Effects of Commercial Finfish Net Pen Aquaculture on Habitats on Washington State-Owned Aquatic Lands

Washington Dept. of Natural Resources Aquatic Division
Date of Draft: 10/21/2024

Contents

i. Executive Summary

This summary shares key findings from a synthesis of the best available science concerning interactions of commercial finfish net-pen aquaculture (CFNPA) with aquatic habitats of interest to Washington State. CFNPA refers to a system of nets, cages, or other containment systems in open water used to cultivate, feed, and raise finfish to marketable size for the purpose of harvesting and selling the fish as a crop. CFNPA does not include operations and containment systems used to raise finfish for open-water release or used to raise finfish solely for tribal ceremonial and subsistence uses. While other reviews of the science in Washington have included impacts to water quality and to fish and wildlife (such as the transfer of disease and genetics) (Hawkins et al., 2019; Washington State Department of Ecology, 2022), DNR has focused this review on potential stressors to the habitat the agency is specifically charged with conserving for future generations.

The science synthesis summarizes literature on habitat interactions with CFNPA based on investigations regionally and globally – including effluent, marine debris, and shading. Based on current siting guidance and best practices, effluent and marine debris were deemed the most applicable interactions among modern CFNPA and habitats on SOAL and are outlined in this executive summary.

Habitats and Habitat Elements

- Native aquatic vegetation - eelgrass beds, kelp forests, and other macroalgae - are recognized for their ecological, economic, and sociocultural values. Macroalgae can range from the intertidal to 30m depth (e.g., crustose corallines).
- Benthic sediments foster a high diversity of macro and micro invertebrates which are critical for the storage and cycling of organic matter and carbon - forming the base of many food chains. Macro invertebrates such as octocorals, gorgonians, and sea pens function as highly rugose habitat in deeper waters (e.g., the cloud sponge ranges from 10m to 1600m depth). These sediment dwelling species are sensitive to changes in sediment and water quality.
- Shellfish habitats and harvest areas contribute a suite of ecological services including provisioning important biogenic nursery habitat, the cycling of nutrients, filtering of pathogens, maintenance of water quality, carbon sequestration, and shoreline stability.

Effluent

- CFNPA effluent is predominantly organic, largely arising from fish feed and its byproducts (e.g., uneaten feed and digested feed excreted to the environment); biofouling material falling off the nets; and fish carcasses (Kalantzi & Karakassis, 2006). Periodic treatment of sea lice can lead to circulation of the pesticide through the surrounding environment, and therapeutics such as antibiotics can also circulate beyond the net pen. Trace metals such as zinc and copper have been found up to 150m from former CFNPA facilities in Canada (Hamoutene et al., 2021).
- Beneath and proximal to CFNPA facilities, impacts to benthic communities include increased oxygen demand and resulting lower-oxygen conditions, reduced species diversity, increased tissue necrosis for some invertebrates, the accumulation of sometimes lethal levels of phytotoxic sulfides in the root zones where aquatic vegetation resides, and shifts in bacterial communities. Climate change impacts such as events of increased water temperature, decreased pH, and decreased dissolved oxygen may lead to greater cumulative effects by posing additional stressors to benthic communities.
- The impact of a CFNPA effluent on benthic communities depends on loading (the quantity and concentration of organic matter deposited); water depth and current velocity (greater dispersion in deeper and faster waters); pen configuration (orientation to current and relative location of pens can impact dispersion); feed type (different settling rates and digestibility); and the composition of the

benthic community beneath the farm (more diverse and productive communities may assimilate waste products more readily but also may experience more impacts)(Hawkins et al., 2019; Parametrix, 1990; Washington State Department of Ecology, 2022). Accounting for these factors can reduce likelihood or extent of impact, reflected by reduced intensity and duration of impact from effluent in modern CFNPA (Rust et al., 2014). Accounting for these factors can reduce likelihood or extent of impact, reflected by reduced intensity and duration of impact from effluent in modern CFNPA (Rust et al., 2014). Accounting for these factors can reduce likelihood or extent of impact, reflected by reduced intensity and duration of impact from effluent in modern CFNPA (Rust et al., 2014). Accounting for these factors can reduce likelihood or extent of impact, reflected by reduced intensity and duration of impact from effluent in modern CFNPA (Rust et al., 2014).

Marine Debris

- CFNPA can produce marine debris, including plastics from sources such as floating collars or handrails; collar floatation; buoys from mooring systems; ropes from mooring systems; net enclosures; predator nets; and feeding systems.
- The impact of CFNPA marine debris on benthic communities depends on exposure to storm or high wave events (which may increase in frequency and/or severity with climate change), waste management practices (methods to reduce waste and to recognize release, and to subsequently respond to that release), installation maintenance (to prevent wear and failure), and/or debris survey methods during the closure of a facility (to ensure any associated debris from the structure is removed) (Hawkins et al., 2019; Huntington, 2019; Washington State Department of Ecology, 2022). Accounting for these factors can reduce likelihood or extent of impact.

Cumulative effects

- When individual stressors are combined, they can have an additive or multiplicative impact–these are known as cumulative effects. These stressors can act towards one another and towards other natural processes, creating a combined impact to the environment that is both different from, and more significant than, an individual impact. For instance, climate change leads to increased intensity of marine heat waves and low-oxygen events, which could impact benthic communities already impacted by CFNPA effluent. Climate-related increased storm activity could also lead to increased risk of marine debris.
- The 2022 multi-agency guidance on CFNPA in Washington advises the consideration of cumulative effects from facility characteristics (e.g., size of farm, presence of nearby farms, dissolved oxygen levels of surrounding habitat) when siting a facility (Washington State Department of Ecology, 2022).

Risks and Uncertainties

- While there is a considerable amount of knowledge surrounding the effects of CFNPA and related stressors on habitats and habitat elements from different parts of the world, there is relatively little information on effects within the Salish Sea. Major knowledge gaps include the effects of primarily effluent, but also shading, marine debris, and hydrodynamics, from CFNPA on sensitive marine habitats within Puget Sound.
- Effects of CFNPA and associated interactions may be mitigated with up-to-date practices, appropriate siting, and frequent monitoring-however, it is difficult to predict their effects on a system lacking prior knowledge and understanding of their effects in that system, and the potential for cumulative effects from other stressors such as climate change.

1. Introduction

The purpose of this document is to summarize the best available science concerning the interactions of commercial finfish net pen aquaculture (CFNPA) with habitats relevant to Washington State Owned Aquatic Lands (SOAL), and documented approaches to reducing stressors from those interactions. CFNPA means a system of nets, cages, or other containment systems in open water used to cultivate, feed, and raise finfish to marketable size for the purpose of harvesting and selling the fish as a crop. CFNPA does not include operations and containment systems used to raise finfish for open-water release or used to raise finfish solely for tribal ceremonial and subsistence uses. While other reviews of the science in Washington have detailed impacts to fish and wildlife such as through disease and genetics (Hawkins et al., 2019; Washington State Department of Ecology, 2022), DNR has focused this review on potential stressors to the habitat the agency is specifically charged with conserving for future generations.

Washington SOAL support a diversity of marine and diadromous species owing to the variety of habitats and habitat elements it fosters and supports. Some areas are shallow enough for sunlight to penetrate to the sediment surface, allowing for the growth of submerged vegetation. Throughout SOAL, sediments host abundant epibenthic and infauna communities that act as ecological subsidies for other ecologically and economically important species while also facilitating important chemical and physical processes. When these habitats are damaged or destroyed, the consequences extend to the broader ecosystem and to organisms well beyond the site of impact. This not only threatens the ecological structure and function of ecological communities, but the economic and sociocultural stability of industries and communities throughout Washington.

CFNPA of Atlantic salmon (*Salmo salar*) has occurred in the Salish Sea Region of Washington (USA) and British Columbia (Canada) beginning in 1969 (Weston, 1986), and more intensively in the Puget Sound region in the 1980s (Nash & Waknitz, 2003). While there is limited data on the specific interactions of CFNPA on habitats and habitat elements in Washington State, this document compiles the best available science to present how certain operations, infrastructure, and mechanisms of net pen aquaculture may influence the health of local nearshore habitats.

From the outset of CFNPA in Washington, certain siting guidance has been recommended from government and non-government experts. The guidance documents have stressed that site-specific conditions may influence guideline applicability on a case-by-case basis (Washington State Department of Ecology, 1986). Most importantly, the guidance dictates the importance of CFNPAs to be sited both away from critical or sensitive habitats (including ‘Habitats of special significance’) and in conditions of high current flow and at water depths for which the rate of organic input does not exceed the assimilative capacity of the benthos and sufficient flushing is present – (roughly deeper than 60 ft). Additionally, if habitats of special significance are found in depths greater than 75 ft, it is recommended that a CFNPA facility not be sited within 300 feet of it in the prevailing tidal current direction, or within 150 feet in any other direction (Washington State Department of Ecology, 1986). Additional guidance from 2022 suggests (in addition to other considerations) that CFNPAs be located 1,500 – 2,000 ft offshore, with at least 15 m between the bottom of pens and the underlying substrate, away from areas either prone to algal blooms or where flow may bring waste into shallow or enclosed areas, and away from areas of Puget Sound with already low levels of dissolved oxygen (Washington State Department of Ecology, 2022). Critical areas for fish and wildlife habitat conservation ([RCW 36.70A.030(5)]) which may include shellfish areas, have specific requirements under counties' Shoreline Management Program (SMP) for buffer distances. (Washington State Department of Ecology, 2022).

DNR has incorporated these recommendations into 'habitat stewardship measures' (HSMs) which are incorporated into use-authorization agreements for CFNPA on SOAL on a case-by-case basis. The following habitat-focused measures comprise some of those HSMs considered for CFNPA agreements: areas that are subject to chronic oxygen depletion and persistent nitrogen depletion in surface waters are not recommended for net pen aquaculture (specific areas where these conditions occur include Budd Inlet, Holmes Harbor, and Hood Canal); minimum depth and current guidelines based on the size class of the CFNPA facility must be followed as per guidance in the 1986 interim guidelines; CFNPA may not be sited within 300 feet (90 meters) of certain habitats (ex., kelp and eelgrass, rocky reef, commercial geoduck tract, other shellfish beds) if those habitats are located in depths less than 75 feet at mean lower low water (MLLW); CFNPA may not be sited within 1,500 feet of seal and sea lion haul-out sites, seabird nesting sites or colonies, or critical areas for feeding/migration of birds and mammals; and CFNPA must not be sited within 300 feet of wildlife refuges, aquatic reserves, marine protected areas, and habitats of endangered or threatened species (regardless of current direction).

Section 2 of this document describes priority habitats on state-owned aquatic lands, in terms of their ecological functionality and status. Section 3 draws from best available science on interactions of CFNPA with habitats (via effluent, shading, and marine debris) to describe: (1) how prevention, mitigation, and response have addressed those interactions; (2) global and regional examples of interaction impacts to habitats; and (3) which factors can lead to an increased impact from interactions. The interactions presented here also have *cumulative effects* on the habitat or habitat elements of concern. Section 4 discusses the potential impact of multiple stressors on the critical habitats surrounding net pens; as others have found with cumulative effects management, the total effects of the combination of each stressor are difficult to predict and are not well understood (Hollarsmith et al., 2022). Section 5 summarizes the risks and uncertainties described throughout the document.

2. Priority habitats and habitat elements

In Washington State, different categories marine habitats and habitat elements are recognized for their unique ecological, economic, and sociocultural value. Several of these habitats and habitat elements have garnered extraordinary state resources to develop and implement policy, regulations, research, conservation, and active restoration to ensure their long-term persistence, and thereby their critical roles in marine ecosystem functionality (see for example (Calloway et al., 2020; Christiaen et al., 2017; Pfister et al., 2018; Puget Sound Partnership, 2018; Washington Marine Resources Advisory Council, 2017). It is of general consensus that CFNPAs have the greatest effect on the environment directly underneath them, and that this effect lessens with distance (Sweetman et al., 2014; Terlizzi et al., 2010; Tsutsumi, 1995; Washington State Department of Ecology, 2022; Weston, 1986; Wildish et al., 2003). Of these habitat and habitat elements, those which may be in the areas *underneath* CFNPA sites include benthic sediments, and those which may be *proximal* to CFNPA sites (owing to siting guidelines) include eelgrass beds, kelp forests, and commercial and recreational shellfish areas. In 2022 the Washington Department of Ecology produced a multi-agency report for the guidance and risk management of commercial net pen aquaculture in Washington. The report *suggests* the deliberate avoidance of sensitive marine habitats when a net pen is sited (Washington State Department of Ecology, 2022). The current document adds detail and context with respect to impacts on certain priority habitats that may be in proximity to a CFNPA facility.

2.1 Eelgrass beds

Seagrasses are ecologically important components of nearshore habitats and support a range of ecosystem functions and services (Duarte & Chiscano, 1999; Hemminga & Duarte, 2000; Orth et al., 2006). For example, seagrasses provide habitat and food web support (Chittaro et al. 2023), sediment stabilization and reductions in water motion and carbon draw-down and sequestration (Fourqurean et al., 2012; Lange et al., 2022; Prentice et al., 2020). These ecosystem services are recognized as being critical in marine habitats around the globe (Orth et al., 2006; Koch et al., 2009; Cullen-Unsworth & Unsworth, 2013; Campagne et al., 2015).

Eelgrass (*Zostera marina*) is a flowering perennial plant, which also grows as an annual, that occurs in nearshore aquatic environments and is the predominant seagrass species in Washington state. Eelgrass serves as important spawning habitats for Pacific herring (*Clupea harengus pallasi*), refugia and migration corridors for juvenile Dungeness crab and salmon (McMillan et al., 1995; Williamson, 2006; Kennedy et al., 2018; Chalifour et al., 2019), and foraging habitat for black brant (*Branta bernicla*) (Wilson & Atkinson, 1995).

In addition to its value as a structural habitat, eelgrass acts as substrate for the growth of epiphytic algae (*Smithora naiadum*), which is an important component in the diets of the numerous nearshore birds, fishes and invertebrates which make up a significant base of the nearshore food chain (Chittaro et al. 2023). Illustrating this value, stable isotope analysis has revealed that eelgrass supports a critical base of the food-chain for some species of rockfish in Puget Sound as they consume species that feed on eelgrass epiphytes and are composed of more than 40% of epiphyte-derived carbon (Chittaro et al. 2023).

Status of Eelgrass in Washington

Global losses of seagrass have been primarily attributed to anthropogenic stressors (Short & Wyllie-Echeverria, 1996; Duarte, 2002; Orth et al., 2006; Waycott et al., 2009; Short et al., 2011) with changes in seagrass density and distribution often indicating a change in the status of ecosystem functionality (Dennison et al., 1993; Short & Burdick, 1996; K.-S. Lee et al., 2004; Orth et al., 2006). Anthropogenic factors known to negatively impact eelgrass in Puget Sound include increased shoreline development and periodic physical disturbances – often leading to degraded water quality (Thom & Hallum, 1990; Dowty et al., 2010; Thom et al., 2011, 2014). Losses of seagrass in Washington State have especially occurred in small embayments and at the heads of shallow bays (Christiaen et al., 2017) contributing to a dynamic spatial distribution over time, even while the overall amount appears stable over the past 40 years (Shelton et al., 2014). The effects of both natural and anthropogenic stressors on eelgrass are expected to in increase with climate change (Lefcheck et al., 2017; Moreno-Marín et al., 2018; Qin et al., 2020).

Given its ecological importance and rapid response to environmental degradation, eelgrass has been identified as a Vital Sign of ecosystem health with a documented Recovery Strategy and identified restoration goals under the Puget Sound Partnership (PSP), the state agency leading the recovery and restoration of Puget Sound (Puget Sound Partnership, 2023). Protecting and restoring submerged Aquatic Vegetation (SAV) including eelgrass and kelp are identified as one of 26 components in the PSP 2022 - 2026 Puget Sound Action Agenda (Puget Sound Partnership, 2022).

2.2 Kelp Forests and other Macroalgae

The brown algae known as kelps (Order Laminariales) are a globally important habitat-forming group of species that occupy 43% of the world's marine ecoregions along coastlines of all continents except Antarctica (Krumhansl et al., 2016). These seaweeds attach with a holdfast to bedrock or cobbles in shallow waters, especially in areas with moderate to high waves or currents. Kelps include species referred to as "floating kelp," those with stipes that grow all the way from their bottom holdfast to the water surface and are kept buoyant by gas filled cysts, and "non-floating" or understory kelp. Kelps have complex life cycles including microscopic stages (spores, gametophytes, gametes), and macroscopic juvenile and adult sporophyte stages (the visible plant stage that is known to most people as "kelp").

Kelp supports the goods and services that humans depend on, such as nursery habitat for socioeconomically important fisheries species, nutrient cycling, and storm defense (Mumford, 2007; Smale et al. 2013, Wernberg et al. 2019). The annual global economic value of kelp habitats is estimated at 500 billion dollars . Kelps are both foundation species and ecosystem engineers in coastal marine systems (Teagle et al., 2017, Werberg et al, 2019). By creating biogenic habitat and exhibiting high rates of primary productivity, kelps are able to alter aspects of the physical environment, such as light, water flow, sedimentation, and pH (Eckman et al., 1989; Krumhansl & Scheibling, 2012; Teagle et al., 2017; Pfister et al., 2019), creating distinct local conditions that ultimately enhance biodiversity, support local food webs, and promote secondary production (Duggins et al., 1989; Krumhansl & Scheibling, 2012; Mann, 1973; Smale et al., 2013; Steneck et al., 2013). Additionally, they support adjacent communities across broader spatial scales via detrital subsidies (Krumhansl & Scheibling, 2012). The production of certain species of pelagic foraging fish are intertwined with kelp communities. In one study, up to 65% of the carbon in black rockfish and 89% of the carbon in kelp greenling were derived from kelp (Von Biela et al., 2016).

Status of Kelp and other Macroalgae in Washington

The distribution and health of kelps is influenced by a range of environmental (e. g. light, temperature, nutrients, substrate) and ecological (e.g. grazing, competition, disease) factors. A world-wide synthesis of kelp forest change by Krumhansl et al. (2016) showed a 38% decline in kelp populations within ecoregions examined with 68% increasing or remaining the same. Any decrease in the cover of kelps or other such functionally critical macroalgae can impact species such as salmonids, herring, and other forage fishes (Shaffer et al., 2020, 2023). In addition to native eelgrass, herring require macroalgae such as red, green, and brown algae as a substrate for spawning (Penttila, 2007). The distribution and health of kelps is influenced by a range of environmental (e. g. light, temperature, nutrients, substrate) and ecological (e.g. grazing, competition, disease) factors.

Kelps have been relied upon as indicators of change because they are high responsivity to changes in environmental conditions (Bell et al., 2015; Wernberg et al., 2013) and are exposed to many human derived forms of coastal disturbance (e.g. pollution, sedimentation, invasive species, fishing, recreation) (Bennett et al., 2016). In Washington State, significant losses of the native floating kelp *(Nereocystis luetkeana*) have been documented throughout Puget Sound (Pfister et al., 2018; H. Berry et al., 2019), with loses primarily occurring in areas with elevated temperature, low nutrients, and low current velocities (Berry et al., 2021). Furthermore, early life history stages of *N. luetkeana* are sensitive to elevated temperatures with populations within the Puget Sound showing a thermal limit for gametophyte growth and sporophyte production between 16-18°C, for which benthic summertime temperatures have recently been recorded (Weigel et al., 2023). Similarly, small *N. leutkeana* and *Saccharina latissima* sporophytes respond negatively to elevated temperatures, showing signs of thermal stress (reduced growth and metabolism) when exposed to short periods of high temperature (21°C). Additionally, it has been suggested that kelps may fare worse at high temperatures when also exposed to high nutrient levels (Fales et al., 2023). Global environmental impacts to kelps are expected to grow along with increasing anthropogenic climate change and its associated stressors (Martínez et al., 2018; Smale, 2020).

In contrast to canopy-forming floating kelp, the distribution of understory macroalgae species is less known. Table 1 lists the depth ranges of several low intertidal and subtidal species whose ranges extend past the 60ft (18m) minimum depth guideline for net pen siting in Washington and may not be encompassed by 'Habitats of Special Significance' (Washington State Department of Ecology, 1986).

Table 1. Depth ranges of an example set of subtidal native aquatic vegetation and habitat-forming invertebrates in the Puget Sound (Lamb & Hanby, 2005)

Species	Category	Depth
Arched red seaweed (*Fryeella gardneri*)	Red seaweed	Subtidal to 25m
Crustose corallines (*Clathomorphum*, *Lithothamnion*, *Melobesia*, *Mesophyllym* genera)	Red seaweed	Intertidal to 30m
Bristly vase sponge (*Leucandra heathi*)	Calcifying sponge	Subtidal to 180m
Sharp lipped tooth sponge (*Rhabdocalyptus dawsoni*)	Siliceous sponge	Subtidal (10m) to >500m
Cloud sponge (*Aphrocallistes vastus*)	Siliceous sponge	Subtidal (10m) to 1600m
Orange finger sponge (*Neoesperiopsis rigida*)	Demosponge	Subtidal (3m) to 30m
Red soft coral (*Gersemia rubiformis*)	Soft coral	Intertidal to >36m
Orange sea pen (*Ptilosarcus gurneyi*)	Sea pen	Subtidal to 135m
Pink candelabrum gorgonian (*Paragorgia pacifica*)	Gorgonian	Subtidal (>15m)
Loose-spiral hydroid (*Hydrallmania distans*)	Hydroid	Intertidal to 45m

2.3 Shellfish habitat and harvest areas

Washington State is home to a wide range of native and introduced bivalve species of immense ecologic, economic, and sociocultural value. Shellfish are critical habitat elements throughout WA state, as they contribute a suite of ecological services including provisioning important biogenic nursery habitat, the cycling of nutrients, filtering of pathogens, maintenance of water quality, carbon sequestration, and shoreline stability (Dumbauld et al., 2009; Theuerkauf et al., 2022; Veggerby et al., 2024). Shellfish species globally are estimated to remove 49,000 tonnes of nitrogen and 6,000 tonnes of phosphorus from aquatic ecosystems each year and have a cumulative annual economic value of 3 to 10 billion dollars worldwide (Van Der Schatte Olivier et al., 2020).

In Washington State, wild harvest and commercial culture of shellfish have a notable economic impact. Wildstock geoduck (*Panopea generosa*) harvest generates over $23 million dollars in state revenue annually; another $3.3 million is generated by the sale of more than 300,000 licenses to harvest clams and oysters each year; and an estimated $27.5 million dollars in revenue is generated through shellfish-related tourism in coastal communities. The shellfish aquaculture industry contributes an estimated $270 million to the state economy, employing over 3200 people in the farming, processing, and sale of shellfish (White Paper: The Washington Shellfish Initiative, 2011). In 2013 it was estimated that Pacific oysters (*Magallana gigas*, formerly *Crassostrea gigas*) made up 38% of the total production weight and total value of shellfish species in Washington State. Manila clams (*Venerupis philippinarum*) followed with 31% and 19%, while geoduck represented 7% and 27% (Washington Sea Grant, 2015). The cultural role of shellfish is significant to indigenous tribes throughout Washington which have harvested wild shellfish, including oysters and clams from Puget Sound and the coastal areas since time immemorial (Belcher, 1985), as well as cultivating and actively managing shellfish in farms or "gardens" (Lepofsky et al., 2015; Matthews & Turner, 2017). Today, these practices are important for maintaining tribal livelihoods and cultural identity (Riccio & Toy, 2016)

Status of Shellfish Habitat and Harvest Areas in Washington

Certain species of shellfish in Washington have seen high losses due to anthropogenic and environmental pressures. Recovery efforts for these species of bivalve are currently in progress through coordinated tribal, state, and non-profit efforts. Washington's only native species of oyster, the Olympia oyster (*Ostrea lurida*), was commercially exploited starting in the 1850s and declined to low numbers by the early 1900s (Dethier, 2006). Point source pollution from pulp mills led to further declines (Dethier, 2006). Further, Olympia oyster recovery has been hampered by several factors, including the removal of dense subtidal shell accumulations, direct competition from exotic species, and the occurrence of introduced shell substrate in the intertidal zone (Trimble et al., 2009). Another threatened native shellfish species, the Pinto abablone (*Haliotis kamtschatkana*), was historically abundant along rocky outcroppings throughout northern Puget Sound, Strait of Juan de Fuca and the San Juan Islands. Unfortunately, and due to overharvest, the species has been reduced to critical numbers and is listed as endangered (Bouma & Hagey, 2014). Climate change and associated ocean acidification, as well as the loss of floating kelp beds, has likely impacted the recovery of this species (Bouma & Hagey, 2014).

2.4 Soft Sediment habitats

Benthic sediments foster a high diversity of macro and micro invertebrates which are critical for the storage and cycling of organic matter and carbon - forming the base of many food chains (Hope et al., 2020). They are home to heterotrophic and chemoautotrophic bacteria that participate in biogeochemical cycles, benthic meiofauna which fill a link between bacterial food sources and larger consumers, and macrofauna such as deposit feeders, burrowers, suspension feeders, and tube builders (Byers & Grabowski, 2014). These different groups create a critical foundation for the larger food chain (Pan & Pratolongo, 2021; Pardo et al., 2023), often providing important forage for salmonids.

Certain species of high cultural and economic value rely on macrofauna that are present in benthic sediments. Dungeness crabs (*Metacarcinus magister*) are designated by the Washington Department of Fish and Wildlife as a priority species. They are opportunistic feeders adapted to foraging in soft sediment habitats. Studies examining the gut content of captured individuals found bivalves to be an important source of prey, along with small benthic fish and shrimp (Butler, 1954; Gotshall, 1977; Stevens et al., 1982). The spot prawn (*Pandalus platyceros*) is a deepwater species that also feeds on polychaetes and small crustaceans that are present in and on top of bottom sediments (Butler, 1980).

Soft sediment and rocky habitats host benthic species such as octocorals, gorgonians, and seapens which serve as habitat and sustenance for other species (Table 1). These species feed by filtering plankton from the water column and can be found at depths ranging from the intertidal zone to the deep sea. These species' distribution and environmental tolerances are not well-studied, particularly for the Puget Sound region; however, their presence is critical to the Salish Sea ecosystem and should be considered during any benthic survey or evaluation of habitat sensitivity.

Status of Soft Sediment Habitats in Washington

Long-term monitoring from sediment stations sampled by the Washington Department of Ecology point to general Puget Sound wide stability within grain size, contaminants (except for HPAH's which are increasing), and benthic invertebrate communities from 1989 – 2015. While at a site level basis, there have been infaunal community shifts largely associated with grain size distribution changes (Partridge et al., 2008).

3. Interaction of CFNPA with habitats and habitat elements

Commercial Finfish Net Pen Aquaculture has the capacity to affect habitats and habitat elements through various interactions – with respect to CFNPA in the Puget Sound, interactions may include effluent, shading, and marine debris and physical damage.

The following section expands on such interactions with habitats and habitat elements of interest to Washington State in terms of information from CFNPA both within the Salish Sea and outside the Salish Sea. It also includes evidence from studies that are not directly topical to CFNPA, but which have additional evidence for how an interaction may impact specific habitats and habitat elements. The habitats and habitat elements discussed below are those outlined in section two: eelgrass and kelp (collectively, submerged aquatic vegetation), shellfish habitat, and soft sediments, as well as others that arose in literature. Habitats and habitat elements are discussed where relevant studies exist with respect to a stressor, however in several cases there have been information gaps identified.

For each interaction, we present (1) an overview of how impacts may result, (2) how current aquaculture practices in the state of Washington aim to prevent, reduce, and respond to impacts, (3) how certain habitats may be affected if impacts do occur, and (4) a summary of factors that may increase risk of impact to habitats.

3.1 Effluent

CFNPA effluent is predominantly organic and arises from fish feed and its' byproducts – e.g., uneaten feed, ingested but undigested feed, and digested feed excreted to the environment (Wang et al., 2013, 2020) in addition to biofouling material falling off the nets and fish carcasses. Another component of effluent arises from the periodic treatment of sea lice infestations – within a CFNPA facility - with pesticide. When needed, farmed salmon are treated with a pesticide solution (sometimes added to their food) which can be circulated into the environment beyond the boundary of the net pen (Hamoutene et al., 2022). Interactions with effluent include its direct effects on water quality (e.g., suspended solids), fecal and chemical pollution, and sediment quality and sea floor degradation directly below farms owing to deposition of carbon, nitrogen, and phosphorous (Boyd et al., 2007; Wang et al., 2020). Effluent has been investigated using parameters in sediment such as total organic carbon (TOC), total organic nitrogen (TON), and loss on ignition (LOI) (Kalantzi & Karakassis, 2006), as well as abundances of heavy metal such as zinc and copper (Hamoutene et al., 2021).

Studies examining the transportation of nutrients from fish farms away from their footprint do not come to a clear consensus, likely due in part to variability in oceanographic conditions and fish biomass among sites. Studies have supported decreased detection of effluent via organic enrichment of the benthos (e.g., TOC, TON) with increasing distance from a CFNPA facility, although that effect is mediated by other factors. For instance, increasing latitude is related to a relatively higher level of benthic enrichment with distance, and sediment type influences the degree to which enrichment changes with distance (Kalantzi & Karakassis, 2006). A study by Navarro et al. (2008) in Scotland found that ammonium and dissolved organic nitrogen concentrations were higher in locations nearest to net pens and coincided with greater heterotrophic microbe abundance and biomass. A similar study by Sanderson et al. (2008) examining the distribution of nutrients for seaweed cultivation around CFNPA facilities in Scotland found no difference in nitrate or phosphate concentrations with distance from net pens, but elevated daytime levels of ammonium were detected for extended periods up to 200 m down current of net pens. Additionally, within 50 m of net pens, water column ammonium concentrations were on average higher than background concentrations and ammonium concentrations throughout small-enclosed bays with fish farms were higher than in similar bays without fish farms (Sanderson et al., 2008). Price et al. (2015) provides a review of the effects of marine cage culture, such as fish farming, on water quality and primary production. Their findings suggest that modern operating conditions and formulated feeds have reduced the environmental impact of fish farms on marine water quality, limiting detectible water column nutrient enrichment within 100 m of farms. However, a study by Sarà (2011) evaluating impacts of offshore aquaculture in the Mediterranean found that nitrogen and phosphorus from CFNPA effluent can be distributed across large spatial scales (bay, gulf, basin level). Additionally, Jansen et al. (2018) demonstrated that increased nutrient concentrations can be detected up to 100 m down current of Norwegian fish farms when fish biomass was high, with the zone of influence occasionally reaching as far or farther than 1000 m. Further, nutrient enhancement depended on biomass as during the first year of operation fish biomass was low and no signs of nutrient enhancement was detected until the farm became more established and fish biomass increased the following year.

Prevention, mitigation, and response

Over the last several decades, reports indicate that CFNPA effluent has been reduced through a series of methods (Rust et al., 2014; Wu, 1995) - for instance, video monitoring of feeding with adjustment of quantity if there is excess observed; monitoring and removing fish carcasses; using vaccines rather than antibiotics, monitoring sediment conditions beneath and proximal to a facility; siting facilities over erosional sediment regimes (C. Price et al., 2015); siting facilities in deeper, high-current areas to improve dispersion to offshore waters (Tsiaras et al., 2022); and cleaning biofouling from nets in an upland area. As described above, this is often reflected by benthic impacts being detectable most predominantly in the area up to 100 m from the boundary of the facility (National Marine Fisheries Service, 2022). Integrated Multi-Trophic Aquaculture (IMTA) has been explored as a method of reducing total effluent via the use of additional organisms – such as sea cucumbers and mussels – that are able to consume the waste products from CFNPA facilities (e.g., C. Price et al., 2015; Wang et al., 2013).

DNR includes several habitat stewardship measures (HSMs) in all CFNPA use-authorization agreements so that lessees may mitigate or reduce sedimentation (e.g., accumulation of effluent particulate in the benthic environment): periodically moving pens and allowing sediment fallowing; siting farms in areas of high current and/or deep water; siting farms in areas of low biological productivity; different orientations and configurations of the farms, i.e. place the long axis perpendicular to prevailing currents (although this may cause the potential of navigation of aesthetic conflicts); and using slow-settling, high digestible feed to maximize food conversions.

Fallow periods between harvesting and restocking are required by NPDES permits (and emphasized by DNR HSMs) for CFNPA facilities in Puget Sound – this minimum 6-week (42-day) period targets a reduction of disease transmission, but also allows the benthos a period of reprieve from effluent for recovery. The 2022 interagency guidance emphasizes the importance of understanding the seasonality of benthic species reproductive processes when considering fallowing length, as the time period may need to be adjusted to allow for resilience of a diverse benthic community (Washington State Department of Ecology, 2022).

In addition to measures that aim to reduce impact from the outset, CFNPA facilities on SOAL must monitor sediment conditions below and surrounding the farm. Baseline surveys required by the DOE under WAC Chapter 173-204 (and associated National Pollution Discharge Elimination System [NPDES] permit) record grain size, TOC, and benthic infauna abundance. TOC is then regularly monitored, and if a certain 'exceedance value' is reached, additional monitoring is required. This additional monitoring includes benthic infauna abundance – and should this monitoring also fail as per the regulating agency's requirements, the facility would begin consultation with DOE to bring the area under compliance standards (see Washington State Department of Ecology, 2021 for all related permits). However, these values alone (TOC and infauna abundance) may not be sufficient to indicate impacts to benthic habitats, pointing to a need for more holistic monitoring – as indicated by Hawkins (2019) and others, infauna community diversity is often most impacted by CFNPA, rather than abundance, which can impact species reliant on this diversity (e.g., as a source of food). Additionally, monitoring infauna community abundance in an ecosystem may not capture the functionality of the ecosystem – for instance, to vertical relief provided by epifaunal invertebrates such as gorgonians and sea pens.

<u>Below, we discuss impacts that could be expected *if* the nature of oceanographic conditions and CFNPA operations led to effluent impacts within a NAV habitat proximal to a CFNPA facility, with the understanding that these impacts may not necessarily occur.</u> We further clarify this discussion by presenting the information by habitat type and by individual components of effluent, to the extent possible.

Impacts to Submerged Aquatic Vegetation

Current guidance on CFNPA in Washington states that facilities must be located at least 300 ft from native aquatic vegetation if that habitat is in less than 75 feet of water. In this case, any impact to NAV would be a result of the transportation of effluent by water current from the facility. US Federal Agencies also assume that the area impacted is within 50-300 m of a net pen facility (National Marine Fisheries Service, 2022). As stated by several review papers and outlined above, it is difficult to detect the entire effluent footprint of a farm because of the influence of current and other temporally variable oceanographic factors (Rust et al., 2014).

Effects from net-pen effluent on seagrasses and seagrass habitat have been documented in several studies. A review on impacts of fish farming facilities to seagrass beds by Pergent-Martini et al. (2006) found that fish farms contributed to a decrease in seagrass meadow surface area and cover, shoot density and size, leaf and rhizome growth and photosynthetic capacity, as well as increases in epiphyte biomass.

Results from research assessing the changes in seagrass meadows exposed to discharge from fish farms show a shift in macrofaunal assemblages towards more of the mollusk, gastropod, amphipod and polychaetae species that are associated with muddy and high organic content sediment and in some cases stressed habitats (Terlizzi et al., 2010). A study in Port Mouton Bay, Nova Scotia Canada assessed differences in environmental variables, *Zostera marina* bed structure and macroinfauna communities at increasing distances from a finfish farm. The results showed nutrient and organic enrichment, higher epiphyte loads, lower eelgrass cover and biomass, and lower macroinfauna biomass, and a change in community structure closer to the farm (Cullain et al., 2018). Ruiz et al. (2010) found changes in a nearshore seagrass meadow community (specifically, epiphyte abundance and herbivore activity) that were linked to effluent from offshore fish farm 3 km away, by comparing the seagrass site with a control and using isotopic analysis to trace nutrients from the farm to the seagrass meadow.

Excessive nutrient input to seagrass beds can also alter their carbon storage capacity. A study by Apostolaki et al. (2011) in the Aegean Sea, Greece found that excess nutrients from fish farms can enhance the amount of carbon, nitrogen, and reduce phosphorus within sediments, shifting the ratio between gross primary productivity and respiration within seagrass sediments below a value of 1. This demonstrated a shift from autotrophy to heterotrophy, indicating a loss in the storage capacity of seagrass beds which jeopardizes their role as carbon sinks.

Effluent component: Suspended Solids

Increased suspended solid load, which in some instances is associated with CFNPA dependent on factors such as water flow and particulate morphology (Schumann & Brinker, 2020), can facilitate an ecosystem shift from large perennial macroalgal species to ephemeral macroalgae such as epiphytic or thin foliose green algae – species which have also been shown to persist in high nutrient conditions (Boyra et al., 2004; Gorgula & Connell, 2004; Karez et al., 2004; Michael Kemp et al., 2004; Watanabe et al., 2016). Waste discharged as particulates into the water column may also reduce water clarity, diminishing light availability which reduces rates of photosynthesis and growth (Cancemi et al., 2003; Cullain et al., 2018; Dolenec et al., 2006). In addition to ephemeral macroalgae being known to succeed in high-sedimentation, low-light-transparency conditions, the survival likelihood of kelp is known to decline, especially for early microscopic life stages (Watanabe et al., 2016).

There are several mechanisms to explain why kelp survival would decrease in these conditions. Particulate discharge from net-pens fine enough to remain suspended in the water column has the potential to reduce water clarity, obstructing light available for all life stages of kelp (Schiel & Foster, 2006). This can influence kelp habitat in the same way shading does, creating conditions that are more favorable for turf algae, that can out compete and displace kelp (Strain et al., 2014). Once turf algae have begun to grow, deposited waste can become trapped therein (similarly to how turf algae is known to trap sediments), creating a nutrient rich layer on the sediment surface that can smother microscopic kelp spores, further preventing reproduction and germination (Filbee-Dexter & Wernberg, 2018; Springer et al., 2010). Further, elevated water turbidity has also been associated with an increase in epiphytes on kelp fronds with heavy epiphyte loads depriving the host plant of sufficient light (Sogn Andersen et al., 2019).

Like kelp, there are multiple studies illustrating how components of effluent can affect eelgrass. Epiphyte biomass (or load) on eelgrass blades increases with increased nutrient concentration in the water (Lee et al., 2004; Lin et al., 1996; McGlathery, 2001; Schmidt et al., 2012; Short, 1987). Studies have consistently demonstrated a negative relationship between epiphyte loading and eelgrass viability (Holmer & Nielsen, 2007; Johnson et al., 2005; Penhale, 1977; Schmidt et al., 2012; Wright et al., 1995).

Impacts to Sediment Communities

Organic effluent, including nitrogen and phosphorus excreted by fish, can accumulate beneath net pen structures (Bannister et al., 2016; Brooks & Mahnken, 2003a); this can lead to heightened oxygen demand by sediments, creating lower-oxygen conditions for benthic communities (Nash, 2001) evidenced by a shift from aerobic to anaerobic metabolism (Holmer & Kristensen, 1992). These conditions usually arise when CFNPA is sited in a location with poor circulation such as within an enclosed bay (Nash, 2001), but studies have found that even when strategically placed in locations with high current, fine particulate organic wastes accumulate beneath net pen structures and become trapped in rugose bottom types (Hall-Spencer et al., 2006; Sweetman et al., 2014). Over time, this accretion can reduce infauna biodiversity both beneath and near net pens resulting in less species rich communities populated by species more tolerant to organic enrichment (Tsutsumi, 1995; Hall-Spencer et al., 2006; Karakassis et al., 1999; Brooks & Mahnken, 2003a).

Particulate and dissolved organic carbon loading can result not only in anaerobic sediment conditions, but also the accumulation of lethal levels of phytotoxic sulfides in the root zones where aquatic vegetation resides (Frederiksen et al., 2007; Holmer & Hasler-Sheetal, 2014). Sediments experiencing long-term exposure to nutrient loading and sulfide accumulation can experience shift in their bacterial communities (Bissett et al., 2007). In certain locations, large white mats of sulfide oxidizing bacteria (Beggiatoa spp.) have been observed beneath net pens (Crawford et al., 2001; DFO, 2021; Findlay & Watling, 1997) and are considered a visual indicator of degraded benthic conditions (Preisler et al., 2007).

Recent research in Norway has examined the impact of CFNPA effluent on sponge and sea pen species both beneath and within several hundred meters of CFNPA facilities. Taormina et al. (2024) found that for the slender sea pen (*Virgularia mirabilis*), animals closer to the facility had higher mucus production, tissue necrosis, and polyp retraction, as well as significantly lower associated fauna. The authors note that the persistence of sea pen populations beneath the 20-year-old CFNPA facility suggests resilience to these impacts. Laroche et al. (2021) considered the response of transplanted *Weberella bursa* sponge to CFNPA facilities (approximately 80m below the farm and at similar depths proximal to the farm) and found only sublethal indicators of response via differences in fatty acid composition, increased gene activity associated with control of cell apoptosis (e.g., cell death), and bacteria associated with nitrogen cycling and biodegradation of compounds found in sea lice treatments. Like Taormina et al., Laroche et al. note that the persistence of *W. bursa* suggests a certain level of resilience despite the CFNPA impacts.

Brooks (2001) evaluated the volatile residue deposition, changes in sediment chemistry, and benthic community response underneath seven commercial net pens in British Columbia. They found that there was a decreased redox potential and increased sulfide concentrations at distances up to 205 meters from any farm (with macrofaunal community changes evident at up to 225 meters down current). In addition to these impacts, benthic infauna diversity declined as sulfide concentrations increased. Large changes in sediment chemistry were found to occur when relatively low numbers of salmon were present. Degraded conditions were found to progress towards habitable sediment conditions for infauna (sulfide levels < 200 µM) when farm sites were not in operation, or fallow. The time required to achieve these conditions was variable and not attained even after 4.5 months of remaining fallow (Brooks, 2001). In another study at Clam Bay (Rich Passage Washington), a group of rare macro benthic infauna species which had vanished with the presence of commercial net pen aquaculture were still absent from farm sediments after 2 years after farming operations had ceased (Mahnken, 1993).

Effluent components

Effluent component: Antibiotics

Antibiotics are milled with fish feed to mitigate disease transfer in CFNPA facilities when deemed necessary; these antibiotics can be transferred to the benthos through uningested feed and particulate excrement, where they can be found in organisms underneath or surrounding the site of impact (Capone et al., 1996; Hargrave, 1949). Different species have shown to accumulate varying levels of antibiotics from net pens. In a study where oysters (*Magallana gigas,* formerly *Crassostrea*) and Red Rock crab (*Cancer productus*) were collected and placed underneath a net pen structure, antibiotic levels were found well above U.S. Food and Drug Administration limits for commercial seafood in crabs, but not oysters (Capone et al., 1996). Sediments sampled next to CFNPA facilities for Atlantic salmon have shown that shifts to more antibiotic resistant microbial communities can occur, and sediments sampled further away (e.g., 8 km from a salmon CFNPA facility) have shown that depending on hydrodynamic conditions and level of antibiotics used, these compounds may travel long distances and have a widespread zone of impact (Buschmann et al., 2012).

Effluent component: Metals and trace-elements

Zinc is present in salmon feeds, and can be transferred to the sediment and water column through uneaten food and feces (Brooks et al., 2002; Brooks & Mahnken, 2003b). While additions of zinc to feed are intended to be bioavailable and thus consumed, in the short term, zinc levels directly underneath salmon net farms have been found at levels that exceed Apparent Effects Thresholds (AETs) (Brooks et al., 2002; Brooks & Mahnken, 2003b). Significant effects to benthic invertebrates (acute, lethal, and sub-lethal) occur at levels that exceed AETs (Puget Sound Estuary Program 1988). Hamoutene et al. (2021) found trace-elements such as copper, zinc, cadmium, molybdenum, and uranium were detected at stations up to 150m from CFNPA sites. Zinc concentrations in particular had an association with distance from CFNPA and was a preferred indicator of benthic enrichment (Hamoutene et al., 2021).

Copper is also a component that is added to salmon feed, but is more often used in antifouling products aimed at reducing the fouling of nets by marine plants and animals (Braithwaite et al., 2007; Brooks & Mahnken, 2003b). If not properly contained, particulate material from these coatings can be transferred to the benthos. Copper levels exceeding lower benchmark thresholds have been found at salmon farms in British Columbia (Brooks & Mahnken, 2003b). Copper is a micronutrient that is toxic (particularly to the larval stage of marine invertebrates) to marine organisms at relatively low levels (Brooks & Mahnken, 2003b). Risk from heavy metals such as Copper and Zinc may be mediated by the concentration of sulfide in the sediment, which reduces the bioavailability of the metals and thus their uptake by sediment biota (Nash, 2001).

Factors associated with increased risk of impact

The 1990 Final Environmental Impact Statement for Fish Culture in Floating Net Pens in Puget Sound describes factors that determine impact of a fish farm on benthic communities: loading, with the number and density of fish leading to the quantity and concentration of organic matter deposited; pen size, with larger (i.e. less concentrated) pens leading to sediments being deposited over a smaller area; water depth and current velocity, with greater dispersion in deeper and faster waters; pen configuration, where orientation to current and relative location of pens can impact dispersion; feed type, with certain feeds having different settling rates and digestibility; and the composition of the benthic community beneath the farm, where more diverse and productive communities may assimilate waste products more readily but also may experience more impacts (Parametrix, 1990). These factors are supported by more recent literature and agency guidance (Hawkins et al., 2019; Washington State Department of Ecology, 2022).

3.2 Shading

Shade cast by overwater structures is deleterious to photosynthetic species and the habitat they create (Lambert et al., 2021; Ono et al., 2010). Docks, piers, rafts, and floating overwater structures - such as those associated with CFNPA - shade submerged aquatic vegetation and can be significant enough to impede the sun's radiant energy from reaching photosynthetic aquatic organisms. This radiant energy, referred to as photosynthetically active radiation (PAR), changes seasonally, with levels typically highest in summer months, and also varies with latitude and time of day (Thom et al., 2008).

Prevention, mitigation, and response

The 2022 inter-agency CFNPA guidance document emphasized the importance of avoiding impacts to submerged aquatic vegetation by avoiding locating farms above those species (Washington State Department of Ecology, 2022). These guidelines are shared in DNR's Habitat Stewardship measures for CFNPA facilities, which require facilities to be in deeper waters where photosynthetic organisms are less common (e.g., 75 ft or greater). We note that some red macroalgae species are known to inhabit these greater than 75-ft depth subtidal areas (Table 1). Kelp and eelgrass species are less likely to occur at greater than 75-ft depths and would only be impacted from overwater structures associated with CFNPA facilities if farms were proposed in shallower nearshore environments.

<u>Below, we discuss impacts that could be expected *if* proposed siting led to shading within a habitat proximal to a CFNPA facility, with the understanding that these impacts would not occur if current Habitat Stewardship Measures and interagency siting guidance are followed.</u>

Impacts to Submerged Aquatic Vegetation

Research from Massachusetts, Florida, and Alabama has shown that docks lead to a reduction in light that can limit the growth and reproduction of seagrasses (Burdick & Short, 1999; Landry et al., 2008; Shafer, 1999). Because of the recognized important ecological role of *Z. marina* in Pacific Northwest marine ecosystems (Duarte & Chiscano, 1999; Hemminga & Duarte, 2000; Orth et al., 2006), there has been a considerable research effort in the region to understand the status and trends of its distribution and how best to sustainably manage and protect the species. While most studies on the effects of shading on submerged aquatic vegetation only refer to 'overwater structures' in the general sense - such as docks, piers, and rafts – CFNPA facilities are often comprised of such structures, and so the effects of dock, pier, or raft on this vegetation can be considered representative of a similar dock, pier, or raft at a CFNPA facility.

A study designed to understand the minimal light needed for eelgrass plants to survive indicated PAR values of at least 3 mol $m^{-2}day^{-1}$, on average, during spring and summer were required (Thom et al., 2008). Washington State Department of Natural Resources quantified and compared light attenuated by overwater structures constructed with deck grating that differed in the percentage of open space. Results showed that all of the floating structures, even those with 70% open space, reduced the amount of light available beneath structures to below the threshold of light required for eelgrass survival (Washington Department of Natural Resources, 2014). Shading is also associated with increased epiphyte biomass on seagrass blades and epiphytes growing on seagrasses - while not parasitic, this growth can block incoming light and reduce photosynthesis of eelgrass (Sand-Jensen, 1977; Brush & Nixon, 2002).

For subtidal kelp, studies have shown that light intensity (e.g., PAR) and water temperature are the two factors primarily responsible for the observed patterns of growth and productivity, when adequate water flow and holdfast substrate is present (Bearham et al., 2013; Lee & Brinkhuis, 1988; Tait, 2019). Minimum required PAR values vary, not only by species, but also by species life stage, with empirical values measured from 0.4 mol $m^{-2}day^{-1}$ for gametophytes (Deysher & Dean, 1986) to greater than 2.0

mol $m^{-2}day^{-1}$ for some juvenile sporophytes (Dean & Jacobsen, 1984; Desmond et al., 2015). Overwater structures can impede the transmission of sufficient light through shading. For instance, in Puget Sound, a study of light transmission beneath overwater structures in topped with 60-70% open space grating found insufficient supply for kelp; even during long, cloudless days, PAR values immediately beneath the structures within 10 cm of the water surface were lower than the 0.4- 2.0 mol $m^{-2}day^{-1}$ range needed for growth and productivity (Gabriel & Donoghue, 2018). If adequate light does reach the sediment bed, juvenile kelp sporophytes may compete for light with fucoids and other macroalgae that require relatively less PAR for survival (Tait, 2019).

Turf algae, a small, fast-growing, opportunistic species with high cover and turnover rates can be highly stress-tolerant compared to larger fleshy macroalgae (Airoldi, 1998). When present, turf algae can prevent the establishment and survival of early-life-stage kelps by quickly overgrowing primary substrate, limiting the availability of hard substrate for kelp-spore settlement (Connell & Russell, 2010; Gorgula & Connell, 2004) and by accumulating sediments which smother other kelp recruits, slowing germination and increasing mortality (Gorman & Connell, 2009; Isæus et al., 2004). In many parts of the world, alterations to the physical environment by humans has mediated the transition from kelp to algal turf dominated systems (Filbee-Dexter & Wernberg, 2018). The macroalgae assemblage shift from kelp forests to these low-structure, mat-like turfs are considered a degradation of habitat with associated losses of biodiversity, food, and productivity (Connell et al., 2014).

Factors leading to increased impact

Impact from shading largely depends upon a facility's overwater structure components being sited in a location directly above or adjacent to submerged aquatic vegetation or other sunlight-dependent species. Certain measures may improve light transmission through overwater structures, such as grating, transparent panels, and orientation of the structure(s) with respect to the sun (Fresh et al., 2006; Lambert et al., 2021).

3.3 Marine Debris and Physical Damage

The operation of CFNPA facilities can produce marine debris, including plastics from sources such as floating collars or handrails; collar floatation; buoys from mooring systems; ropes from mooring systems; net enclosures; predator nets; and feeding systems (pipes and hoppers). Release can occur from storm or high-wave events, poor waste management, installation wear and failure, and/or the closure of a facility without removing all associated debris (Hawkins et al., 2019; Huntington, 2019; Washington State Department of Ecology, 2022).

Prevention, Mitigation, and Response

To reduce the likelihood and extent of impacts from marine debris, the 2022 interagency guidance advises net biofouling management, thoroughly inspecting the seafloor when a net pen is removed or replaced, and marking of gear that may be blown or washing away for greater ease of identification and removal (Washington State Department of Ecology, 2022).

Impacts to Habitats

While little research has captured the cumulative impact of debris from aquaculture in the Puget Sound, there is some evidence on nets - Good et al. (2010) found that of 902 fishing nets recovered from the sound and Northwest Straits during their study, one was attributable to aquaculture. More generally, the different types of plastic that various materials are made from leads to different impacts from their release from the facility. Potential impacts from plastic debris to sensitive habitats include microplastic formation and accumulation, abrasion of habitat, and accumulation of buoyant plastic on beaches (Huntington, 2019). Anchor chains, net pen anchors, and mooring lines - if they are installed and maintained according to best available practices, are thought to have an infrequent minimal area of impact with only minor and localized benthic disturbance (National Marine Fisheries Service, 2022). We note that sonar surveys conducted of past CFNPA facilities in Puget Sound reflect a significant amount of marine debris remaining after facility closures despite initial cleanup efforts – although some debris may be difficult to attribute to a specific CFNPA facility, this suggests the difficulty of following best practices for marine debris reduction and the subsequent increased likelihood of more than minimal impacts to the benthic ecosystems both below and proximal to the farm (Bright, 2023).

Factors Increasing the Risk of Impact

The impact of CFNPA marine debris on benthic communities depends on exposure to storm or high wave events (which may increase in frequency and/or severity with climate change), waste management practices (methods to reduce waste and to recognize release, and to subsequently respond to that release), installation maintenance (to prevent wear and failure), and/or debris survey methods during the closure of a facility (to ensure any associated debris from the structure is removed) (Hawkins et al., 2019; Huntington, 2019; Washington State Department of Ecology, 2022).

4. Cumulative Effects

Section three outlined the individual stressors that appear most frequently in the literature with respect to impacts of CFNPA to habitats and habitat elements. However, stressors can also have an additive or multiplicative impact when combined – these are known as cumulative effects. Cumulative effects are defined as changes caused by the combined impact of different agents or activities. In marine environments, cumulative effects have been classified as additive, synergistic, or antagonistic (Crain et al., 2008). These effects can act towards one another and towards other natural processes, creating a combined impact to the environment that is both different from, and more significant than, an individual impact. Multiple impacts can result in a single stressor, and other times overlap in time and space, producing multiple different stressors that can then interact, creating more complex impacts. As a result, it can be difficult to predict and manage cumulative effects and usually requires reliable baseline environmental information, understanding of ecological processes at play, and estimates of scale.

Understanding the cumulative effects from multiple stressors can be used to proactively increase climate resilience (IPCC, 2022). For a given habitat, habitat element, or species known to be vulnerable to climate change stressors, reducing the stressors that are within the scope of an actor to control can reduce the impacts from anthropogenic climate change that are outside the scope of the actor to control (Alderman & Hobday, 2017).

Cumulative effects resulting from multiple stressors have appeared throughout science and policy for the Salish Sea region. For instance, Calloway et al. (2020) identified multiple stressors affecting kelp in the Puget Sound region and the importance of addressing them holistically: namely, biological stressors such as fishing pressure and the introduction of non-native species in tandem with anthropogenic climate change-associated stressors such as increased water temperature, increased storm severity and associated increased turbidity, and ocean-acidification-associated hypoxia. Given that increased water temperature can exacerbate these other stressors, in order to allow kelp the best chance of persisting through increased water temperatures, these other stressors are recommended to be reduced.

The 2022 multi-agency guidance on CFNPA in Washington advises the consideration of cumulative effects from facility characteristics (e.g., size of farm, presence of nearby farms, dissolved oxygen levels of surrounding habitat) when siting a facility (Washington State Department of Ecology, 2022). As an example, the cumulative effect of diminished flushing and continuous input of nutrients from fecal matter and excess fish food can be increased organic loading and subsequently higher rates of bacterial decomposition. This lowers the dissolved oxygen in the water column and surface sediment and increases sulfate concentrations (Cullain et al. 2018). Such conditions may make the area proximal to net pens uninhabitable for eelgrass; sulfide rich sediments and hypoxic conditions are known to decrease eelgrass photosynthetic activity, biomass, and increase tissue decay (Holmer and Bondgaard 2001). In the province of British Columbia, Canada, impacts to aquatic habitats are assessed through a 'cumulative effects framework' that considers the entire set of stressors – from the past, present, and future – that may impact a habitat's values (Provincial Aquatic Ecosystems Technical Working Group, 2020). The cumulative effects framework assessment for a given area then informs management decisions concerning that area, recognizing that these stressors can act in synergy to impact the habitat.

Climate Change and Commercial Finfish Net Pen Aquaculture

Anthropogenic climate change has the potential to create additive impacts with the previously detailed interactions to the coastal ecosystem. While the ultimate impacts of climate change on CFNPA are unknown and variable by species and region, it is expected that climate change influences important mechanisms such as water levels and currents, eutrophication, stratification, acidification, UV light penetration, runoff, temperature, and weather extremes (Marcogliese, 2008). It is also expected that adjustments to these natural cycles may have increased and additive impacts to the risk of pathogens and parasites (Marcogliese, 2008), eutrophication and harmful algal blooms, and the presence of invasive and non-native species (Collins et al., 2020; Maulu et al., 2021). Taking farmed Atlantic salmon as a specific example (which is a species no longer expected to be farmed in Washington SOAL following Engrossed House Bill 2957), climatic changes are likely to increase the survival and virulence of sea lice, and the regularity of which they are treated with pesticide (Groner et al., 2014; Maulu et al., 2021). While there is strong variability in the conditions productive for Harmful algal blooms, it is possible that they are more prevalent with climate change, elevating concerns related to nutrients and eutrophication surrounding salmon net farms (Collins et al., 2020).

Habitat Connectivity Impacts

In addition to individual habitats and habitat elements, DNR considers how uses on SOAL impact habitat connectivity as a function of that habitat. Habitat connectivity, or ecological spatial connectivity, is the primary process by which nutrients, sediment, and organisms at their various life stages move among spatially distinct habitats, populations, communities, or ecosystems (Carr et al., 2017). In the marine environment, spatial connectivity at an appropriate ecological scale is essential for species survival and ecosystem function.

In the context of CFNPA, to preserve ecosystem productivity and promote resilience, conservation of discrete habitats must be considered through a broad seascape -scale lens (Durrant et al., 2014; Olds et al., 2016). This means that a certain population, population segment, or area – such as the habitats directly impacted by net pen aquaculture – is considered in the broader context of how it affects, or is affected by, linked habitats and populations in different spatial areas. For instance, for organisms such as salmonids that move through different habitats at different life stages, survival often requires successful movement among spatially discontinuous habitats including inhabitable gaps. Creating barriers or blocks within uninhabitable gaps along a movement corridor between habitats may be equally or more destructive to ecosystem function as directly impacting a critical habitat.

Impacts to habitat connectivity can be illustrated through migratory species - CFNPA facilities can attract wild fish (Dempster et al., 2002, 2009; Hawkins et al., 2019; Holmer, 2010), birds (Buschmann et al., 2006, 2009), and marine mammals such as bottlenose dolphins (Harnish et al., 2023), pinnipeds (C. Price et al., 2015), and sharks (Nash et al., 2005) with this attraction largely attributed to the addition of uneaten food to the area as well as the presence of physical structure. Previous work has shown a number of impacts including changes to migration patterns, changes in growth and reproductive success, and altered interspecies trophic interactions for these animals (Callier et al., 2018; Dunlop et al., 2024; Homziak, 1992). Studies have also shown the distribution of commercially valuable demersal fish and crustaceans are impacted by the presence of CFNPAs (Dunlop et al., 2024).

5. Summary of Risks and Uncertainties

The current body of research and evidence demonstrates the potential for stressor-induced risks of CFNPA to the habitats and habitat elements of importance to Washington State Aquatic Lands and the ecosystems throughout them. There are also areas of uncertainty, such as where a stressor is known to have risks to aquatic ecosystems but has not been specifically investigated within the Salish Sea.

5.1 Known risks

In Washington, current-day agency guidelines and management best practices aim to avoid or reduce impacts to habitats by siting farms away from known sensitive areas (e.g., native aquatic vegetation, low dissolved oxygen regions) and in higher-flow, deeper waters; by implementing practices to reduce effluent discharge; and by monitoring benthic conditions to respond to detectable adverse impacts (Washington State Department of Ecology, 2022). In this report, we presented risks examined by the scientific literature and from technical reporting, while noting that guidelines and best practices may serve to prevent, reduce, or mitigate those risks.

Effluent

The effluent from net pens can affect proximal vegetation directly through nutrient loading and sedimentation (Cancemi et al., 2003; Cullain et al., 2018; Pergent-Martini et al., 2006; Ruiz et al., 2010; Terlizzi et al., 2010). Other studies have linked an indirect effect from nutrient promotion of epiphyte biomass on eelgrass viability (Delgado et al., 1996; Holmer & Nielsen, 2007; Johnson et al., 2005; K.-S. Lee et al., 2004; Lin et al., 1996; McGlathery, 2001; Penhale, 1977; Schmidt et al., 2012; Short, 1987; Wright et al., 1995)

Net pen aquaculture effluent also has an observable impact on the soft sediment benthic environment below and proximal to a facility (Boyd et al., 2007; Braithwaite et al., 2007; Brooks, 2001; Brooks et al., 2002; Brooks & Mahnken, 2003a; Buschmann et al., 2012; Capone et al., 1996; Ernst et al., 2001; Hargrave, 1949; Wang et al., 2020; Wu, 1995). This impact increases with reduced water flow and shallow depth; however, it has still been observed for net pens sited in "ideal" locations (Hall-Spencer et al., 2006; Sanderson et al., 2008; Sweetman et al., 2014). Studies show that the benthos below net pens as well as the epifauna and infauna associated with the habitat there can be impacted through direct mortality, chemical or metal accumulation, and compositional shifts due to organic loading, with a common impact of reduced community diversity (Apostolaki et al., 2011; Bannister et al., 2016; Brooks, 2001; Brooks & Mahnken, 2003a; Buschmann et al., 2012; Capone et al., 1996; Crawford et al., 2001; DFO, 2021; Ernst et al., 2001; Findlay & Watling, 1997; Hall-Spencer et al., 2006; Hamoutene et al., 2022; Hargrave, 1949; Johansen et al., 2011; Karakassis et al., 1999; Mahnken, 1993; Sanderson et al., 2008; Sarà et al., 2011; Terlizzi et al., 2010; Tsutsumi, 1995).

Shading

There is a substantial amount of information linking overwater structures and negative impacts on vegetation like eelgrass and macroalgae (Burdick & Short, 1999; Lambert et al., 2021; Ono et al., 2010; Pfister et al., 2018; Reed & Foster, 1984; Shafer, 1999; Szypulski, 2018; Thom et al., 2008). However, there are few studies that directly look at CFNPA facilities and their shading effects, likely owing to best practices avoiding the siting of facilities above or directly near submerged aquatic vegetation. While this represents a gap in our collective knowledge, there are direct similarities between the infrastructure of net pens (walkways, floating docks, netting) and other types of overwater structures which have been observed to impact light availability to photosynthetic organisms. If CFNPA facilities were sited in relatively shallow locations, they could hinder the development of submerged vegetation which would have sequential impacts to other important species, as eelgrass and macroalgae are important structural habitat and sources of food (Hollarsmith et al., 2022; Kennedy et al., 2018; Koenigs et al., 2015; Von Biela et al., 2016).

5.2 Uncertainties

CFNPA in the Salish Sea

While there is a considerable amount of knowledge surrounding the effects of CFNPA and related stressors on habitats and habitat elements from different parts of the world, there is relatively little information on their effects within the Salish Sea. Major knowledge gaps include the effects of primarily effluent, but also shading, marine debris, and hydrodynamics, from CFNPA on sensitive marine habitats within Puget Sound.

Effects of CFNPA and associated interactions may be mitigated with up-to-date practices, including appropriate siting and frequent monitoring (Washington State Department of Ecology, 2022) - however it is difficult to predict their effects on a system lacking prior knowledge and understanding of their effects in that system, and the potential for cumulative effects with other stressors such as climate change.

Shellfish habitat and harvest areas

In Washington State, the primary shellfish species for commercial and recreational harvest shellfish include geoduck, oyster, mussels, and manila clams (Washington Sea Grant, 2015). These species are filter feeders, meaning they pull water through their siphons using their gills to capture particulate food such as phytoplankton and detritus from the water (Arapov et al., 2010). Despite this mechanism, there is minimal information regarding the impact of CFNPA on shellfish.

One study found that Pacific oysters (*Magallana gigas,* formerly *Crassostrea gigas*) growing beneath a net pen did not accumulate an antibiotic at levels harmful to human health (Capone et al., 1996).

However, Increased nutrients released from net pens has been linked to elevated feeding activity as well as accelerated growth rates in shellfish near net pens (Handå et al., 2012; MacDonald et al., 2011). Likewise, laboratory trials have demonstrated shellfish are capable of capturing and assimilating fish feed and feces (Lefebvre et al., 2000; MacDonald et al., 2011), but other research has found shellfish to not assimilate fish farm waste (Sanz-Lazaro & Sanchez-Jerez, 2017). Higher levels of antibiotic resistant bacteria have also been observed in the tissues of blue mussels (*Mytilus edulis*) near fish farms after medication compared to mussel tissue from before medication or from untreated areas (Ervik et al., 1994). However, aquatic environments can harbor antibiotic resistant bacteria from other sources such as wastewater treatment plant and agriculture (Marti et al., 2014). Given the poorly understood relationship between aquaculture and the development of antibiotic-resistant bacteria, as well as its environmental consequences (Done et al., 2015; Miranda et al., 2018; Reverter et al., 2020), it is challenging to identify the sources of antibiotic-resistant bacteria in aquatic environments (Reverter et al., 2020).

Certain pesticides can have lethal effects at relatively long intervals on non–target organisms such as benthic amphipods. They can be acutely lethal to macrofauna found within some distance from the treatment zone. Toxic effects to lobster and shrimp have been found within several hundred meters from a farm, and last for multiple hours (Brooks et al., 2002), while residue from sea lice treatment has been found in sea pens located 170-190m beneath a farm. Due to the longevity of toxic conditions, this effect is likely to be present at large spatial scales where up to hectares of benthos are impacted (Ernst et al., 2001).

Sediment transport

Floating structures, such as net pens, may affect current flow and transport of sediments and farm waste. While examples of altered sediment drift cell currents in the Salish Sea due to floating aquaculture are not readily available, it is possible that certain circumstances of siting (proximity to feeder bluff, density of CFNPAs, and siting in shallow water) could lead to changes in natural sediment deposition through this mechanism. Notably, many studies on hydrodynamic impacts of net pens concern larger arrays than have been installed in the Puget Sound, making inferences challenging. Notably, many studies on hydrodynamic impacts of net pens concern larger arrays than have been installed in the Puget Sound, making inferences challenging. One study in Cherbourg France, using simple and detailed modeling of current velocities showed that fish farming cages can act as barriers to flow causing velocities to increase beneath the cage and for turbulent cells to form behind them (Nguyen, 2018). Likewise, sediments and waste can then get trapped in these cells rather than being transported away from the farm. *In situ* measurements of current and turbidity upstream and downstream of cages confirmed the presence of strong flow velocities below cages and increased turbidity behind them (Nguyen, 2018).

Examples of decreased current velocity surrounding CFNPA come from lab-based experiments (James & O'Donncha, 2019a; Zhao et al., 2015a) and field studies in Norway (Michelsen et al., 2019), where flow reductions due to CFNPA were recorded up to 320 meters downstream. These changes result in site-specific increased particulate deposition and an accumulation of finer material beneath the pens, as demonstrated by Cromey et al. (2002) in a combined modelling, lab, and field-based study in Scotland. Further, when flushing rates are low due to tidal or seasonal shifts in water currents, fine particulate discharge from net pens can remain suspended in the water column and reduce water clarity - obstructing light penetration (Price & Morris, 2013).

Cascading ecosystem effects

While habitat impacts that transcend one trophic level to impact multiple dimensions of an ecosystem are well documented in marine science (Gabara et al., 2021; Piñeiro-Corbeira et al., 2022) this has not been explicitly studied for CFNPA, particularly for the Salish Sea. However, the complex interconnections between habitats and species in this region emphasize the importance of these potential impacts. Important forage fish species such as Pacific herring (*Clupea pallasii*), which use eelgrass and other types of marine vegetation as a substrate for spawning (Hay, 1985) could be impacted by the effects of net pen effluent on marine vegetation if a CFNPA is close to NAV. If the quantity of submerged aquatic vegetation proximal to a net pen either declines or is inhibited from growing, herring would have less substrate to spawn. Documented herring spawning zones are present nearby the locations of several prior CFNPA sites in Washington State (Washington Department of Fish and Wildlife, 2024). Given that herring are prey for other species, such changes can have cascading effects on higher trophic levels within ecosystems (Sigler & Csepp, 2007).

6. References

Alderman, R., & Hobday, A. J. (2017). Developing a climate adaptation strategy for vulnerable seabirds based on prioritization of intervention options. *Deep Sea Research Part II: Topical Studies in Oceanography*, *140*, 290–297. https://doi.org/10.1016/j.dsr2.2016.07.003

Apostolaki, E., Holmer, M., Marbà, N., & Karakassis, I. (2011). Reduced carbon sequestration in a Mediterranean seagrass (*Posidonia oceanica*) ecosystem impacted by fish farming. *Aquaculture Environment Interactions*, *2*(1), 49–59. https://doi.org/10.3354/aei00031

Arapov, J., Ezgeta, D., & Peharda, M. (2010). Bivalve feeding—How and what they eat? *Croatian Journal of Fisheries*, *68*(3), 105–116.

Bannister, R. J., Johnsen, I. A., Hansen, P. K., Kutti, T., & Asplin, L. (2016). Near- and far-field dispersal modelling of organic waste from Atlantic salmon aquaculture in fjord systems. *ICES Journal of Marine Science*, *73*(9), 2408–2419. https://doi.org/10.1093/icesjms/fsw027

Bearham, D., Vanderklift, M., & Gunson, J. (2013). Temperature and light explain spatial variation in growth and productivity of the kelp *Ecklonia radiata*. *Marine Ecology Progress Series*, *476*, 59–70. https://doi.org/10.3354/meps10148

Belcher, W. R. (1985). Shellfish utilization among the Puget sound Salish. *Northwest Anthropological Research Notes*, *19*(1), 83–92.

Bell, T. W., Cavanaugh, K. C., Reed, D. C., & Siegel, D. A. (2015). Geographical variability in the controls of giant kelp biomass dynamics. *Journal of Biogeography*, *42*(10), 2010–2021. https://doi.org/10.1111/jbi.12550

Bennett, S., Wernberg, T., Connell, S. D., Hobday, A. J., Johnson, C. R., & Poloczanska, E. S. (2016). The "Great Southern Reef": Social, ecological and economic value of Australia's neglected kelp forests. *Marine and Freshwater Research*, *67*(1), 47. https://doi.org/10.1071/MF15232

Berry, H., Calloway, M., & Ledbetter, J. (2019). Bull kelp monitoring in south Puget Sound in 2017 and 2018. *Washington Department of Natural Resources*, 72.

Berry, H. D., Mumford, T. F., Christiaen, B., Dowty, P., Calloway, M., Ferrier, L., Grossman, E. E., & VanArendonk, N. R. (2021). Long-term changes in kelp forests in an inner basin of the Salish Sea. *PLOS ONE*, *16*(2), Article 2. https://doi.org/10.1371/journal.pone.0229703

Bissett, A., Burke, C., Cook, P. L. M., & Bowman, J. P. (2007). Bacterial community shifts in organically perturbed sediments. *Environmental Microbiology*, *9*(1), 46–60. https://doi.org/10.1111/j.1462-2920.2006.01110.x

Bouma, J., & Hagey, W. (2014). *Status review report for pinto abalone (*Haliotis kamtschatkana*)* (p. 240). National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Boyd, C. E., Tucker, C., Mcnevin, A., Bostick, K., & Clay, J. (2007). Indicators of Resource Use Efficiency and Environmental Performance in Fish and Crustacean Aquaculture. *Reviews in Fisheries Science*, *15*(4), 327–360. https://doi.org/10.1080/10641260701624177

Boyra, A., Nascimento, F. J. A., Tuya, F., Sanchez-Jerez, P., & Haroun, R. J. (2004). Impact of sea-cage fish farms on intertidal macrobenthic assemblages. *Journal of the Marine Biological Association of the United Kingdom*, *84*(3), 665–668. https://doi.org/10.1017/S0025315404009713h

Braithwaite, R. A., Carrascosa, M. C. C., & McEvoy, L. A. (2007). Biofouling of salmon cage netting and the efficacy of a typical copper-based antifoulant. *Aquaculture*, *262*(2–4), 219–226. https://doi.org/10.1016/j.aquaculture.2006.11.027

Bright, K. (2023). *Hope island sonar survey, target identification, and debris material removal* (p. 17). Cook Aquaculture.

Brooks, K. M. (2001). *An evaluation of the relationship between salmon farm biomass, organic inputs to sediments, physicochemical changes associated with those inputs and the infaunal response – with emphasis on total sediment sulfides, total volatile solids, and oxidation reduction potential* *as surrogate endpoints for biological monitoring*. Aquatic Environmental Sciences. llbc.leg.bc.ca/public/PubDocs/bcdocs/350262/Focused_Study_Final_Report1.pdf

Brooks, K. M., & Mahnken, C. V. W. (2003a). Interactions of Atlantic salmon in the Pacific northwest environment II. Organic wastes. *Fisheries Research*, *62*(3), 255–293. https://doi.org/10.1016/S0165-7836(03)00064-X

Brooks, K. M., & Mahnken, C. V. W. (2003b). Interactions of Atlantic salmon in the Pacific Northwest environment III. Accumulation of zinc and copper. *Fisheries Research*, *62*(3), 295–305. https://doi.org/10.1016/S0165-7836(03)00065-1

Brooks, K. M., Mahnken, C. V. W., & Nash, C. (2002). Environmental effects associated with netpen waste with emphasis on salmon farming in the Pacific Northwest. *Responsible Marine Aquaculture*.

Brush, M., & Nixon, S. (2002). Direct measurements of light attenuation by epiphytes on eelgrass *Zostera marina*. *Marine Ecology Progress Series*, *238*, 73–79. https://doi.org/10.3354/meps238073

Burdick, D. M., & Short, F. T. (1999). The effects of boat docks on eelgrass beds in coastal waters of Massachusetts. *Environmental Management*, *23*(2), Article 2. https://doi.org/10.1007/s002679900182

Buschmann, A. H., Tomova, A., López, A., Maldonado, M. A., Henríquez, L. A., Ivanova, L., Moy, F., Godfrey, H. P., & Cabello, F. C. (2012). Salmon aquaculture and antimicrobial resistance in the marine environment. *PLoS ONE*, *7*(8), e42724. https://doi.org/10.1371/journal.pone.0042724

Butler, T. (1954). *Food of the commercial crab in the Queen Charlotte Island region*.

Butler, T. H. (1980). Shrimps of the Pacific Coast of Canada. *Canadian Bulletin of Fisheries and Aquatic Sciences*, *202*, 280.

Byers, J. E., & Grabowski, J. H. (2014). Soft-sediment communities. In *Marine community ecology and conservation* (pp. 227–249). Sinauer Associates, Inc.

Callier, M. D., Byron, C. J., Bengtson, D. A., Cranford, P. J., Cross, S. F., Focken, U., Jansen, H. M., Kamermans, P., Kiessling, A., Landry, T., O'Beirn, F., Petersson, E., Rheault, R. B., Strand, Ø., Sundell, K., Svåsand, T., Wikfors, G. H., & McKindsey, C. W. (2018). Attraction and repulsion of mobile wild organisms to finfish and shellfish aquaculture: A review. *Reviews in Aquaculture*, *10*(4), 924–949. https://doi.org/10.1111/raq.12208

Calloway, M., Oster, D., Berry, H., Mumford, T., Naar, N., Peabody, B., Hart, L., Tonnes, D., Copps, S., Selleck, J., Allen, B., & Toft, J. (2020). *Puget Sound Kelp Conservation and Recovery Plan* (p. 52). Prepared for NOAA-NMFS. https://nwstraits.org/our-work/kelp/.

Campagne, C. S., Salles, J.-M., Boissery, P., & Deter, J. (2015). The seagrass *Posidonia oceanica*: Ecosystem services identification and economic evaluation of goods and benefits. *Marine Pollution Bulletin*, *97*(1–2), 391–400. https://doi.org/10.1016/j.marpolbul.2015.05.061

Cancemi, G., Falco, G. D., & Pergent, G. (2003). Effects of organic matter input from a fish farming facility on a Posidonia oceanica meadow. *Estuarine, Coastal and Shelf Science*, *56*(5–6), 961–968. https://doi.org/10.1016/S0272-7714(02)00295-0

Capone, D. G., Weston, D. P., Miller, V., & Shoemaker, C. (1996). Antibacterial residues in marine sediments and invertebrates following chemotherapy in aquaculture. *Aquaculture*, *145*(1–4), 55–75. https://doi.org/10.1016/S0044-8486(96)01330-0

Carr, M. H., Robinson, S. P., Wahle, C., Davis, G., Kroll, S., Murray, S., Schumacker, E. J., & Williams, M. (2017). The central importance of ecological spatial connectivity to effective coastal marine protected areas and to meeting the challenges of climate change in the marine environment. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *27*(S1), 6–29. https://doi.org/10.1002/aqc.2800

Chalifour, L., Scott, D., MacDuffee, M., Iacarella, J., Martin, T., & Baum, J. (2019). Habitat use by juvenile salmon, other migratory fish, and resident fish species underscores the importance of estuarine habitat mosaics. *Marine Ecology Progress Series*, *625*, 145–162. https://doi.org/10.3354/meps13064

Chittaro, P., Andrews, K., Tolimieri, N., Gates, J., Buckner, E., Ylitalo, G., & Tonnes, D. (2023). *Dietary connections of marine species to kelp and eelgrass* [Preprint]. In Review. https://doi.org/10.21203/rs.3.rs-2809764/v1

Christiaen, B., Ferrier, L., Dowty, P., Gaeckle, J., & Berry, H. (2017). *Puget Sound seagrass monitoring report: Monitoring year 2015* (p. 61). Nearshore Habitat Team. Department of Natural Resources.

Collins, C., Bresnan, E., Brown, L., Falconer, L., Guilder, J., Jones, L., Kennerley, A., Malham, S., Murray, A., & Stanley, M. (2020). Impacts of climate change on aquaculture [Pdf]. *MCCIP Science Review 2020*, 39 pages. https://doi.org/10.14465/2020.ARC21.AQU

Crain, C. M., Kroeker, K., & Halpern, B. S. (2008). Interactive and cumulative effects of multiple human stressors in marine systems. *Ecology Letters*, *11*(12), 1304–1315. https://doi.org/10.1111/j.1461-0248.2008.01253.x

Crawford, C. M., Mitchell, I. M., & MacLeod, C. (2001). Video assessment of environmental impacts of salmon farms. *ICES Journal of Marine Science*, 445–452.

Cromey, C. J., Nickell, T. D., & Black, K. D. (2002). DEPOMOD—modelling the deposition and biological effects of waste solids from marine cage farms. *Aquaculture*, *214*(1–4), 211–239. https://doi.org/10.1016/S0044-8486(02)00368-X

Cullain et al. - 2018—Potential impacts of finfish aquaculture on eelgra.pdf. (n.d.).

Cullain, N., McIver, R., Schmidt, A. L., Milewski, I., & Lotze, H. K. (2018). Potential impacts of finfish aquaculture on eelgrass (*Zostera marina*) beds and possible monitoring metrics for management: A case study in Atlantic Canada. *PeerJ*, *6*, e5630. https://doi.org/10.7717/peerj.5630

Cullen-Unsworth, L., & Unsworth, R. (2013). Seagrass meadows, ecosystem services, and sustainability. *Environment: Science and Policy for Sustainable Development*, *55*(3), 14–28. https://doi.org/10.1080/00139157.2013.785864

Dean, T. A., & Jacobsen, F. R. (1984). Growth of juvenile *Macrocystis pyrifera* (Laminariales) in relation to environmental factors. *Marine Biology*, *83*(3), 301–311. https://doi.org/10.1007/BF00397463

Delgado, O., Grau, A., Pou, S., Riera, F., Massuti, C., Zabala, M., & Ballesteros, E. (1996). Seagrass regression caused by fish cultures in Fornells Bay (Menorca, Western Mediterranean). *Oceanologica Acta*, *20*(3).

Dempster, T., Sanchez-Jerez, P., Bayle-Sempere, J., Giménez-Casalduero, F., & Valle, C. (2002). Attraction of wild fish to sea-cage fish farms in the south-western Mediterranean Sea: Spatial and short-term temporal variability. *Marine Ecology Progress Series*, *242*, 237–252. https://doi.org/10.3354/meps242237

Dempster, T., Uglem, I., Sanchez-Jerez, P., Fernandez-Jover, D., Bayle-Sempere, J., Nilsen, R., & Bjørn, P. (2009). Coastal salmon farms attract large and persistent aggregations of wild fish: An ecosystem effect. *Marine Ecology Progress Series*, *385*, 1–14. https://doi.org/10.3354/meps08050

Dennison, W. C., Orth, R. J., Moore, K. A., Stevenson, J. C., Carter, V., Kollar, S., Bergstrom, P. W., & Batiuk, R. A. (1993). Assessing water quality with submersed aquatic vegetation. *BioScience*, *43*(2), 86–94. https://doi.org/10.2307/1311969

Desmond, M. J., Pritchard, D. W., & Hepburn, C. D. (2015). Light limitation within Southern New Zealand kelp forest communities. *PLOS ONE*, *10*(4), e0123676. https://doi.org/10.1371/journal.pone.0123676

Dethier, M. N. (2006). *Native shellfish in nearshore ecosystems of Washington State* (No. 2006-04; Puget Sound Nearshore Partnership Report). U.S. Army Corps of Engineers, Seattle District.

Deysher, L. E., & Dean, T. A. (1986). Interactive effects of light and temperature on sporophyte production in the giant kelp *Macrocystis pyrifera*. *Marine Biology*, *93*(1), 17–20. https://doi.org/10.1007/BF00428650

DFO. (2021). *Effects of finfish aquaculture activities on hard seabed ecosystems in British Columbia and advice on monitoring protocols* (Science Advisory Report 2021/038; p. 13). DFO Canadian Science Advisory Secretariat.

Dolenec, T., Lojen, S., Lamba[sbreve]a, [Sbreve]ivana, & Dolenec, M. (2006). Effects of fish farm loading on sea grass *Posidonia oceanica* at Vrgada Island (Central Adriatic): A nitrogen stable isotope study. *Isotopes in Environmental and Health Studies*, *42*(1), 77–85. https://doi.org/10.1080/10256010500384697

Done, H. Y., Venkatesan, A. K., & Halden, R. U. (2015). Does the Recent Growth of Aquaculture Create Antibiotic Resistance Threats Different from those Associated with Land Animal Production in Agriculture? *The AAPS Journal*, *17*(3), 513–524. https://doi.org/10.1208/s12248-015-9722-z

Dowty, P., Berry, H., & Gaeckle, J. (2010). *Developing indicators and targets for eelgrass in Puget Sound: A science assessment* (p. 82). Nearshore Habitat Program, Aquatic Resources Division, Washington Department of Natural Resources.

Duarte, C. M. (2002). The future of seagrass meadows. *Environmental Conservation*, *29*(2), 192–206. https://doi.org/10.1017/S0376892902000127

Duarte, C. M., & Chiscano, C. L. (1999). Seagrass biomass and production: A reassessment. *Aquatic Botany*, *65*(1–4), 159–174. https://doi.org/10.1016/S0304-3770(99)00038-8

Duggins, D. O., Simenstad, C. A., & Estes, J. A. (1989). Magnification of secondary production by kelp detritus in coastal marine ecosystems. *Science*, *245*(4914), 170–173. https://doi.org/10.1126/science.245.4914.170

Dumbauld, B. R., Ruesink, J. L., & Rumrill, S. S. (2009). The ecological role of bivalve shellfish aquaculture in the estuarine environment: A review with application to oyster and clam culture in West Coast (USA) estuaries. *Aquaculture*, *290*(3–4), 196–223. https://doi.org/10.1016/j.aquaculture.2009.02.033

Dunlop, K., Strammer, I., & Keeley, N. (2024). Attraction and avoidance of wild demersal fish and crustaceans to open-net aquaculture pens resolved by baited and towed underwater camera surveys. *Frontiers in Marine Science*, *11*, 1400195. https://doi.org/10.3389/fmars.2024.1400195

Durrant, H. M. S., Burridge, C. P., Kelaher, B. P., Barrett, N. S., Edgar, G. J., & Coleman, M. A. (2014). Implications of macroalgal isolation by distance for networks of marine protected areas: Isolation by distance of macroalgae. *Conservation Biology*, *28*(2), Article 2. https://doi.org/10.1111/cobi.12203

Eckman, J. E., Duggins, D. O., & Sewell, A. T. (1989). Ecology of under story kelp environments. I. Effects of kelps on flow and particle transport near the bottom. *Journal of Experimental Marine Biology and Ecology*, *129*(2), 173–187. https://doi.org/10.1016/0022-0981(89)90055-5

Eger, A. M., Marzinelli, E. M., Beas-Luna, R., Blain, C. O., Blamey, L. K., Byrnes, J. E. K., Carnell, P. E., Choi, C. G., Hessing-Lewis, M., Kim, K. Y., Kumagai, N. H., Lorda, J., Moore, P., Nakamura, Y., Pérez-Matus, A., Pontier, O., Smale, D., Steinberg, P. D., & Vergés, A. (2023). The value of ecosystem services in global marine kelp forests. *Nature Communications*, *14*(1), 1894. https://doi.org/10.1038/s41467-023-37385-0

Ernst, W., Jackman, P., Doe, K., Page, F., Julien, G., MacKay, K., & Sutherland, T. (2001). Dispersion and toxicity to non-target aquatic organisms of pesticides used to treat sea lice on salmon in net pen enclosures. *Marine Pollution Bulletin*, *42*(6), 433–444.

Ervik, A., Thorsen, B., Eriksen, V., Lunestad, B., & Samuelsen, O. (1994). Impact of administering antibacterial agents on wild fish and blue mussels *Mytilus edulis* in the vicinity of fish farms. *Diseases of Aquatic Organisms*, *18*, 45–51. https://doi.org/10.3354/dao018045

Fales, R. J., Weigel, B. L., Carrington, E., Berry, H. D., & Dethier, M. N. (2023). Interactive effects of temperature and nitrogen on the physiology of kelps (*Nereocystis luetkeana* and *Saccharina latissima*). *Frontiers in Marine Science*. https://doi.org/10.3389/fmars.2023.1281104

Filbee-Dexter, K., & Wernberg, T. (2018). Rise of turfs: A new battlefront for globally declining kelp forests. *BioScience*, *68*(2), 64–76. https://doi.org/10.1093/biosci/bix147

Findlay, R., & Watling, L. (1997). Prediction of benthic impact for salmon net-pens based on the balance of benthic oxygen supply and demand. *Marine Ecology Progress Series*, *155*, 147–157. https://doi.org/10.3354/meps155147

Fourqurean, J. W., Duarte, C. M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M. A., Apostolaki, E. T., Kendrick, G. A., Krause-Jensen, D., McGlathery, K. J., & Serrano, O. (2012). Seagrass ecosystems as a globally significant carbon stock. *Nature Geoscience*, *5*(7), Article 7. https://doi.org/10.1038/ngeo1477

Frederiksen, M., Holmer, M., Díaz-Almela, E., Marba, N., & Duarte, C. (2007). Sulfide invasion in the seagrass *Posidonia oceanica* at Mediterranean fish farms: Assessment using stable sulfur isotopes. *Marine Ecology Progress Series*, *345*, 93–104. https://doi.org/10.3354/meps06990

Fresh, K. L., Wyllie-Echeverria, T., Wyllie-Echeverria, S., & Williams, B. W. (2006). Using light-permeable grating to mitigate impacts of residential floats on eelgrass *Zostera marina* L. in Puget Sound, Washington. *Ecological Engineering*, *28*(4), Article 4. https://doi.org/10.1016/j.ecoleng.2006.04.012

Gabara, S. S., Konar, B. H., & Edwards, M. S. (2021). Biodiversity loss leads to reductions in community-wide trophic complexity. *Ecosphere*, *12*(2), 15.

Gabriel, A., & Donoghue, C. (2018). *PAR and light extinction beneath various dock deck types, Pleasant Harbor Marina, WA* [Poster Presentation]. South Sound Science Symposium, Shelton, WA. https://www.dnr.wa.gov/publications/aqr_aamt_phlight_extinction.pdf?zjjbmn

Good, T. P., June, J. A., Etnier, M. A., & Broadhurst, G. (2010). Derelict fishing nets in Puget Sound and the Northwest Straits: Patterns and threats to marine fauna. *Marine Pollution Bulletin*, *60*(1), 39–50. https://doi.org/10.1016/j.marpolbul.2009.09.005

Gorgula, S. K., & Connell, S. D. (2004). Expansive covers of turf-forming algae on human-dominated coast: The relative effects of increasing nutrient and sediment loads. *Marine Biology*, *145*(3). https://doi.org/10.1007/s00227-004-1335-5

Gotshall, D. W. (1977). Stomach contents of northern California Dungeness crabs, *Cancer magister*. *California Fish and Game*, *63*(1), 43–51.

Groner, M. L., Gettinby, G., Stormoen, M., Revie, C. W., & Cox, R. (2014). Modelling the impact of temperature-induced life history plasticity and mate limitation on the epidemic potential of a marine ectoparasite. *PLoS ONE*, *9*(2), e88465. https://doi.org/10.1371/journal.pone.0088465

Hall-Spencer, J., White, N., Gillespie, E., Gillham, K., & Foggo, A. (2006). Impact of fish farms on maerl beds in strongly tidal areas. *Marine Ecology Progress Series*, *326*, 1–9. https://doi.org/10.3354/meps326001

Hamoutene, D., Hua, K., Lacoursière-Roussel, A., Page, F., Baillie, S. M., Brager, L., Salvo, F., Coyle, T., Chernoff, K., Black, M., Wong, D., Nelson, E., Bungay, T., Gaspard, D., Ryall, E., Mckindsey, C. W., & Sutherland, T. F. (2021). Assessing trace-elements as indicators of marine finfish aquaculture across three distinct Canadian coastal regions. *Marine Pollution Bulletin*, *169*, 112557. https://doi.org/10.1016/j.marpolbul.2021.112557

Hamoutene, D., Oldford, V., & Donnet, S. (2022). Drug and pesticide usage for sea lice treatment in salmon aquaculture sites in a Canadian province from 2016 to 2019. *Scientific Reports*, *12*(1), 4475. https://doi.org/10.1038/s41598-022-08538-w

Handå, A., Min, H., Wang, X., Broch, O. J., Reitan, K. I., Reinertsen, H., & Olsen, Y. (2012). Incorporation of fish feed and growth of blue mussels (*Mytilus edulis*) in close proximity to salmon (*Salmo salar*) aquaculture: Implications for integrated multi-trophic aquaculture in Norwegian coastal waters. *Aquaculture*, *356–357*, 328–341. https://doi.org/10.1016/j.aquaculture.2012.04.048

Hargrave, B. T. (Ed.). (1949). *Modelling benthic impacts of organic enrichment from marine aquaculture: Vol. Canadian Technical Report of Fisheries and Aquatic Sciences 1949*.

Harnish, A. E., Baird, R. W., Corsi, E., Gorgone, A. M., Perrine, D., Franco, A., Hankins, C., & Sepeta, E. (2023). Long-term associations of common bottlenose dolphins with a fish farm in Hawaiʻi and impacts on other protected species. *Marine Mammal Science*, *39*(3), 794–810. https://doi.org/10.1111/mms.13010

Hawkins, J. L., Bath, G. B., Dickhoff, W. W., & Morris, J. A. (2019). *State of science on net pen aquaculture in Puget Sound Washington* (p. 219). Report to National Oceanic and Atmospheric Administration.

Hay, D. E. (1985). Reproductive biology of Pacific Herring (*Clupea harengus pallasi*). *Canadian Journal of Fisheries and Aquatic Sciences*, *42*(S1), s111–s126. https://doi.org/10.1139/f85-267

Hemminga, M. A., & Duarte, C. M. (2000). *Seagrass ecology*. Cambridge University Press.

Hollarsmith, J. A., Andrews, K., Naar, N., Starko, S., Calloway, M., Obaza, A., Buckner, E., Tonnes, D., Selleck, J., & Therriault, T. W. (2022). Toward a conceptual framework for managing and conserving marine habitats: A case study of kelp forests in the Salish Sea. *Ecology and Evolution*, *12*(1), Article 1. https://doi.org/10.1002/ece3.8510

Holmer, M. (2010). Environmental issues of fish farming in offshore waters: Perspectives, concerns and research needs. *Aquaculture Environment Interactions*, *1*(1), 57–70. https://doi.org/10.3354/aei00007

Holmer, M., & Bondgaard, E. J. (2001). Photosynthetic and growth response of eelgrass to low oxygen and high sulfide concentrations during hypoxic events. *Aquatic Botany*, *70*(1), 29–38. https://doi.org/10.1016/S0304-3770(00)00142-X

Holmer, M., & Hasler-Sheetal, H. (2014). Sulfide intrusion in seagrasses assessed by stable sulfur isotopes -a synthesis of current results. *Frontiers in Marine Science*, *1*. https://doi.org/10.3389/fmars.2014.00064

Holmer, M., & Kristensen, E. (1992). Impact of marine fish cage farming on metabolism and sulfate reduction of underlying sediments. *Marine Ecology Progress Series*, *80*, 191–201. https://doi.org/10.3354/meps080191

Holmer, M., & Nielsen, R. M. (2007). Effects of filamentous algal mats on sulfide invasion in eelgrass (*Zostera marina*). *Journal of Experimental Marine Biology and Ecology*, *353*(2), 245–252. https://doi.org/10.1016/j.jembe.2007.09.010

Homziak, J. (1992). *Environmental guidelines for siting and monitoring net-pen aquaculture operations in the Northern Gulf of Mexico*. *13*, 733–742.

Hope, J. A., Paterson, D. M., & Thrush, S. F. (2020). The role of microphytobenthos in soft-sediment ecological networks and their contribution to the delivery of multiple ecosystem services. *Journal of Ecology*, *108*(3), 815–830. https://doi.org/10.1111/1365-2745.13322

Huntington, T. (2019). *Marine litter and aquaculture gear – White Paper.* (1539-ASC/R/01/C; p. 20). Aquaculture Stewardship Council.

IPCC. (2022). *Climate change 2022 – Impacts, adaptation and vulnerability: Working group II contribution to the sixth assessment report of the intergovernmental panel on climate change* (1st ed.). Cambridge University Press. https://doi.org/10.1017/9781009325844

James, S. C., & O'Donncha, F. (2019a). Drag coefficient parameter estimation for aquaculture systems. *Environmental Fluid Mechanics*, *19*(4), 989–1003. https://doi.org/10.1007/s10652-019-09697-7

James, S. C., & O'Donncha, F. (2019b). Drag coefficient parameter estimation for aquaculture systems. *Environmental Fluid Mechanics*, *19*(4), 989–1003. https://doi.org/10.1007/s10652-019-09697-7

Johansen, L.-H., Jensen, I., Mikkelsen, H., Bjørn, P.-A., Jansen, P. A., & Bergh, Ø. (2011). Disease interaction and pathogens exchange between wild and farmed fish populations with special reference to Norway. *Aquaculture*, *315*(3–4), 167–186. https://doi.org/10.1016/j.aquaculture.2011.02.014

Johnson, M. P., Edwards, M., Bunker, F., & Maggs, C. A. (2005). Algal epiphytes of *Zostera marina*: Variation in assemblage structure from individual leaves to regional scale. *Aquatic Botany*, *82*(1), 12–26. https://doi.org/10.1016/j.aquabot.2005.02.003

Kalantzi, I., & Karakassis, I. (2006). Benthic impacts of fish farming: Meta-analysis of community and geochemical data. *Marine Pollution Bulletin*, *52*(5), 484–493. https://doi.org/10.1016/j.marpolbul.2005.09.034

Karakassis, I., Hatziyanni, E., Tsapakis, M., & Plaiti, W. (1999). Benthic recovery following cessation of fish farming:a series of successes and catastrophes. *Marine Ecology Progress Series*, *184*, 205–218. https://doi.org/10.3354/meps184205

Karez, R., Engelbert, S., Kraufvelin, P., Pedersen, M. F., & Sommer, U. (2004). Biomass response and changes in composition of ephemeral macroalgal assemblages along an experimental gradient of nutrient enrichment. *Aquatic Botany*, *78*(2), 103–117. https://doi.org/10.1016/j.aquabot.2003.09.008

Kennedy, L. A., Juanes, F., & El-Sabaawi, R. (2018). Eelgrass as valuable nearshore foraging habitat for juvenile Pacific salmon in the early marine period. *Marine and Coastal Fisheries*, *10*(2), 190–203. https://doi.org/10.1002/mcf2.10018

Koch, E. W., Barbier, E. B., Silliman, B. R., Reed, D. J., Perillo, G. M., Hacker, S. D., Granek, E. F., Primavera, J. H., Muthiga, N., Polasky, S., Halpern, B. S., Kennedy, C. J., Kappel, C. V., & Wolanski, E. (2009). Non-linearity in ecosystem services: Temporal and spatial variability in coastal protection. *Frontiers in Ecology and the Environment*, *7*(1), 29–37. https://doi.org/10.1890/080126

Koenigs, C., Miller, R., & Page, H. (2015). Top predators rely on carbon derived from giant kelp Macrocystis pyrifera. *Marine Ecology Progress Series*, *537*, 1–8. https://doi.org/10.3354/meps11467

Krumhansl, K. A., Okamoto, D. K., Rassweiler, A., Novak, M., Bolton, J. J., Cavanaugh, K. C., Connell, S. D., Johnson, C. R., Konar, B., Ling, S. D., Micheli, F., Norderhaug, K. M., Pérez-Matus, A., Sousa-Pinto, I., Reed, D. C., Salomon, A. K., Shears, N. T., Wernberg, T., Anderson, R. J., … Byrnes, J. E. K. (2016). Global patterns of kelp forest change over the past half-century. *Proceedings of the National Academy of Sciences*, *113*(48), 13785–13790. https://doi.org/10.1073/pnas.1606102113

Krumhansl, K. A., & Scheibling, R. (2012). Production and fate of kelp detritus. *Marine Ecology Progress Series*, *467*, 281–302. https://doi.org/10.3354/meps09940

Lamb, A., & Hanby, B. P. (2005). *Marine life of the Pacific Northwest: A photographic encyclopedia of invertebrates, seaweeds and selected fishes*. Harbour Pub.

Lambert, M. R., Ojala-Barbour, R., Jr, R. V., McIntyre, A. P., & Quinn, T. (2021). *Small overwater structures: A review of effects on Puget Sound habitat and salmon* (p. 31). Washington Department of Fish and Wildlife.

Landry, J. B., Kenworthy, W. J., & Carlo, G. D. (2008). *The effects of docks on seagrasses, with particular emphasis on the threatened seagrass,* Halophila johnsonii (p. 32). NOAA/Center for Coastal Fisheries and Habitat Research.

Lange, T., Oncken, N., Svane, N., Steinfurth, R., Kristensen, E., & Flindt, M. (2022). Large-scale eelgrass transplantation: A measure for carbon and nutrient sequestration in estuaries. *Marine Ecology Progress Series*, *685*, 97–109. https://doi.org/10.3354/meps13975

Lee, J. A., & Brinkhuis, B. H. (1988). Seasonal light and temperature interaction effects on development of *Laminaria Saccharina* (Phaeophyta) Gametophytes and juvenile sporophytes. *Journal of Phycology*, *24*(2), 181–191. https://doi.org/10.1111/j.1529-8817.1988.tb00076.x

Lee, K.-S., Short, F. T., & Burdick, D. M. (2004). Development of a nutrient pollution indicator using the seagrass, *Zostera marina*, along nutrient gradients in three New England estuaries. *Aquatic Botany*, *78*(3), 197–216. https://doi.org/10.1016/j.aquabot.2003.09.010

Lefcheck, J. S., Wilcox, D. J., Murphy, R. R., Marion, S. R., & Orth, R. J. (2017). Multiple stressors threaten the imperiled coastal foundation species eelgrass (*Zostera marina*) in Chesapeake Bay, USA. *Global Change Biology*, *23*(9), 3474–3483. https://doi.org/10.1111/gcb.13623

Lefebvre, S., Barillé, L., & Clerc, M. (2000). Pacific oyster (*Crassostrea gigas*) feeding responses to a fish-farm effluent. *Aquaculture*, *187*(1), 185–198. https://doi.org/10.1016/S0044-8486(99)00390-7

Lepofsky, D., Smith, N. F., Cardinal, N., Harper, J., Morris, M., (Elroy White), G., Bouchard, R., Kennedy, D. I. D., Salomon, A. K., Puckett, M., & Rowell, K. (2015). Ancient shellfish mariculture on the Northwest coast of North America. *American Antiquity*, *80*(2), 236–259. https://doi.org/10.7183/0002-7316.80.2.236

Lin, H.-J., Nixon, S. W., Taylor, D. I., Granger, S. L., & Buckley, B. A. (1996). Responses of epiphytes on eelgrass, Zostera marina L., to separate and combined nitrogen and phosphorus enrichment. *Aquatic Botany*, *52*(4), 243–258. https://doi.org/10.1016/0304-3770(95)00503-X

MacDonald, B. A., Robinson, S. M. C., & Barrington, K. A. (2011). Feeding activity of mussels (*Mytilus edulis*) held in the field at an integrated multi-trophic aquaculture (IMTA) site (*Salmo salar*) and exposed to fish food in the laboratory. *Aquaculture*, *314*(1–4), 244–251. https://doi.org/10.1016/j.aquaculture.2011.01.045

Mahnken, C. V. W. (1993). *Benthic faunal recovery and succession after removal of a marine fish farm: A dissertation submitted in partial fulfillment of the requirements for the degree of Doctor of Philosophy*. University of Washington.

Mann, K. H. (1973). Seaweeds: Their productivity and strategy for growth: The role of large marine algae in coastal productivity is far more important than has been suspected. *Science*, *182*(4116), 975–981. https://doi.org/10.1126/science.182.4116.975

Marcogliese, D. (2008). The impact of climate change on the parasites and infectious diseases of aquatic animals. *Revue Scientifique et Technique (International Office of Epizootics)*, *27*, 467–484.

Marti, E., Variatza, E., & Balcazar, J. L. (2014). The role of aquatic ecosystems as reservoirs of antibiotic resistance. *Trends in Microbiology*, *22*(1), 36–41. https://doi.org/10.1016/j.tim.2013.11.001

Martínez, B., Radford, B., Thomsen, M. S., Connell, S. D., Carreño, F., Bradshaw, C. J. A., Fordham, D. A., Russell, B. D., Gurgel, C. F. D., & Wernberg, T. (2018). Distribution models predict large contractions of habitat-forming seaweeds in response to ocean warming. *Diversity and Distributions*, *24*(10), 1350–1366. https://doi.org/10.1111/ddi.12767

Matthews, D. L., & Turner, N. J. (2017). Ocean cultures: Northwest Coast ecosystems and Indigenous management systems. In *Conservation for the Anthropocene Ocean: Interdisciplinary Science in Support of Nature and People* (pp. 166–199). Academic Press.

Maulu, S., Hasimuna, O. J., Haambiya, L. H., Monde, C., Musuka, C. G., Makorwa, T. H., Munganga, B. P., Phiri, K. J., & Nsekanabo, J. D. (2021). Climate change effects on aquaculture production:

Sustainability implications, mitigation, and adaptations. *Frontiers in Sustainable Food Systems*, *5*, 609097. https://doi.org/10.3389/fsufs.2021.609097

McGlathery, K. J. (2001). Macroalgal blooms contribute to the decline of seagrass in nutrient-enriched coastal waters. *Journal of Phycology*, *37*(4), 453–456.

McMillan, R. O., Armstrong, D. A., & Dinnel, P. A. (1995). Comparison of intertidal habitat use and growth rates of two Northern Puget Sound cohorts of 0+ age Dungeness crab, *Cancer magister*. *Estuaries*, *18*(2), 390. https://doi.org/10.2307/1352321

Michael Kemp, W., Batleson, R., Bergstrom, P., Carter, V., Gallegos, C. L., Hunley, W., Karrh, L., Koch, E. W., Landwehr, J. M., Moore, K. A., Murray, L., Naylor, M., Rybicki, N. B., Court Stevenson, J., & Wilcox, D. J. (2004). Habitat requirements for submerged aquatic vegetation in Chesapeake Bay: Water quality, light regime, and physical-chemical factors. *Estuaries*, *27*(3), Article 3. https://doi.org/10.1007/BF02803529

Michelsen, F. A., Klebert, P., Broch, O. J., & Alver, M. O. (2019). Impacts of fish farm structures with biomass on water currents: A case study from Frøya. *Journal of Sea Research*, *154*, 101806. https://doi.org/10.1016/j.seares.2019.101806

Miranda, C. D., Godoy, F. A., & Lee, M. R. (2018). Current status of the use of antibiotics and the antimicrobial resistance in the Chilean salmon farms. *Frontiers in Microbiology*, *9*, 1284. https://doi.org/10.3389/fmicb.2018.01284

Moreno-Marín, F., Brun, F. G., & Pedersen, M. F. (2018). Additive response to multiple environmental stressors in the seagrass *Zostera marina* L. *Limnology and Oceanography*, *63*(4), 1528–1544. https://doi.org/10.1002/lno.10789

Nash, C. E. (2001). The net-pen salmon farming industry in the Pacific Northwest. *U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-49*, 125.

Nash, C. E., Burbridge, P. R., & Volkman, J. K. (2005). Guidelines for ecological risk assessment of marine fish aquaculture. *U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-49*, 90.

Nash, C. E., & Waknitz, F. W. (2003). Interactions of Atlantic salmon in the Pacific Northwest. *Fisheries Research*, *62*(3), 237–254. https://doi.org/10.1016/S0165-7836(03)00063-8

National Marine Fisheries Service. (2022). *Reinitiation of Endangered Species Act Section 7(a)(2) Biological Opinion, and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response for the Environmental Protection Agency's Approval of Washington State Department of Ecology's Sediment Management Standards (WAC 173-204-412) Regarding Marine Finfish Rearing Facilities* (NMFS No: WCRO-2018-00286; p. 219). NOAA/National Marine Fisheries Service.

Nguyen, T. H. Y. (2018). *Modeling of flow and sediment transport in the vicinity of immersed structures: Application to aquaculture cages. Fluids mechanics [physics.class-ph].* Normandie Université.

Olds, A. D., Connolly, R. M., Pitt, K. A., Pittman, S. J., Maxwell, P. S., Huijbers, C. M., Moore, B. R., Albert, S., Rissik, D., Babcock, R. C., & Schlacher, T. A. (2016). Quantifying the conservation value of seascape connectivity: A global synthesis. *Global Ecology and Biogeography*, *25*(1), 3–15. https://doi.org/10.1111/geb.12388

Ono, K., Simenstad, C. A., Toft, J. D., Southard, S. L., Sobocinski, K. L., & Borde, A. (2010). *Assessing and mitigating dock shading impacts on the behavior of juvenile Pacific salmon (*Oncorhynchus *spp.): Can artificial light mitigate the effects?* (WA-RD 755.1; p. 94). Washington State Transport Center.

Orth, R. J., Carruthers, T. J. B., Dennison, W. C., Duarte, C. M., Fourqurean, J. W., Heck, K. L., Hughes, A. R., Kendrick, G. A., Kenworthy, W. J., Olyarnik, S., Short, F. T., Waycott, M., & Williams, S. L. (2006). A global crisis for seagrass ecosystems. *BioScience*, *56*(12), 987–996. https://doi.org/10.1641/0006-3568(2006)56[987:AGCFSE]2.0.CO;2

Pan, J., & Pratolongo, D. (2021). Soft-bottom marine benthos. In J. Pan & P. D. Pratolongo, *Marine biology a functional approach to the oceans and their organisms* (1st ed., pp. 180–210). CRC Press. https://doi.org/10.1201/9780429399244-10

Parametrix. (1990). *Final programmatic environmental impact statement fish culture in floating net pens* (p. 203). Washington Department of Fisheries. https://fortress.wa.gov/ecy/publications/documents/1206019.pdf

Pardo, J. C. F., Poste, A. E., Frigstad, H., Quintana, C. O., & Trannum, H. C. (2023). The interplay between terrestrial organic matter and benthic macrofauna: Framework, synthesis, and perspectives. *Ecosphere*, *14*(5), e4492. https://doi.org/10.1002/ecs2.4492

Partridge, V., Burgess, D., Weakland, S., Dutch, M., & Eagleston, A. (2008). *Sediment quality in Puget Sound changes in chemical contaminants and invertebrate communities at 10 sentinel stations, 1989–2015* (18-03–005; p. 63). Washington Department of Ecology.

Penhale, P. A. (1977). Macrophyte-epiphyte biomass and productivity in an eelgrass (*Zostera marina* L.) community. *Journal of Experimental Marine Biology and Ecology*, *26*(2), 211–224. https://doi.org/10.1016/0022-0981(77)90109-5

Penttila, D. (2007). *Marine forage fishes in Puget Sound:* (p. 30). Defense Technical Information Center. https://doi.org/10.21236/ADA478081

Pergent-Martini, C., Boudouresque, C., Pasqualini, V., & Pergent, G. (2006). Impact of fish farming facilities on *Posidonia oceanica* meadows: A review. *Marine Ecology*, *27*(4), 310–319. https://doi.org/10.1111/j.1439-0485.2006.00122.x

Pfister, C. A., Altabet, M. A., & Weigel, B. L. (2019). Kelp beds and their local effects on seawater chemistry, productivity, and microbial communities. *Ecology*, *100*(10), e02798. https://doi.org/10.1002/ecy.2798

Pfister, C. A., Berry, H. D., & Mumford, T. (2018). The dynamics of kelp forests in the Northeast Pacific Ocean and the relationship with environmental drivers. *Journal of Ecology*, *106*(4), 1520–1533. https://doi.org/10.1111/1365-2745.12908

Piñeiro-Corbeira, C., Barrientos, S., Barreiro, R., & De La Cruz-Modino, R. (2022). Assessing the importance of kelp forests for small-scale fisheries under a global change scenario. *Frontiers in Marine Science*, *9*, 973251. https://doi.org/10.3389/fmars.2022.973251

Preisler, A., De Beer, D., Lichtschlag, A., Lavik, G., Boetius, A., & Jørgensen, B. B. (2007). Biological and chemical sulfide oxidation in a *Beggiatoa* inhabited marine sediment. *The ISME Journal*, *1*(4), 341–353. https://doi.org/10.1038/ismej.2007.50

Prentice, C., Poppe, K. L., Lutz, M., Murray, E., Stephens, T. A., Spooner, A., Hessing-Lewis, M., Sanders-Smith, R., Rybczyk, J. M., Apple, J., Short, F. T., Gaeckle, J., Helms, A., Mattson, C., Raymond, W. W., & Klinger, T. (2020). A synthesis of blue carbon stocks, sources, and accumulation rates in eelgrass (*Zostera marina*) meadows in the Northeast Pacific. *Global Biogeochemical Cycles*, *34*(2), e2019GB006345. https://doi.org/10.1029/2019GB006345

Price, C., Black, K., Hargrave, B., & Morris, J. (2015). Marine cage culture and the environment: Effects on water quality and primary production. *Aquaculture Environment Interactions*, *6*(2), 151–174. https://doi.org/10.3354/aei00122

Price, C. S., & Morris, J. A. (2013). *Marine cage culture and the environment: Twenty-first century science informing a sustainable industry* (Technical Memorandum NOS NCCOS 164; p. 158). NOAA.

Provincial Aquatic Ecosystems Technical Working Group. (2020, December). *Interim assessment protocol for aquatic ecosystems in British Columbia – Standards for assessing the condition of aquatic ecosystems under British Columbia's cumulative effects framework. Version 1.3*. Ministry of Environment and Climate Change Strategy and Ministry of Forests, Lands and Natural Resource Operations and Rural Development.

Puget Sound Partnership. (2018). *The 2018-2022 action agenda for Puget Sound*. Puget Sound Partnership.

Puget Sound Partnership. (2022). *2022-2026 action agenda for Puget Sound*. https://www.psp.wa.gov/2022AAupdate.php

Puget Sound Partnership. (2023). *2023 State of the Sound*.

Qin, L.-Z., Kim, S. H., Song, H.-J., Suonan, Z., Kim, H., Kwon, O., & Lee, K.-S. (2020). Influence of regional water temperature variability on the flowering phenology and sexual reproduction of the seagrass *Zostera marina* in Korean coastal waters. *Estuaries & Coasts*, *43*(3), 449–462. https://doi.org/10.1007/s12237-019-00569-3

Reed, D. C., & Foster, M. S. (1984). The effects of canopy shadings on algal recruitment and growth in a giant kelp forest. *Ecology*, *65*(3), Article 3. https://doi.org/10.2307/1938066

Reverter, M., Sarter, S., Caruso, D., Avarre, J.-C., Combe, M., Pepey, E., Pouyaud, L., Vega-Heredía, S., De Verdal, H., & Gozlan, R. E. (2020). Aquaculture at the crossroads of global warming and antimicrobial resistance. *Nature Communications*, *11*(1), 1870. https://doi.org/10.1038/s41467-020-15735-6

Riccio, R. W., & Toy, K. (2016). *Maintaining livelihoods and cultural identity through shellfish aquaculture*. Salish Sea Ecosystem Conference, Vancouver, B.C.

Ruiz, J. M., Marco-Méndez, C., & Sánchez-Lizaso, J. L. (2010). Remote influence of off-shore fish farm waste on Mediterranean seagrass (*Posidonia oceanica*) meadows. *Marine Environmental Research*, *69*(3), 118–126. https://doi.org/10.1016/j.marenvres.2009.09.002

Rust, M. B., Amos, K. H., Bagwill, A. L., Dickhoff, W. W., Juarez, L. M., Price, C. S., Morris, J. A., & Rubino, M. C. (2014). Environmental performance of marine net-pen aquaculture in the United States. *Fisheries*, *11*, 508–524.

Sanderson, J. C., Cromey, C. J., Dring, M. J., & Kelly, M. S. (2008). Distribution of nutrients for seaweed cultivation around salmon cages at farm sites in north–west Scotland. *Aquaculture*, *278*(1–4), 60–68. https://doi.org/10.1016/j.aquaculture.2008.03.027

Sand-Jensen, K. (1977). Effect of epiphytes on eelgrass photosynthesis. *Aquatic Botany*, *3*, 55–63. https://doi.org/10.1016/0304-3770(77)90004-3

Sanz-Lazaro, C., & Sanchez-Jerez, P. (2017). Mussels do not directly assimilate fish farm wastes: Shifting the rationale of integrated multi-trophic aquaculture to a broader scale. *Journal of Environmental Management*, *201*, 82–88. https://doi.org/10.1016/j.jenvman.2017.06.029

Sarà, G., Lo Martire, M., Sanfilippo, M., Pulicanò, G., Cortese, G., Mazzola, A., Manganaro, A., & Pusceddu, A. (2011). Impacts of marine aquaculture at large spatial scales: Evidences from N and P catchment loading and phytoplankton biomass. *Marine Environmental Research*, *71*(5), 317–324. https://doi.org/10.1016/j.marenvres.2011.02.007

Schiel, D. R., & Foster, M. S. (2006). The population biology of large brown seaweeds: Ecological consequences of multiphase life histories in dynamic coastal environments. *Annual Review of Ecology, Evolution, and Systematics*, *37*(1), 343–372. https://doi.org/10.1146/annurev.ecolsys.37.091305.110251

Schmidt, A. L., Wysmyk, J. K. C., Craig, S. E., & Lotze, H. K. (2012). Regional-scale effects of eutrophication on ecosystem structure and services of seagrass beds. *Limnology and Oceanography*, *57*(5), 1389–1402. https://doi.org/10.4319/lo.2012.57.5.1389

Schumann, M., & Brinker, A. (2020). Understanding and managing suspended solids in intensive salmonid aquaculture: A review. *Reviews in Aquaculture*, *12*(4), 2109–2139. https://doi.org/10.1111/raq.12425

Shafer, D. J. (1999). The Effects of dock shading on the seagrass *Halodule wrightii* in Perdido Bay, Alabama. *Estuaries*, *22*(4), Article 4. https://doi.org/10.2307/1353073

Shaffer, A. J., Gross, J., Black, M., Kalagher, A., & Juanes, F. (2023). Dynamics of juvenile salmon and forage fishes in nearshore kelp forests. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *33*(8), 822–832. https://doi.org/10.1002/aqc.3957

Shaffer, A. J., Munsch, S. H., & Cordell, J. R. (2020). Kelp forest zooplankton, forage fishes, and juvenile salmonids of the northeast Pacific nearshore. *Marine and Coastal Fisheries*, *12*(1), Article 1. https://doi.org/10.1002/mcf2.10103

Shelton, A., Francis, T., Williams, G., Feist, B., Stick, K., & Levin, P. (2014). Habitat limitation and spatial variation in Pacific herring egg survival. *Marine Ecology Progress Series*, *514*, 231–245. https://doi.org/10.3354/meps10941

Short, F. T. (1987). Effects of sediment nutrients on seagrasses: Literature review and mesocosm experiment. *Aquatic Botany*, *27*(1), 41–57. https://doi.org/10.1016/0304-3770(87)90085-4

Short, F. T., & Burdick, D. M. (1996). Quantifying eelgrass habitat loss in relation to housing development and Nitrogen loading in Waquoit Bay, Massachusetts. *Estuaries*, *19*(3), 730. https://doi.org/10.2307/1352532

Short, F. T., Polidoro, B., Livingstone, S. R., Carpenter, K. E., Bandeira, S., Bujang, J. S., Calumpong, H. P., Carruthers, T. J. B., Coles, R. G., Dennison, W. C., Erftemeijer, P. L. A., Fortes, M. D., Freeman, A. S., Jagtap, T. G., Kamal, A. H. M., Kendrick, G. A., Judson Kenworthy, W., La Nafie, Y. A., Nasution, I. M., … Zieman, J. C. (2011). Extinction risk assessment of the world's seagrass species. *Biological Conservation*, *144*(7), 1961–1971. https://doi.org/10.1016/j.biocon.2011.04.010

Short, F. T., & Wyllie-Echeverria, S. (1996). Natural and human-induced disturbance of seagrasses. *Environmental Conservation*, *23*(1), 17–27. https://doi.org/10.1017/S0376892900038212

Sigler, M. F., & Csepp, D. J. (2007). Seasonal abundance of two important forage species in the North Pacific Ocean, Pacific herring and walleye pollock. *Fisheries Research*, *83*(2–3), 319–331. https://doi.org/10.1016/j.fishres.2006.10.007

Smale, D. A. (2020). Impacts of ocean warming on kelp forest ecosystems. *New Phytologist*, *225*(4), 1447–1454. https://doi.org/10.1111/nph.16107

Smale, D. A., Burrows, M. T., Moore, P., O'Connor, N., & Hawkins, S. J. (2013). Threats and knowledge gaps for ecosystem services provided by kelp forests: A northeast Atlantic perspective. *Ecology and Evolution*, *3*(11), 4016–4038. https://doi.org/10.1002/ece3.774

Sogn Andersen, G., Moy, F. E., & Christie, H. (2019). In a squeeze: Epibiosis may affect the distribution of kelp forests. *Ecology and Evolution*, *9*(5), 2883–2897. https://doi.org/10.1002/ece3.4967

Springer, Y., Hays, C. G., Carr, M. H., & Mackey, M. R. (2010). Toward ecosystem-based management of marine macroalgae—The bull kelp, *Nereocystis Luetkeana*. In *Oceanography and Marine Biology*. Chapman and Hall/CRC.

Steneck, R. S., Leland, A., McNaught, D. C., & Vavrinec, J. (2013). Ecosystem flips, locks, and feedbacks: The lasting effects of fisheries on Maine's kelp forest ecosystem. *Bulletin of Marine Science*, *89*(1), 31–55. https://doi.org/10.5343/bms.2011.1148

Stevens, B. G., Armstrong, D. A., & Cusimano, R. (1982). Feeding habits of the dungeness crab *Cancer magister* as determined by the index of relative importance. *Marine Biology*, *72*(2), 135–145. https://doi.org/10.1007/BF00396914

Strain, E. M. A., Thomson, R. J., Micheli, F., Mancuso, F. P., & Airoldi, L. (2014). Identifying the interacting roles of stressors in driving the global loss of canopy-forming to mat-forming algae in marine ecosystems. *Global Change Biology*, *20*(11), 3300–3312. https://doi.org/10.1111/gcb.12619

Sweetman, A. K., Norling, K., Gunderstad, C., Haugland, B. T., & Dale, T. (2014). Benthic ecosystem functioning beneath fish farms in different hydrodynamic environments. *Limnology and Oceanography*, *59*(4), 1139–1151. https://doi.org/10.4319/lo.2014.59.4.1139

Szypulski, E. J. (2018). *Ecological effects of overwater structures on subtidal kelp, northern Puget Sound, Washington* [Master of Science]. Central Washington University.

Tait, L. W. (2019). Giant kelp forests at critical light thresholds show compromised ecological resilience to environmental and biological drivers. *Estuarine, Coastal and Shelf Science*, *219*, 231–241. https://doi.org/10.1016/j.ecss.2019.02.026

Taormina, B., Kutti, T., Olsen, S. A., Sævik, P. N., Hannisdal, R., Husa, V., & Legrand, E. (2024). Effects of aquaculture effluents on the slender sea pen *Virgularia mirabilis*. *Scientific Reports*, *14*(1), 9385. https://doi.org/10.1038/s41598-024-59613-3

Teagle, H., Hawkins, S. J., Moore, P. J., & Smale, D. A. (2017). The role of kelp species as biogenic habitat formers in coastal marine ecosystems. *Journal of Experimental Marine Biology and Ecology*, *492*, 81–98. https://doi.org/10.1016/j.jembe.2017.01.017

Terlizzi, A., De Falco, G., Felline, S., Fiorentino, D., Gambi, M. C., & Cancemi, G. (2010). Effects of marine cage aquaculture on macrofauna assemblages associated with *Posidonia oceanica* meadows. *Italian Journal of Zoology*, *77*(3), 362–371. https://doi.org/10.1080/11250000903464075

Theuerkauf, S. J., Barrett, L. T., Alleway, H. K., Costa-Pierce, B. A., St. Gelais, A., & Jones, R. C. (2022). Habitat value of bivalve shellfish and seaweed aquaculture for fish and invertebrates: Pathways, synthesis and next steps. *Reviews in Aquaculture*, *14*(1), Article 1. https://doi.org/10.1111/raq.12584

Thom, R. M., & Hallum, L. (1990). *Long-term changes in the areal extent of tidal marshes, eelgrass meadows and kelp forests of Puget Sound* (p. 116). University of Washington.

Thom, R. M., Judd, C., Buenau, K., & Cullinan, V. (2011). Eelgrass (Zostera marina L.) Stressors in Puget Sound. *Pacific Northwest National Laboratory*.

Thom, R. M., Southard, S., & Borde, A. (2014). Climate-linked mechanisms driving spatial and temporal variation in eelgrass (*Zostera marina L.*) growth and assemblage structure in Pacific Northwest Estuaries, U.S.A. *Journal of Coastal Research*, *68*, 1–11. https://doi.org/10.2112/SI68-001.1

Thom, R. M., Southard, S. L., Borde, A. B., & Stoltz, P. (2008). Light requirements for growth and survival of eelgrass (*Zostera marina* L.) in Pacific Northwest (USA) Estuaries. *Estuaries and Coasts*, *31*(5), Article 5. https://doi.org/10.1007/s12237-008-9082-3

Trimble, A. C., Ruesink, J. L., & Dumbauld, B. R. (2009). Factors preventing the recovery of a historically overexploited shellfish species, *Ostrea lurida* Carpenter 1864. *Journal of Shellfish Research*, *28*(1), 97–106. https://doi.org/10.2983/035.028.0116

Tsiaras, K., Tsapakis, M., Gkanassos, A., Kalantzi, I., Petihakis, G., & Triantafyllou, G. (2022). Modelling the impact of finfish aquaculture waste on the environmental status in an Eastern Mediterranean allocated zone for aquaculture. *Continental Shelf Research*, *234*, 104647. https://doi.org/10.1016/j.csr.2022.104647

Tsutsumi, H. (1995). Impact of fish net pen culture on the benthic environment of a cove in South Japan. *Estuaries*, *18*(1), 108. https://doi.org/10.2307/1352286

Van Der Schatte Olivier, A., Jones, L., Vay, L. L., Christie, M., Wilson, J., & Malham, S. K. (2020). A global review of the ecosystem services provided by bivalve aquaculture. *Reviews in Aquaculture*, *12*(1), 3–25. https://doi.org/10.1111/raq.12301

Veggerby, K. B., Scheuerell, M. D., Sanderson, B. L., Kiffney, P. M., & Ferriss, B. E. (2024). Shellfish aquaculture farms as foraging habitat for nearshore fishes and crabs. *Marine and Coastal Fisheries*, *16*(2), e10282. https://doi.org/10.1002/mcf2.10282

Von Biela, V. R., Newsome, S. D., Bodkin, J. L., Kruse, G. H., & Zimmerman, C. E. (2016). Widespread kelp-derived carbon in pelagic and benthic nearshore fishes suggested by stable isotope analysis. *Estuarine, Coastal and Shelf Science*, *181*, 364–374. https://doi.org/10.1016/j.ecss.2016.08.039

Wang, X., Andresen, K., Handå, A., Jensen, B., Reitan, K., & Olsen, Y. (2013). Chemical composition and release rate of waste discharge from an Atlantic salmon farm with an evaluation of IMTA feasibility. *Aquaculture Environment Interactions*, *4*(2), 147–162. https://doi.org/10.3354/aei00079

Wang, X., Cuthbertson, A., Gualtieri, C., & Shao, D. (2020). A review on mariculture effluent: Characterization and management tools. *Water*, *12*(11), 2991. https://doi.org/10.3390/w12112991

Washington Department of Fish and Wildlife. (2024). *Forage Fish Spawning Map—Washington State*. https://arcg.is/1bzTHv

Washington Department of Natural Resources. (2014). *Comparison of light transmitted through different types of decking used in nearshore over-water structures*. DNR AAMT.

Washington Marine Resources Advisory Council. (2017). *Ocean Acidification: From knowledge to action, Washington State's strategic response*.

Washington Sea Grant. (2015). Shellfish aquaculture in Washington state. *Final Report to the Washington State Legislature*, 84.

Washington State Department of Ecology. (1986). *Recommended interim guidelines for the management of salmon net-pen culture in Puget Sound.* (Ecology Contract No. C-0087110; p. 48). Washington Department of Ecology.

Washington State Department of Ecology. (2021). *Salmon net pen water quality permits*. https://ecology.wa.gov/Water-Shorelines/Shoreline-coastal-management/Aquaculture/Net-pens

Washington State Department of Ecology. (2022). *Commercial marine finfish net pen aquaculture in Puget Sound and Strait of Juan de Fuca*. https://apps.ecology.wa.gov/publications/documents/2206008.pdf

Watanabe, H., Ito, M., Matsumoto, A., & Arakawa, H. (2016). Effects of sediment influx on the settlement and survival of canopy-forming macrophytes. *Scientific Reports*, *6*, 18677. https://doi.org/10.1038/srep18677

Waycott, M., Duarte, C. M., Carruthers, T. J. B., Orth, R. J., Dennison, W. C., Olyarnik, S., Calladine, A., Fourqurean, J. W., Heck, K. L., Hughes, A. R., Kendrick, G. A., Kenworthy, W. J., Short, F. T., & Williams, S. L. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences*, *106*(30), 12377–12381. https://doi.org/10.1073/pnas.0905620106

Weigel, B. L., Small, S. L., Berry, H. D., & Dethier, M. N. (2023). Effects of temperature and nutrients on microscopic stages of the bull kelp (*Nereocystis luetkeana*, Phaeophyceae). *Journal of Phycology*, *59*(5), 893–907. https://doi.org/10.1111/jpy.13366

Wernberg, T., Smale, D. A., Tuya, F., Thomsen, M. S., Langlois, T. J., de Bettignies, T., Bennett, S., & Rousseaux, C. S. (2013). An extreme climatic event alters marine ecosystem structure in a global biodiversity hotspot. *Nature Climate Change*, *3*(1), 78–82. https://doi.org/10.1038/nclimate1627

Weston, D. P. (1986). *The environmental effects of floating mariculture in Puget Sound* (p. 148). Schoola of Oceanography University of Washington. https://repository.library.noaa.gov/view/noaa/9240

White Paper: The Washington Shellfish Initiative. (2011). State of Washington.

Wildish, D. J., Hargrave, B. T., MacLeod, C., & Crawford, C. (2003). Detection of organic enrichment near finfish net-pens by sediment profile imaging at SCUBA-accessible depths. *Journal of Experimental Marine Biology and Ecology*, *285–286*, 403–413. https://doi.org/10.1016/S0022-0981(02)00540-3

Williamson, K. J. (2006). *Relationships between eelgrass (*Zostera marina*) habitat characteristics and juvenile Dungeness crab (*Cancer magister*) and other invertebrates in Southern Humboldt Bay, CA, USA*. Humboldt State University.

Wilson, U. W., & Atkinson, J. B. (1995). Black brant winter and spring-staging use at two Washington coastal areas in relation to eelgrass abundance. *The Condor*, *97*(1), 91–98. https://doi.org/10.2307/1368986

Wright, A., Bohrer, T., Hauxwell, J., & Valiela, I. (1995). Growth of epiphytes on *Zostera marina* in estuaries subject to different nutrient loading. *The Biological Bulletin*, *189*(2), 261–261. https://doi.org/10.1086/BBLv189n2p261

Wu, R. S. S. (1995). The environmental impact of marine fish culture: Towards a sustainable future. *Marine Pollution Bulletin*, *31*(4–12), 159–166. https://doi.org/10.1016/0025-326X(95)00100-2

Zhao, Y.-P., Bi, C.-W., Chen, C.-P., Li, Y.-C., & Dong, G.-H. (2015a). Experimental study on flow velocity and mooring loads for multiple net cages in steady current. *Aquacultural Engineering*, *67*, 24–31. https://doi.org/10.1016/j.aquaeng.2015.05.005

Zhao, Y.-P., Bi, C.-W., Chen, C.-P., Li, Y.-C., & Dong, G.-H. (2015b). Experimental study on flow velocity and mooring loads for multiple net cages in steady current. *Aquacultural Engineering*, *67*, 24–31. https://doi.org/10.1016/j.aquaeng.2015.05.005