**Comments and Recommendations for Benthic and Water Quality Standards for Potential Inclusion in Marine Fish Aquaculture Model Standards for Iceland**

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The magnitude of potential environmental (benthic and water quality) impacts from sea-cage finfish aquaculture will vary depending on local current speeds, substrate (soft/mud, mixed, gravel/cobble bottom) type, water depth, bathymetry, latitude (temperate, tropical), farm size and husbandry methods (Holmer et al. 2008; Borja et al.2009; Hargrave 2010, Price et al. 2015). The importance of using an ecosystem-based approach as the process for evaluating site suitability for marine finfish aquaculture operations cannot be overstated. This approach includes: the protection of non-target species, vulnerable species, habitats and trophic interactions; the protection of essential habitats to sustain species diversity and abundance; and the protection of endangered and threatened species. Estimates of carrying capacity, robust site selection processes and the use of local ecological knowledge are key tools in implementing an ecosystem-based approach.

The following comments and recommendations are made in relation to the benthic and water quality standards established by the Aquaculture Stewardship Council (version 3.1).

**Benthic Indicators**

* Redox Potential (EhNHE) is a measure of oxidation-reduction potential in sediments and is an indirect indicator of aerobic versus anaerobic conditions

A redox value of < 0 mV (milliVolt) is widely accepted as a threshold value that reflects the transition from normal benthic status to polluted status (Hargrave 2010). Maintaining redox values at > 0 mV is a widely accepted standard.

* Sulphide level reported in µM (micromolar) is a measure of the accumulation of 'free' dissolved sulphides that include Σ S2-, HS-, H2S. Sulphides are a major product of sulfate reduction that occurs under anaerobic conditions. This is a sensitive indicator of habitat degradation due to organic loading and currently the main indicator currently used to determine direct impact of an aquaculture operation.

A sulphide level of ≥ 1500 µM is widely accepted as a threshold value indicating organic enrichment (pollution) impacts from fish farms on soft-bottom benthic habitats. Maintaining benthic sulphide levels at ≤ 1500 µM is a widely accepted standard.

For hard bottom (cobble, rock) or patchy (soft sediment and bedrock) benthic habitats, it is often difficult to obtain consistent and sufficient volumes of benthic sediment needed to measure redox potential and sulphides. Consequently, jurisdictions such as Canada, US (Maine) and Norway utilize visual (video) monitoring methods to assess benthic impact (Hamoutene et al. 2018; Hamoutene 2014). In Canada, operators cannot restock sites if visual monitoring reveals 70% of the sea bottom locations specified in the monitoring standard is covered by white mats of sulphide oxidizing bacteria (*Beggiatoa* spp.), opportunistic polychaete complexes (OPC), and/or barren substrates (i.e., with no visible epifauna) (AAR 2015).

Hamoutene et al (2018) examine the video monitoring data collected for 12 sea-cage aquaculture operations in Newfoundland as per the protocols established in Canadian regulatory and advisory documents (DFO 2012). They found that the present Canadian regulatory threshold (70%) for hard bottoms would likely correspond to a 100% reduction in taxonomic richness (number of species) in the near-cage area (within 50m from netpens) (Harmoutene et al. 2018). Verhoeven et al. (2018) examined the bacterial community composition in ﬂocculent matter (loose clumps or aggregation of decaying ﬁsh-feed pellets, ﬁsh fecal matter, microbes and other organic matter) on hard bottoms around fish farms in Newfoundland . Using 16S rRNA gene high throughput sequencing , they characterized the bacterial communities in samples collected over a period of 3 years and at various distances from cages (0–200 meters) at production and fallow (3–35 months) salmon aquaculture sites. Bacteria associated with "high impacts" were still found under net pens that had been fallowed for 35 months indicating a lack of benthic recover (Verhoeven et al. 2018).

Limited research suggests that a monitoring threshold of 70% coverage of bacteria mats, barren stations and opportunistic polychaete complexes (OPC) on hard bottom environments is likely too high to sufficiently protect local biodiversity and habitat quality and that hard bottoms may be prone to slow recovery (Salvo et al. 2017; Verhoeven et al. 2018).

* ***Recommendation - Until further research is done to detail the biological community response and recovery to organic loading associated with sea-cage aquaculture on hard bottom benthic communities, the regulatory threshold for compliance should be set at a precautionary level of 25% cover by bacterial mats, aggregates of OPC, and/or barren substrates.***

 **Faunal Index Scores**

* AZTI Marine Biotic Index (AMBI) Score ≤ 3.3

Shannon-Weiner Index Score > 3

Benthic Quality Index (BQI) score ≥ 15

Infaunal Trophic Index (ITI) score ≥ 25

These indices are measures of the response of the benthic biological community in soft bottom environments to organic pollution (loading/enrichment). Indices can be simple measure such as Shannon-Weiner Index which is a measure of species diversity (abundance and eveness) or complex multi-metric measures such as BQI and AMBI. These indices rely on the collection and identification of benthic macroinvertebrate indicator species which is time consuming, expensive and requires taxonomic expertise. Because of the high costs per unit/area sampled, compliance monitoring generally involves collecting only a few samples per farm site once a year and therefore providing low spatial and temporal resolution to the environmental monitoring (Pawlowski et al. 2016; Stoeck et al. 2018).

Several studies have evaluated the suitability of various indices for assessing benthic environment quality (Grémare et al. 2009; Martínez-Crego et al. 2011; Borja et al. 2015; Berthelsen et al. 2018) including impacts of aquaculture (Borja et al. 2009; Keeley et al. 2012). Each index has strengths and weakness in different environmental conditions (e.g. low vs high current flow regimes), biogeographic provinces (e.g. Arctic, cold temperate northeast Atlantic, tropical Atlantic) and transitional waters (e.g., coastal, estuarine, lagoons). Grémare et al. (2009) compared the performance of two of the more widely used indices (AMBI and BQI) on a pan-European scale that included 2158 marine stations located in the Celtic-Biscay Shelf, the Mediterranean, the North Sea and the Norway and Barents Seas). They found major differences in the way these indices assessed the sensitivity/tolerance level of individual species and indices were shown to locally result in different ecological status assessments (Grémare et al. 2009). Consequently, a combination of indices is generally recommended (Keeley et al. 2012). A major weakness in the calculation of faunal indices is the lack of definition of reference or baseline conditions (Martínez-Crego et al. 2011). Faunal indices for aquaculture operation are calculated and compared to each other during production cycles rather than against baseline, pre-production values.

* ***Recommendation - Pre-production faunal indices should be calculated using a combination of indices. Farm-derived faunal indices and ecological status should be evaluated against baseline faunal scores.***

As indicated earlier, these indices apply to assessing benthic community impacts on soft bottom environments where samples are collected using grabs or cores. In hard bottom environments, it is more difficult to get a consistent sample volume using these methods. Over the past few years bacterial and foraminiferal (amoeboid protists) eDNA metabarcoding have been examined as an alternative cost-eﬀective tool for assessing benthic community impacts from aquaculture (Pawlowski et al. 2016; Stoeck et al. 2018; Rubio-Portillo et al. 2019), including hard bottom environments (Verhoeven et al. 2018). This method uses taxonomically informative genes obtained from environmental samples and high-throughput sequencing to inform the composition of either bacterial or formaniferal communities in environmental samples. Samples can be analyzed quickly, at a lower cost than traditional monitoring of macrofaunal communities and data can be analyzed in days rather than months (Stoeck et al. 2018). However, several issues will have to be addressed before the implementation of bacterial metabarcodes into routine monitoring programs becomes feasible (Stoeck et al. 2018).

* ***Recommendation - given the lack of faunal indices of benthic community change on hard bottom communities and the potential for hard bottoms environment to be prone to slow recovery, a regulatory threshold for environmental compliance of 25% cover by bacterial mats, aggregates of OPC, and/or barren substrates will be even more critical to protecting local biodiversity and environmental quality.***

 **Copper**

* < 34 mg Cu / kg dry sediment weigh

Algae, molluscs and crustaceans are among the most sensitive species to copper (Burridge et al, 2010), hence its use as an antifoulant in nets and paint. Copper is also a trace element used in fish feed. The Canadian Council of Ministers for the Environment (CCME) have set an interim copper sediment (top 5 cm) guideline of 18.7 mg/kg Cu dry weight to protect marine life and a probable effects level of 108 mg/kg Cu dry weight (CCME 1999). In Canada, copper levels of 100-150 mg/kg dry weight have generally been reported at fish farms in southwest New Brunswick (Burridge et al. 2010). In Scotland, the concentration of copper (and zinc) in 5 cm depth exceeded the Scottish Sediment Quality Criteria at both the upper 270 mg/kg dry weight and lower (108 mg/kg dry weight) levels for substantial part of the aquaculture Allowable Zone of Effects (AZE) (Dean et al. 2007). Behavioural effects (e.g. slower burrowing times) on clams have been reported at sediment copper levels ranging from 5.8 - 15 mg/kg Cu dry weight (Burridge et al. 2010).

* **Recommendation - Sediment copper threshold value should be lowered to at least 15 mg/kg Cu dry weight.**

Copper from sediments can be transferred to the sea surface from fish feed oils or sulphide gas bubbles released from decomposing fish feces/feed where they can concentrate and move horizontally (Loucks et al. 2012). Due to buoyancy, lipids (fish feed oils) released from a farm site (seen as a visible slick on the sea surface) can be expected to disperse widely in the sea surface microlayer as it is moved by wind and tidal action (Price et al. 2015). Little research or monitoring data are available directly addressing lipid levels in the water or on the sea surface near marine cages. In a Canadian study, copper levels measured in the sea surface microlayer one kilometer away from a fish farm fallowed for two years were several times higher than marine life protection guidelines (3 µg Cu/L) in British Columbia (Canada)and in excess of the B.C. guidelines (Loucks et al. 2012). Copper levels of 0.5 µg Cu/L have been recommended for the protection of larval planktonic phases of decapod species such as lobster (Mariño-Balsa et al. 2000) which can be found in coastal waters (Cobb and Wahle 1994).

* **Recommendation - Where a surface slick is seen originating from a fish farm, sea surface sampling must be performed. Sea surface copper levels should not exceed 0.5 µg Cu/L.**

**Water Quality**

* Dissolved oxygen ≥ 70% oxygen saturation

Natural seasonal, tidal and diurnal fluxes can cause low levels of dissolved oxygen. Proper siting of farms in areas with sufficient flushing rates is recommended.

* Nitrogen (NO2, NO3)

Phosphorous

pH

Salinity

Turbidity

Phytoplankton

Ammonium

Natural marine water quality can be highly variable seasonally, temporally and spatially and it would be difficult to establish a set of regulatory/monitoring water quality values that would apply to all aquaculture operations. Studies to detect changes in phosphorous, dissolved inorganic nitrogen (DIN), ammonium, phytoplankton (chlorophyll *a*), and turbidity around fish farms are highly variable in their results (Price et al. 2015). Several factors account for a lack of detection or response in the pelagic ecosystem to nutrient loading. DIN and other dissolved nutrients are highly labile and their concentrations vary diurnally. Unless sampling is frequent, DIN can be easily undetected. In temperate, nutrient-limited waters such as Iceland, not all DIN will be assimilated in the lower planktonic food web. Once inorganic nitrogen is released from a near-shore sea-cage fish farm, it is transported by currents and quickly available to numerous biological sinks in the pelagic ecosystem including marine bacteria and macrophytes in subtidal and intertidal areas, as well as by phytoplankton (McIver et al. 2018). Fast-growing annual macroalgae (also referred to as 'nuisance' or 'opportunistic' algae) can quickly take up nutrients. Increased growth and biomass of annual algal epiphytes on perennial brown canopy-forming algae (e.g. kelp, rockweed), green annual algae, *Ulva* sp., and a loss of seagrasses have been reported near fish farms at distances greater than compliance monitoring requirements (Robinson et al. 2005; Cullain et al. 2018; McIver et al. 2018)

* **Recommendation - It is important that proposed aquaculture operations be required to collect a comprehensive suite of local baseline water quality levels that account for seasonal, temporal, and spatial variability for the parameters identified above.**

**A note on far-field effects and cumulative effects**

Environmental effects monitoring of sea-cage aquaculture is generally limited to an area 25-50m from the sea-cage edge, an area referred to as the 'zone of effects' or 'zone of impacts', and reference sites (generally two sites; one upstream and one downstream from the sea-cages) that vary from 100-500 m from the sea-cages. Despite an increasing number of studies that have identified a range of environmental impacts (e.g. eutrophication, benthic and water quality changes, seagrass loss, lobster catch) at distances greater than sampling distances prescribed in regulations (Loucks et al. 2012; Law et al. 2014, Price et al. 2015; Bannister et al. 2016, Cullain et al. 2018; McIver et al. 2018; Milewski et al. 2018), regulators have not incorporated any metrics to monitor far-field effects. In addition, assessing and monitoring the cumulative impacts of impacts of multiple marine finfish aquaculture operation alone or in combination with other human activities has proven difficult to incorporate into environmental assessment and monitoring processes (King and Pushchak 2008). Both of these gaps could be addressed with a meaningful integration of local (community) ecological knowledge, particularly the knowledge of fishers, with traditional science knowledge (Wiber et al. 2012; Brattland 2013; Maillet et al. 2017; Milewski et al. 2018) for more effective marine resource management and avoiding fisheries and ecosystem impacts from sea-cage aquaculture operations.

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