fish such as goldfish (*C. auratus*; Kezuka et al., 1988; Iigo et al., 1997), pike (*Esox lucius*; Falcón et al., 1987, 1989), zebrafish (Khan et al., 2018) and other freshwater species (Porter et al., 2001; Vera et al., 2005; Brüning et al., 2015, 2018b), as well as in

marine species such as the Atlantic salmon (*Salmo salar*), European sea bass (*Dicentrarchus labrax*) and Arctic charr (*Salvelinus alpinus*; Porter et al., 2001; Bayarri et al., 2002; Rahman et al., 2004; Vera et al., 2005; Nikaido et al., 2009; Park et al., 2014; Choi et al., 2017; Liu et al., 2019). Tropical marine species have shown some of the strongest responses to light at night of 1 lux (e.g., Rahman et al., 2004; Nikaido et al., 2009; Carazo et al., 2013; Park et al., 2014). Increasing light intensity had little to no effect suggesting that there is likely a threshold level of ALAN near 1 lux that alters the circadian rhythm in fish (Brüning et al., 2015, 2016; 2018a). Even lower light levels, such as skyglow (0.01–0.1 lux), suppress nocturnal melatonin levels (Kupprat et al., 2020): the strong light pollution observed in harbours can be expected to suppress melatonin secretion at night in fish.

The pineal gland develops early and detects light even during early embryonic stages (Ekström et al., 1983; Östholm et al., 1987; Ekström and Meissl, 1997). Egg hatching, which normally takes place under the cover of darkness, is impacted by the disturbance of the circadian rhythm by ALAN. Egg hatching was delayed in the European perch *Perca fluviatilis*, roach *Rutilus rutilus*, and bleak *Alburnus alburnus* (Brüning et al., 2011) and completely prevented in laboratory studies of tropical fish species (Fobert et al., 2019). One explanation may be that the modified ratio of prolactin (a pituitary hormone) and melatonin resulted

in a change in the egg hatching enzyme (Brüning et al., 2011).

Melatonin also regulates the secretion of several components of the HPG axis, such as gonadotropins, sex steroids or gonadal maturation (Khan and Thomas, 1996; Amano et al., 2000; Chattoraj et al., 2005; Bhattacharya et al., 2007; Sébert et al., 2008; Carnevali et al., 2011). Experimental white light levels as low as 1 lux (but not other wavelengths of light such as blue, green, red) as well as street-lighting in a natural setting (13.3–16.5 lux) suppress mRNA expression of LH and FSH in male and female European perch and roach (Brüning et al., 2016; 2018b). ALAN also reduces circulating sex hormones, 17 β -estradiol in female European perch and roach (Brüning et al., 2018b), Senegalese sole (*Solea senegalensis*, García-López et al., 2006), and in male and female perch (Migaud et al., 2004), as well as 11-ketotestosterone in male European perch and roach (Brüning et al., 2018b), sea bass (Rodríguez et al., 2005; Felip et al., 2008) and Senegalese sole (García-López et al., 2006). The cascading impacts of ALAN on sex hormones cause a subsequent failure in gonad maturation for several fish species, including sea bass (Rodríguez et al., 2005), Atlantic cod (*Gadus morhua*, L.; Taranger et al., 2006), turbot (*Scophthalmus maximus*; Imsland et al., 2003) and Nile tilapia (*Oreochromis niloticus*; Rad et al., 2006).

The impact of ALAN on fish cortisol levels, a commonly measured indicator of stress (Mommson et al., 1999), has been mixed. ALAN increased plasma cortisol and glucose in farmed Atlantic salmon (Migaud et al., 2007) but did not affect cortisol levels in dispersing Atlantic salmon fry (Newman et al., 2015). There was also no effect of ALAN on the cortisol response of European perch (Brüning et al., 2015), red sea bream (*Pagrus major*) and striped knifejaw (*Oplegnathus fasciatus*; Biswas et al., 2006, 2008), Bluefin tuna (*Thunnus orientalis*; Honryo et al., 2013) or juvenile bonefish (*Albula vulpes*, Szekeres et al., 2017). To date, the evidence shows that the hormonal impacts of ALAN in fish are driven by the changes of melatonin levels rather than cortisol levels.

In most cases, all the above experiments on ALAN were completed in laboratory settings lacking many environmental and ecological factors, such as species interactions between predators and prey, as well as refuges linked to habitat complexity (Brüning et al., 2015; Schligler et al., 2021). It is therefore unclear whether ALAN in more natural settings, and over longer periods of time, will lead to similar alterations to hormone levels. However, one recent study in a natural setting showed that long-term exposure to light pollution over 18–23 months negatively impacts the survival and growth of a wild coral reef fish (Schligler et al., 2021). The higher mortality of orange-fin anemonefish, *A. chrysopterus*, in the wild (Schligler et al., 2021) agrees with a laboratory study showing increased predation under ALAN of coral reef fish larvae, the convict tang, *Acanthurus riostegus* (O'Connor et al., 2019). Despite no studies to date on the impact of ALAN specifically in harbours, the disruption to hormonal rhythms, including those associated with gonadogenesis, as well as the impacts of growth and survival, strongly suggests that the high levels of light pollution in harbours will have downstream consequences for fitness and population dynamics (Brüning et al., 2018b).

3.6. Temperature variations in water exchange-limited harbours

The often-enclosed topography of harbours can prevent efficient water cycling with the open ocean. This can cause higher variation in temperature than in “open” coastal areas (i.e., the water tends to be colder in the winter and warmer in the summer). Harbours can hence potentially provide a natural setting in which to study of the effects of temperature variation on hormones and hormone-dependent physiological processes. Multiple studies have highlighted the effect of temperature on a range of processes – e.g., metabolic rates in zebrafish, ATP production in mosquitofish *Gambusia affinis* – controlled by thyroid hormone action in organisms (cf. review by Little 2021). In addition, temperature variation in harbours will only be heightened by climate change. Fish are most commonly ectotherms (Little, 2021), and thus do not regulate their internal temperatures, but rather adapt their

metabolic processes to environmental conditions. This fine-tuning is strongly reliant upon the thyroid axis for numerous taxa: it can be based on the regulation of T3 and T2 levels (in zebrafish; Little et al., 2013) but it also depends on other parts of the signalling cascade, such as the temperature-sensitive differential expression of nuclear thyroid receptors (in tadpoles; Nakajima et al., 2020). Variations in temperature thus have direct effects on the thyroid hormone signalling cascade and can be expected to modify the physiological responses controlled by thyroid hormones (higher T2 and T3 levels in warm conditions, higher metabolic rate, and transcription of ATPase in cold conditions; Little et al., 2013). Furthermore, the combination of other stressors impairing thyroid hormone pathways and heightened temperature variability can have negative impacts on fish. For instance, hypothyroid zebrafish have been found to have impaired swimming performances under cold conditions (Little et al., 2013). Harbours can thus be key sites to study the cumulative consequences of numerous stressors and stronger temperature variability than in natural coastlines with more efficient water renewal.

3.7. Combined effects of anthropogenic stressors

Anthropogenic stressors (e.g., chemicals, plastics, boat noise, ALAN, and invasive species) thus occur in harbours and some of their effects on fish endocrine systems are already documented, mostly based on laboratory studies (non-exhaustive list in Table 1). Laboratory studies are particularly suited to isolate causal relationships between a type of contaminant and physiological issues (e.g., Grinwis et al., 2000), and these relationships are extrapolated to predict field conditions (Ankley and Villeneuve 2006). However, laboratory studies have numerous limitations when compared to field studies, which is an issue raised by ecotoxicologists over the past decades (Kimball and Levin, 1985, Chapman 2002): they consist of artificial set-ups, with exposure routes differing between the laboratory and field-exposed fish, chemical contaminants often at higher concentrations than those found in harbours, restrained times of exposure to one or multiple contaminants, low number of successive generations, and limited experimental set-up and space. Crucially, they do not replicate all conditions experienced by wild organisms, in particular combinations of numerous stressors (Crain et al., 2008) at various levels of acuteness through time (e.g., varying bioavailability of chemical pollutants depending on water parameters (Ankley and Villeneuve 2006). Wider-scale laboratory studies with microcosms or mesocosms have been suggested (Kimball and Levin, 1985) but involve complex protocols and are again limited in scope. Experimental manipulations of whole ecosystems require numerous impact assessments (e.g., recent experiment with wastewater release into a confined stream, with before-after/control-impact; Pereda et al., 2020) and would be difficult to put in place in open water or coastal settings where target fish species grow. However, harbours can be considered as large-scale and long-term ecosystem manipulations and thus provide an ideal study ground to assess the effects of combined anthropogenic stressors on fish.

Indeed, combinations of stressors may have cumulative effects on resident organisms. In the laboratory, synergistic effects of chemical pollutants on TH homeostasis have been identified in rats exposed to a range of endocrine disruptors (dioxins, PCBs, polybrominated diphenyl ethers), with combined impacts higher than in a simple dose additive manner (Crofton et al., 2005). The combination of invasive species and heavy metal pollution in harbours has unexpected outcomes: copper and tin contamination result in the recruitment and growth of various invasive species above ‘natural’ levels in high traffic areas of Sydney harbour, while the recruitment of native species decreases (Piola and Johnston, 2008; Dafforn et al., 2009). Increased larval recruitment of invasive species and increased metal contamination may therefore act in combination to diminish the abundance of native species (Dafforn et al., 2009). The additional impact of climate change and/or ocean acidification onto these local anthropogenic stressors is also predicted to be

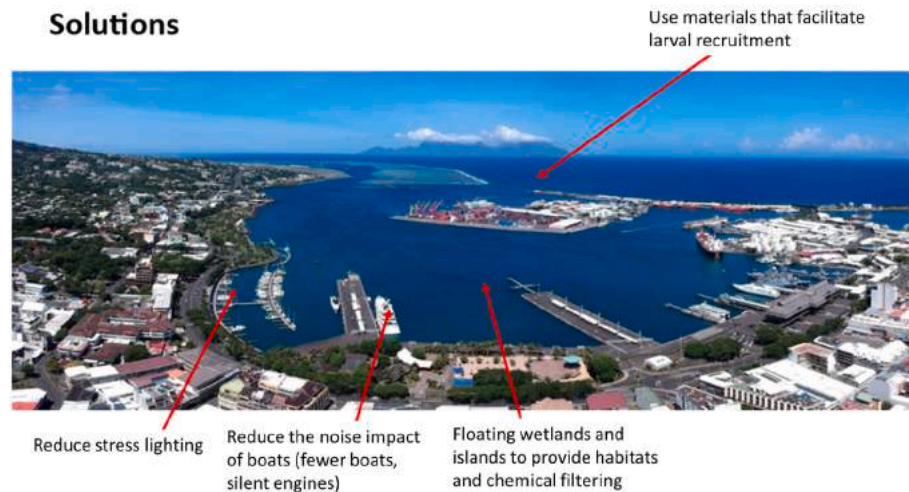


Fig. 3. Illustration of different precautionary measures to improve survival and health in marine organisms that could be adopted in the port of Papeete, Tahiti, French Polynesia.

largely additive. Increased temperature and decreased pH may induce a higher toxicity of many common contaminants in harbours (Crain et al., 2008). Due to the complexity and importance of understanding how global change and multiple local anthropogenic stressors interact, determining the combined impacts of stressors on fish hormonal systems should be a priority to inform managers and stakeholders and improve conservation practices in harbours.

4. Recommendations for future studies

Harbours are often considered as secondary sources of coastal pollution, notably when compared with direct wastewater inputs. This review emphasises the lack of studies on the endocrine systems of fish in harbours, despite harbours being potential *in situ* laboratories where multiple stressors occur simultaneously. Characterising the interactions between the hormonal systems of fish and a wide range of stressors in harbours can shed light on numerous issues: how are different endocrine pathways affected by multiple stressors? What consequences do stressors have on the biology and fitness of fish and on their distribution and abundance (*i.e.*, consequences from the molecular and organism level to the population and ecosystem level)? Such integrative studies would provide insights for the development of effective policies to manage and preserve marine biodiversity and would guide solutions to enhance water quality in harbours. Recommendations for future studies are detailed below.

4.1. Choice of target fish species

To assess the need for harbour management plans, indicator species can be used. Ford et al. (2005) focused on changes in the structure and function of the microbial community in New Bedford Harbour (USA) in response to toxic contaminant exposure, with the goal of using microbes as ecotoxicological tools. Sessile invertebrates are often used to monitor harbour environments as harbours serve as sinks for pollutants that bind to sediments and contaminants, such as heavy metals and hydrocarbons, accumulate in the tissues of benthic marine species (*e.g.*, marine mussels; Wade et al., 1998; Corsi et al., 2005). Corals have also been identified as environmental indicators of heavy metal contamination and have been used to identify pollution hotspots in Safaga harbour (Egypt; Shabib et al., 2021). Mobile fish species can also be used as key species to reflect the water quality of harbours, as organic contaminants and trace metals can also accumulate in fish tissue (Bolton et al., 2004). Studies conducted in Vancouver Harbour and in the neighbouring Puget Sound

estuary reported toxicopathic liver lesions in the English sole *Pleuronectes vetulus*, a benthic species, which were associated with sediment contaminant levels (Goyette, 1988; Myers et al., 1990, 1998). Velusamy et al. (2014) confirmed that benthic species are well-suited to assess the impacts of sediment-associated pollution as trace metals highly accumulated in demersal fishes, followed by neritic and pelagic fishes in Mumbai harbour. Thus, benthic fish species (as opposed to pelagic and demersal fish species) could be used in harbours not only as a bio-indicator of marine ecosystem health, but also to study the relationship between sediment contaminant concentrations and the prevalence of diseases. Benthic fish species, due to their ecological niche, cannot avoid environmental stressors, and can be sensitive indicators of natural and anthropogenic disturbances in harbours. Conversely, in the event of restoration action in ports, the species to be favoured will be demersal species, which are independent of the substrate (*e.g.*, most coastal fish typical of rocky substrate in the case of the Mediterranean Sea).

4.2. Increasing larval recruitment in harbours

Numerous management solutions for harbours have focused on promoting larval recruitment in harbour areas. However, the relevance of such solutions can be questioned as harbours are generally unfavourable habitats for fish growth and quality. Larval fish recruiting in harbours are exposed to multiple stressors that may impact their ability to grow properly (deformity), to metamorphose, to reproduce (sterility), or to protect themselves from predation. Why should fish recruitment be promoted in harbours if fish survival and metamorphosis are uncertain, and hence if the renewal capacity of fish stocks is not guaranteed? The first ecological restoration actions in ports to rehabilitate the nursery function are encouraging as they show greater species diversity, greater abundance, and better survival (Bouchoucha et al., 2016; Mercader et al., 2017). This is linked to the fact that the artificial habitats only target demersal species which have food sources independent from polluted harbour substrates. Bouchoucha et al. (2018) found little to no significant differences between the concentrations of some heavy metals in the muscle tissues of seabream juveniles in a harbour and a natural area. Studies testing the synergistic effects of the multiple stresses present in harbours on the hormonal systems of fish are yet to be performed. Research must focus on these questions to clarify whether the restoration of juvenile nursery areas in harbours is efficient in the long term to mitigate the impacts of coastal modifications. Answering this question is crucial to implement relevant and efficient conservation actions. Indeed, even if harbours attempt to limit their impacts, for instance by obtaining

ecological labels (which should be the direction to follow; [Satir and Doğan-Sağlamtimur, 2018](#)), the creation of a marine protected area in their vicinity could be a better option. However, not all areas are suitable for the implementation of protection measures and restoring degraded habitats in harbours should be considered as a complementary step. Conservation and restoration actions must be adapted to local characteristics and specificities. In any case, the priority remains preventing the degradation of ecosystems and associated ecosystem services.

4.3. Recommendations to harbour authorities and researchers

Environmental protection perspectives in harbours should focus around four themes: 1) an analysis of the nursery function of the harbour, through a dynamic and quantitative description of the recruitment of juvenile and of adult fish populations on a long-term basis (as practiced in French Mediterranean harbours which benefit from ecological restoration action); 2) a definition of the physiological “quality” of juvenile fish (eco-physiological status of the recruits, established by measuring the levels of thyroid hormones and baseline cortisol levels); 3) compensation solutions to protect or restore the nursery function of harbours (e.g., [Bouchoucha et al., 2016](#); [Mercader et al., 2017b](#); [Mercader 2018](#)); and 4) proposals for good practices and communication with the general public about possible ecological restoration solutions in and near harbours. The interactive and cumulative effects of anthropogenic stressors on marine fishes in harbours remain largely unknown. These knowledge gaps must be addressed and used along with precautionary principles to develop coherent environmental protection plans. However, based on precautionary principles, the following guidelines can already be formulated ([Fig. 3](#)): put in place structures to provide shelters for marine species; clean up and reduce the inputs of plastics and chemical pollution (notably with nature-based methods, such as putting in place wetlands that can filter water in addition to providing habitats; [Wood 1995](#)); avoid the discharge of ballast water; reduce the speed of boats to lessen sound pollution; reduce light pollution at night.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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