Aligned ASC Farm Standard

TWG Recommendations to ASC for Revised Water Quality Indicators

Lakes and Reservoirs

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Background

This document details the recommendations to ASC arising from the Water Quality (WQ) Technical Working Group (TWG) review meetings held between June and December 2021. The TWG was tasked by ASC to revise the current approach in the ASC Standards and recommend water quality "criteria" and "indicators" that collectively reflect and address risks from aquaculture in all major production systems[1] that discharge into different water types[2]. This should reflect the latest scientific knowledge and current best practices within the aquaculture industry. This first set of recommendations focuses on open production systems in lakes and reservoirs[3]. Recommendations for other production systems discharging into other water types will be developed by the TWG during 2022.

Scope

Existing ASC water quality indicators aim to minimise the risks of anthropogenic eutrophication in sensitive aquatic ecosystems – arising from 'waste' nutrients originating from aquaculture feed and (pond) fertiliser inputs (in faeces, waste feed and as metabolic by-products). For this revision, three associated criteria were formulated as a basis for the development of indicators and related requirements:

1. ***Siting:*** farm siting requirements based on water body classification according to trophic status and more localised stratification characteristics.
2. ***Impacts*:** monitoring causal and more direct biotic impacts in receiving waters.
3. ***Input-output management (IOM)***: limiting nutrient inputs and outputs.

Revision

1. 'Siting requirements' linked to water body classification

Trophic classification and assimilative capacity modelling in current standards

Currently, only indicators for cage-systems in freshwater lakes and reservoirs directly link siting and receiving water monitoring (i.e., *impact*) requirements to a classification of trophic status. The salmonid (FW Trout and Salmon) and Tilapia standards use a modified version of the trophic status system developed by the Organization for Economic Cooperation Development (OECD; Vollenweider and Kerekes, 1982) using total phosphorus concentration [TP] as an indicator of trophic status. The salmonid standards stipulate that regardless of the source of nutrient input, if [TP] rises to the point that the lake's trophic status changes, or if it rises more than 25% from a measured baseline, production would no longer be certifiable in that lake as this would likely result in a significant ecosystem structure and function alteration. Open cage (net-pen) farms in smaller lakes with a surface area <1,000 km^2 must also demonstrate that an assimilative capacity assessment has been conducted to determine if there is sufficient capacity for proposed additional loading. Assimilative capacity modelling is not required for larger lakes >1,000km^2 '*because of the difficultly of conducting such studies and linking them to the appropriate production levels of an individual farm*' at these scales. Consequently, these systems also impose a more precautionary limit of 20% TP change from

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[1] Major production systems are: cages, suspended/off-bottom, in or on-bottom and land-based (point-discharge systems, e.g. ponds, race-way, flow-through and RAS).

[2] Water types are: marine, brackish, freshwater. Other interacting abiotic factors including hydraulic residence-time, energy conditions and water temperature profile are further key determinants of nutrient fate.

[3] Although reservoirs are man-made systems constructed for primary utility functions (e.g., irrigation, hydro-electric, water supply) they can also support other important ecosystem functions and services. Under this context, and for the purpose of this document, when lakes are mentioned they shall also encompass reservoirs.

baseline and farms must instead demonstrate location at sites least sensitive to nutrient discharges characterised as being (i) exposed to more energetic conditions, (ii) having a connection to deep offshore waters and/ or (iii) out with hydrodynamically isolated embayments. Based on these criteria farms in larger water bodies must be located in a **'Type 3 Site'**[4] defined as *exposed locations where the hypolimnion (the densest bottom layer in a thermally stratified lake) is also well flushed*.

Stratification

Anoxic conditions in the hypolimnion cause reduction and liberation of oxidised phosphorous in the benthos, which can come to the surface during turn-over events in seasonally stratified (holomictic) lakes causing algal blooms and acute fish kills. Deep oligotrophic lakes, even if permanently stratified (meromictic), can have significant oxygen and support fish life down to the depth to which seasonal mixing occurs but will become oxygen-deficient (hypoxic) below this depth. Thus, standard requirements aim to preclude turn-over risks in the meta and epilimnion (middle and surface waters) and exacerbation of hypoxia at depth. The salmonid standards (FW Trout and Salmon) preclude siting in meromictic lakes for the latter reason.

Indicators

Whilst there is a broad consensus around P being limiting for eutrophication in temperate FW systems, greater uncertainty exists for tropical systems where under certain conditions N may become the limiting macro-nutrient (and N&P may become co-limiting in mesotrophic systems). Challenges are also associated with the more reactive/labile nature and highly variable volatilisation rates of N compared to P. Consequently, the Tilapia standard mandates receiving water (RW) monitoring requirements for P and Chl-a; the latter being a direct indicator of eutrophication impact[5], based on a tiered decision-tree with steps contingent on Secchi disk (SD) transparency readings as a simple and robust indicator of trophic status at lower primary productivity levels where more precautionary measures including siting preclusions may be merited.

Only the Salmon Standard requires estimation of RW BOD, a second direct indicator of eutrophication, requiring a production-cycle based mass-balance calculation based on estimation of TN and TC in feed and harvested fish[6] i.e., highlighting the fact that N is likely to be the primary macro-nutrient limiting bacterial growth with relatively greater influence below the photic-zone at depth. No limit is currently set on BOD; only monitoring is required. Nor is there any specific requirement for laboratory analysis of water samples that would provide a more direct measure of localised impact.
The Salmon, FW Trout and Tilapia standards also preclude certification of cage-farms sited in mesotrophic systems ([TP] > 20 ug/l) with derogations for closed land-based systems, whilst the Tilapia Standard also precludes cage-farm certification (only) in ultra-oligotrophic systems (SD > 10m) and (mesotrophic) lakes with Chlorophyll-a concentration ≥ 4.0 µg/L.

Guiding Ideas

In the context of the above, the TWG has strived to develop a review of ASC's current approach to WQ and built on the following guiding ideas:

- To allow certification/siting of farms in highly sensitive e.g., oligotrophic lakes and reservoirs, there should be high confidence in the water body having sufficient assimilative capacity to absorb the additional nutrients without unduly threatening the water body's natural ecosystem structure and function.

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[4] Based on a classification developed by the Ontario Ministry of Environment (Boyd et al 2001). The Salmon Standard also requires that if the farm's local regulator has not already classified the site using such a system, or it is not in place, an independent consultant must certify (with detailed supporting analysis) that the farm's location is consistent with the Type 3 definition.

[5] Particularly in tropical/ sub-tropical systems with less marked seasonal light and temperature shifts.

[6] BOD is calculated as: ((total N in feed – total N in fish) *4.57) + ((total C in feed – total C in fish) *2.67). A farm may deduct N or C that is captured, filtered or absorbed through approaches such as IMTA or through direct collection of nutrients wasted. In this equation, "fish" refers to harvested fish. Reference for calculation methodology: Boyd C. 2009

- Otherwise, siting should account for the hydrological conditions and bathymetry within a given water body or area of a water body, ensuring there is sufficient depth, flow and mixing for broad distribution of waste i.e., that does not create unacceptable down-stream impact.
- Confidence comes from having sufficient data collected to monitor the trophic status and understanding whether and when a water body (or localised part of a water body) may be approaching a significant trophic transition point.
- Siting should be permitted only in areas where the cumulative impact of all farms and other anthropogenic and natural inputs avoid “tipping points” that are difficult to reverse.
- Requirements also need to consider the farm’s actual contribution and the possibility for collective coordinated efforts around nutrient loading at a ‘landscape’/ area-based level.

1.1. TWG recommendations for ‘siting’ indicators

Based on the guiding ideas above, the TG is recommending the following:

Classification of trophic status

- Initial trophic status characterization should require data collection on a uniform bundle of indicators -- across all settings, from temperate to tropical, to include TN, TP, and Secchi disk measurements.
- Chl-a could be added as an impact indicator in a second level in a tiered decision tree, i.e., reflecting co-variance due to other secondary factors (e.g., seasonal influence on temperature and light levels). Its inclusion should also ideally be subject to the availability of rapid *in-situ* measurement methods. Appropriate Chl-a sensors are now available. Further discussion is required as to whether Chl-a should be retained as a primary, i.e., baseline, impact metric also requiring more routine monitoring.
- A requirement for DO and temperature depth profiling around cages should be introduced. This could augment existing siting requirements linked to stratification characteristics and/ or impact requirements in Section 2 of this summary.

  Rationale: This list of indicators incorporates a mix of correlated/ complementary indirect (causal) and direct impact indicators – improving confidence around data validity and potential to assess shifts in N or P becoming the limiting factor for eutrophication.

Recommendation for a revised indicator:

1) To conduct an initial trophic status characterization of the water body based on measurement of tier 1 indicators: [TN], [TP], and transparency (Secchi disk).
2) To add [Chl-a] as a tier 2 indicator - considering co-variance associated with other secondary abiotic factors (e.g., seasonal/ latitude influence on temperature and light levels).
3) To measure baseline DO & temperature depth-profiles around the unit(s) of certification, in order to monitor and preclude impacts associated with localised stratification characteristics (further detail provided under the ‘impact’ criteria in Section 2).

Note: The TWG also acknowledged that a universal requirement for a suite of indicators may create an unnecessary burden in some situations (e.g., in oligotrophic temperate lochs/ reservoirs where P alone may be sufficient). An appropriately designed tiered decision-making approach may provide a solution to this concern (the existing decision tree in the Tilapia Standard is a good example but would require further conditionalities to accommodate temperate systems).

Assimilative capacity modelling

- For smaller systems (<1,000km^2), cage-farms in ultra-oligotrophic waters should be certifiable under specific conditions (see notes) if there is a robust[7] assimilative capacity study that shows there will be no change in trophic status.
- The TWG will revisit whether there are other requirements beyond such a study and how this should be addressed in larger systems.

  Rationale: In some Nordic aquaculture may be permitted in highly oligotrophic systems under well-regulated conditions based on carrying-capacity assessments.

- Source apportionment modeling should be used to account for and differentiate between natural and anthropogenic sources of nutrient enrichment. This would (i) facilitate the setting of baseline conditions accounting for natural eutrophication processes and (ii) potentially inform other derogations on eutrophication limits[8].
- Assimilation model results should be systematically validated against empirical evidence, including data collected by certified farmers. Further consideration should be given as to how this might be incorporated in area-based management requirements (see below Section 3).

Recommendation for a revised indicator:

1) In smaller (<1,000km^2) ultra-oligotrophic lakes, to allow certification of cage farms providing that an assimilative capacity study has shown there will be no change in trophic status (subject to qualifications associated with N & P limiting conditions discussed below).
2) To require the use of source apportionment modeling to account for and differentiate between natural and anthropogenic sources of nutrient enrichment.
3) The standard should also permit certification of cage farms in naturally meso-eutrophic systems provided assimilative capacity modelling can be reliably confirmed, based on historic data and/ or a source apportionment model.

To complement the recommendations above, the TWG also recommends the inclusion of the guidance:

- There should be clearer guidance on the interpretation of what 'surplus capacity' conclusions mean consistent with standard requirements on change in baseline nutrient concentration and shifts in trophic status i.e., phrasing to specifically prohibit crossing of trophic boundary limits and % changes in nutrient concentrations relative to baseline conditions.
- The modelling approach should reflect the degree of eutrophication risk based on a preliminary assessment of anthropogenic pressure relative to the trophic status and scale of the system. At a minimum this could incorporate a review of secondary data and catchment land and water use characteristics from satellite imagery.
- Guidance should advise that the assimilative capacity model used should be appropriate for the type of lake under study, including the systems stratification characteristics

Siting requirements in larger water bodies (>1,000km^2)

- Whilst the 1,000km^2 delineation between smaller and larger water bodies is somewhat arbitrary it was considered to be a broadly appropriate heuristic for not requiring assimilative capacity modelling of entire systems. However, in follow-up discussions, there was a suggestion that depth characteristics might also be considered.

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[7] See points regarding model verification and accreditation for implementing experts/ institutions

[8] Two approved VRs presented independent evidence indicating that the contribution of aquaculture effluents to deterioration of water quality (& eelgrass beds) in the Danish Storebæt (a strait in the Kattegat Sea) was low relative to other sectors being targeted for remediation including agriculture, industry and domestic wastewater. Note –approval appears somewhat inconsistent with example of the VR for Andean Trout farm footnoted above – where approval was contingent on demonstration of natural eutrophication processes – not other anthropogenic causes.

Rationale: Wide differences were noted between the results of multiple assimilative capacity modelling studies in Lake Toba, Indonesia (approx. 1,130km^2).

Recommendation for a revised indicator:

1) DO and temperature-depth profiling should be required for all systems with marked seasonal or longer-term stratification characteristics.
2) To add monitoring of longer-term DO and temperature stratification trends around the farm as further potential siting indicator. There should be a moratorium on setting specific siting requirements until 5 years of farm data have been collected and reviewed as part of an impact indicator requirement (Section 2). This review should apply to both 'large' and 'small' lakes.
3) In large lakes, where current standards allow for assimilative capacity modelling to be more localised (e.g., for embayments) the standard should go further in trying to preclude farms contributing to wider unacceptable effects elsewhere in the water body as well in the immediate vicinity of a farm.
4) Siting requirements should also consider depth transition zones associated with greater risk of nutrient resuspension associated with a return period of extreme weather events – subject to the 5-year review described above.

Note: Current standards also apply preclusions to permanently stratified systems; limiting siting in larger water bodies to areas where the hypolimnion is well-flushed (i.e., not stratified) and outside hydrodynamically isolated embayments. However, The TWG felt there should be greater specificity regarding lesser degrees/ variability of stratification -- reflecting the potential for more localised impacts, particularly in larger systems.

Rationale: For example (i) the historic return-period of extreme weather-related turn-over events and probability of acute (potentially localised) impacts associated with stratification characteristics on the epilimnion might be evaluated or (ii) the ecological risk associated with more progressive eutrophication of the hypolimnion (potentially also below the photic zone in deeper/ higher trophic status systems) (iii) in deeper permanently stratified systems the benthos may be naturally anoxic but still constitute a stable nutrient repository[9]. This and avoidance of progressively adverse effects on the hypolimnion could be addressed through more precise requirements for depth profiling in baseline setting and monitoring requirements (below).

Determination of baseline trophic status

- The required duration for a baseline data collection period and its sample design (e.g., encompassing the location and frequency of measurements) should reflect the anticipated variability in a system. In the absence of historic data, a minimum of 2 years of data collection was anticipated, though this should be extended if the period encompasses any exceptional events[10]. Background (e.g., collected by local regulatory bodies) as well as foreground (farm) data should be acceptable.

  Rationale: Steady-state equilibria should not be anticipated as eutrophication is a process subject to ongoing and naturally variable rates of change. Thus, in practice, the baseline state will be a parameter average or rate over a defined period of time for which historic water quality data is available/ selected.

[9] Note: a concern was expressed in follow-up discussions about treating lake-bottoms as a dumping ground.
[10] The example of a massive HAB induced by volcanic activity in Lake Toba in 2016 was cited.

Recommendation for a revised indicator:

1) To require that the duration for a baseline data collection period and its sample design reflect the anticipated variability in a water body, and that this period be no shorter than 24 months.
2) The TWG suggested that a geometric mean averaged over at least 12 consecutive months of data (to account for seasonality) would be better than the existing use of an arithmetic mean for baseline and monitoring requirements[11]. Whichever population parameter is applied, it could also be more meaningfully expressed as a confidence interval[12]. Further discussion/ review of best practices is required[13].

   Note: In the absence of historic data, a minimum of 24 months of baseline data collection was anticipated, though this should be extended if the period encompasses any exceptional events.

2. Receiving water 'impacts'

Working alone or in tandem with nutrient input-output indicators (see next section), these indicators are intended as empirical measures of eutrophication levels and impacts. They require measurements (with and without metric limits in existing species-specific standards), on parameters in receiving waters including total organic and inorganic N, total P and ortho-P, total organic carbon (TOC) and more direct indicators of eutrophication impacts; Chl-a concentration, BOD or its impacts; dissolved oxygen (DO), suspended solids/ turbidity levels[14].

Some standards set upper limits on concentrations in receiving waters, others limit net differences, or percentage change in concentration levels in receiving waters (e.g., diurnal changes in DO or from baseline TP), or in the case of 'input-output' efficiency indicators as the difference between levels measured in farm effluent-outflows ('influents') and source-inflow waters. TN & TP receiving water quality monitoring requirements for freshwater cage systems are closely linked to the trophic classification concepts and limits summarised above (unlike input-output management (IOM) requirements – see below). With regards to DO, cage related standards[15] variously set DO limits in receiving waters based on two approaches:

A. Min % saturation of DO averaged over weekly monitoring periods on farm (within cages). Requirement: ≥ 70% saturation.
B. Max % of samples falling below a threshold saturation (% or mg/l) on farm or above bottom sediments (below cages). Requirement: 5% & <2 mg/l.

The Salmon Standard permits an exception [to the 70% limit] for farms that can demonstrate consistency with a reference site in the same water body (i.e., allowing for hypoxic events beyond the control of the farm). Only the FW Trout Standard requires a DO saturation (≥60%) 50cm above bottom sediments consistent with preclusion of the certification of smolt of cage farms in highly stratified lake systems with anoxic bottom conditions. The Salmon Standard also mandates monitoring at '*a depth 5m… where the conditions of the water will be similar to those the fish experience*' i.e., reflecting a further overlap of WQ and welfare requirements.

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[11] A geometric mean is better suited to rate of change metrics (such as eutrophication). Specifically, it would be less sensitive than an arithmetic mean to extreme outlier-events that could distort or mask longer term underlying trends (the geometric will always be lower than the arithmetic mean).

[12] The EU water framework directive and Australian regulatory regimes use quintile ranges to deal with outliers

[13] i.e., confidence intervals based on traditional normal-based calculation or bootstrapping, etc.

[14] Closed production systems (exchanging ≤10% of total water volume per day) in the Salmon Standard are further exempted from certain discharge requirements.

[15] Requirements for land-based systems focus more specifically on localised impacts with a zone of mixing downstream of the effluent discharge point.

Cage farms need to demonstrate 12 months of receiving water TP and DO data, or >= 6 months of data prior to the first audit. Standards also variously require submission of supplementary data to the ASC, to be evaluated to determine the effectiveness of existing and potential creation of new metrics. The Tilapia Standard contains one the most extensive receiving water quality monitoring templates for cage systems, mandating monthly sampling of 8 parameters at 3 stations (Standard Annex 3)

2.1. TWG recommendations on receiving water 'impacts'

Metric limits on eutrophication

- Within its deliberations, the TWG agreed on[16]:
  - Differentiating requirements between situations where N & P become limiting or co-limiting using the mass-balance ratio approach proposed in Annex 1. The simple steady-state nutrient loading model permits estimation of current N & P inputs based on limited epilimnetic WQ data and which macronutrient, N or P, is likely to be limiting[17] (this is viewed as a basic minimum and farms can and should use more robust approaches where needed)
  - The existing 25% and 20% maximum increase from baseline limit for P in smaller and larger water bodies respectively should be maintained where P is limiting, also applying the same maxima to N where it is the limiting nutrient (or both P & N when N & P are co-limiting)
  - Trophic shifts should also be disallowed in any system where P is limiting based on [TP] (the existing case), and/ or when [N] & [P] become co-limiting.
  - It was not considered practicable to impose TSI boundary requirements in N-limited situations (more typical of many tropical systems) as yet there is no recognised classification system based on [N]. This points to a need for greater reliance on existing or additional impact indicators in such situations (potentially including BOD. See below)
  - In natural situations, phosphorus and nitrogen inputs from a catchment do generally increase together[18]. This points to the potential to setting limits on differential shifts in N&P ratios likely to be more indicative of anthropogenic effects (i.e. impacts could be assessed by deviation from a 'natural' ratio for an individual site. Further deliberation is required.

Rationale: Whilst the rate change indicators appear to set relatively arbitrary limits, they were observed to be more conservative than some regulatory approaches. There was also discussion regarding the appropriateness of the first % change requirements as a relative measure of change -- however, this was considered effective operated in conjunction with the absolute requirement to preclude trophic status ('tipping point') shifts together with limits on other direct indicators of eutrophication. Further discussion is required regarding how it could practicably be determined when N becomes limiting and how requirements might be factored in a tiered monitoring approach.

Recommendation for a revised indicator:

1) To differentiate requirements between situations where N & P become limiting or co-limiting using the mass-balance ratio approach proposed in Annex 1.

[16] There was also a suggestion to dispense with P entirely as an indicator of trophic state shift in tropical systems though it was considered that this would forgo its effective use in the many contexts when its application is practicable.

[17] Paerl et al (2015) propose a trophic status index (TSI) incorporating a ratio of N:P.
However, greater Challenges remain regarding ability to collect representative samples of [TN] compared to [TP]

[18] Wetzel, R.G. 2001 Limnology: lake and river ecosystems

2) To maintain the existing 25% and 20%[19] maximum increase from baseline limit for P in smaller and larger water bodies respectively.

3) To maintain the existing disallowance on the trophic shift of any water body under conditions when P is determined to be the limiting factor for eutrophication or when N & P are determined to be co-limiting.

Notes:

- WQ parameters with monitoring requirements across standards beyond those already cited above (TN, TP, Transparency [Secchi disk] and Chl-a) include water conductivity, BOD, ToC, TSS, NH_4 NO_3, and Ortho-P. The TWG considered NH_4 to be a potential *in-situ* indicator of welfare for the cultured stock but not appropriate as a receiving water quality indicator. Conductivity provides a general measure of water quality and lake water-specific conductivity has been used as a proxy for residence time. Similarly, conditions exist where organic carbon inputs can drive eutrophication. However, no clear rationale or metric limits are associated with existing standard requirements for measurement of these two parameters.
- The TWG also questioned the practicality of a requirement to sample DO immediately (0.5m) above the benthos (in the Salmon Standard). Here, it was felt that improved sensor technology could even allow measurement in the benthos. However, requirements should also be cross-checked with Redox measurement requirements in the benthic criterion i.e., a proxy of oxygenation at this interstitial zone.
- Current BOD requirements, limited to the Salmon Standard, require farms to estimate BOD associated with entire production cycles as an IOM indicator (with no performance limit) using a nutrient mass-balance approach[20] (rather than laboratory analysis of samples at specific time-points). Whilst Chl-a provides a direct indication of phytoplankton levels, BOD provides accounts for bacterial metabolism, i.e., it could serve as an 'Impact' indicator of eutrophication impacts in sub-photic zones (the Salmon Standard also refers to BOD monitoring of effluents, but with no specific requirement). A BOD indicator may also have application in more meso-eutrophic/ mixotrophic contexts (see above). However, the need for lengthy and costly laboratory analysis and sample-management constraints are likely to present practical implementation challenges in many settings. The TWG acknowledged practical limitations around laboratory-based analysis of BOD (and Chl-a, the two most direct indicators of eutrophication) in many production settings (advances in sensor technology making the possibility of cost-effective accurate in-situ measurement of Chl-a, a feasible option are not available for BOD).
- Inability to reliably classify and impose limits around trophic state 'tipping-points' in nitrogen limited systems, points to a need for alternative and/or more demanding impact indicators. Nitrogen limitation (or co-limitation with P) is most common in eutrophic conditions with high Ps loadings[21]. Under such conditions, it may be practicable to set limits on the modelled BOD contribution described above. This could be applied at the individual farm level (the recommendation below) and/or landscape level. The latter option is has been appended to the Section 3 (input-output management).

[19] The TWG deliberated whether there should be differential rate-change limitss for N & P (potentially with greater allowance for N change), but decided to adopt a more precautionary position given the lack of detailed nitrogen budgets of mesotrophic lakes, particularly in the tropics.

[20] Based on estimation of the difference between N & C input/ output in feed and fish production over a production cycle

[21] Nitrogen can also be limiting in oligotrophic waters, particularly in mountainous regions or high latitudes where both phosphorus and nitrogen are naturally in short supply.

Recommendation for a revised indicator:

1) A requirement for farm-level monitoring of BOD based on mass-balance estimation approach should be retained; particularly in instances when N is determined to be the limiting nutrient for eutrophication in mesotrophic contexts (farm level measurements based on water quality samples were considered less practicable due to sampling and analytical challenges).

Water body stratification and eutrophication impacts

- The TWG felt strongly that sampling guidance should be developed to require measurement of oxygen and temperature-depth profiles in stratified systems, again with the precise sample design reflecting seasonal and wider system variability.
- Profiling should be extended below the photic zone[22] and to the start of the anoxic zones or depth at which [DO] variability ceases to be significant. Sampling should be conducted in the proximity of aquaculture sites and reference sites over different seasons (e.g., dry/wet – summer/winter). Requirements should also reflect knowledge of the biological activity of different strata in lakes with an oxygenated hypolimnion, e.g., presence of deep-water fish species such as salmonids[23] in temperate systems.
- Limits on change could be applied relative to baseline conditions (see below) with focus on precluding progressive hypoxia and chronic deterioration of the hypolimnion. Monitoring requirements/ intensity should reflect natural seasonal variability and known periods of greater associated vulnerability.
- Further consideration as to if and how depth-profile limits could be meaningfully set is required. Until such time as these limits are established, profiling should be part of a more general requirement for good environmental stewardship requiring evidence of management interventions to limit deteriorating conditions associated with nutrient enrichment e.g., changes in feed quantity/ quality, stocking density reduction, improved waste interception measures etc.

  Rationale: The TWG observed that monitoring of DO/temperature profile trends can be an excellent indicator of longer-term eutrophication impacts in stratified systems – further noting that climate change is pushing all lakes into less mixed i.e., meromictic states. Current guidelines referring to permanent stratification in meromictic lakes do not adequately address more localised stratification conditions in deeper lakes of variable depth, or natural variability of stratification over time. If a water body can be demonstrated to be meromictic over a sufficiently long period, nutrients falling through the active zone to bottom waters/sediments are likely to be sequestered over the longer term. This supports a case for a minimum of 2 years of monitoring of mixing-status (‘meromictic intensity’) prior to certification (subsequent to the proposed 5 year review period). Background checks on longer-term secondary WQ data (where available), HAB and fish-kill events indicative of stratification turn-over events should also be mandated.

Recommendation for a revised indicator:

1) To require measurement of oxygen and temperature-depth profiles in stratified water bodies and that the sample design reflects seasonal and wider system variability. Certified farms should be required to collect data over 5 years following the introduction of this requirement, in order to evaluate potential for setting metric limits and associated management intervention requirements and, potentially additional siting requirements (see Section 1).

[22] If aquaculture is purely in photic zone, e.g., shallower areas, sampling should potentially be extended to deeper areas.
[23] E.g. char or lake trout in N. America

3. 'Inputs' (input-output management)

Standards for fed culture systems set P and N limits per kg of harvest for (i) inputs in feeds and/ or fertilisers based on mass-balance accounting net of removals in harvested fish[24], (ii) release for P (only) based on measurement of effluent concentrations or mass-balance accounting, net of removals in harvested fish and unassimilated P intercepted within the culture system). Species and system-specific requirements consider removals at the following production stages (i) pre-culture; linked to feed quality (fines content) management, (ii) culture; assimilation in harvested fish and (iii) post culture measures; solids settlement and storage systems, biofiltration and adsorption of dissolved P[25]. Results are expressed as TN & TP input and/ or release (discharge) per tonne of harvest over 12 months. Associated requirements set metric limits for input and/ or output (release to RW; TP only).

Input limits are only applied in the Pangasius (cage) and Tilapia (cage and pond) standards[26]. All the freshwater standards go on to set output limits on the total amount of phosphorus that can be released into receiving waters. The lowest input limits are set for the FW trout and salmon smolt farms, with a net release of up to 4 kg/t of fish produced over 12-months. Tilapia farms (cage and ponds) are permitted to release up to 20kg/mt. The Tilapia and Salmon standards provide assumptions regarding the typical P content of farmed species for estimation of P removal in harvested fish. Tilapia are assumed to incorporate 0.75% P as a proportion of live weight.

Mass-balance accounting of nitrogen in effluents is complicated by its highly reactive nature and variable volatilization rates under different environmental conditions. Interacting factors include variability in timing of feeding and pond draining for harvests, organic matter decomposition rates, pond substrate characteristics and lack of any point-source of effluent from cages. Only two ASC standards; for Pangasius and Tilapia include indicators that require monitoring of N input efficiency per mt of fish harvested over 12 months (three standards -- for pangasius, tilapia and shrimp -- go on to set limits on N effluent loads per unit harvest.) No cage standards impose any such release requirements or metric limits. For mass balance calculations, tilapias (only) are assumed to incorporate 2.12% N as a proportion of live weight.

Beyond the feed testing requirements described above, only the Salmon Standard imposes a further assurance step in limiting the fines content of feeds.

3.1. TWG recommendations nutrient 'Inputs' (input-output management)

The TWG discussion for lakes and cages paid particular attention to mass-balance accounting of feed-harvest input-output ratios; 'nutrient-efficiency' in current standards, consistent with the limited ability to introduce other interception measures in open cage systems.

Nutrient use efficiency

- The TWG noted that nutrient efficiency requirements are effectively precautionary in nature and should ideally operate in a more integrated manner in conjunction with receiving water siting (trophic classification) and impact indicators. For example, this could potentially lower limits on IOM indicators in situations where nutrient discharge is unlikely to have any significant eutrophication impact[27].

[24] Framed as 'nutrient-use efficiency'.

[25] Assumptions are also made for adsorption of dissolved P in earthen ponds.

[26] For TP requirements set maxima of 20 kg/mt for pangasius cages and 27 kg/mt for tilapia ponds or cages)

[27] This will merit further consideration when the TWG considers impacts in flowing water or high energy systems. A case could also be made for maintenance of efficiency requirements as precautionary measure linking more generally improvement of husbandry measures with positive impacts on other criteria e.g., marine ingredient use, health management etc).

Dependencies with welfare requirements

Recommendation for a revised indicator:

1) Efficiency metric limits should also be adjusted to account for the life stage as well as culture species.

   Rationale: Juvenile life stages (including trout and salmon smolts) require higher dietary P levels than adult stages. This has been a source of a number of VRs approved on the basis of animal welfare justifications.

Feed quality assurance

- The TWG was supportive of further development of feed quality requirements. Consideration is therefore required as to how these factors are implicitly addressed in the nutrient-efficiency requirements of existing species standards and what if any improvements/ explicit requirements might be made/ added. At a minimum, such quality concerns could be highlighted in standard guidance on root causes of non-compliance and corrective actions (which currently implicitly focus on eFCR gains through improved husbandry as a means to enhance nutrient assimilation in harvests).

  Rationale: Such assurance requirements would provide a strategic way to leverage upstream market entities (e.g., feed manufacturers) to innovate quality improvements to mitigate impacts (e.g., with lower N&P levels but higher digestibly, excluding or minimising effects of anti-nutritional factors, switching from sinking to extruded diets).

Recommendation for a revised indicator:

1) Feed quality efficiency metrics should be extended to require assessment/limits on feed protein digestibility (as the main source of N & P in fish feeds – with particular emphasis on P) along with simple on-farm tests for feed water stability and pellet sink-rates.

WQ data collection and transparency

- This is an overarching issue spanning each of the WQ criteria discussed above. Requirements can be broadly divided into (i) primary data required to demonstrate compliance with current ASC indicator requirements and (ii) background data to underpin future standards development. Both may incorporate baseline and monitoring data.
- The TWG felt certified farms should be required not just to submit their primary baseline and monitoring data to ASC, but to transparently present it in a publicly available platform. Only data linked to prevailing standard requirements and metric limits should be considered.

  Rationale: The TWG believes that such transparency, codified through certification, can support a company's social license to operate and thereby also incentivise participation.

Recommendation for a revised indicator:

1) To require farms to submit to ASC and make publicly available primary baseline and monitoring data.
2) To require farms to submit data to allow mass-balance modelling the sectoral contribution to BOD in mesotrophic contexts (see Section 2). This may also require farms to solicit such data from non-

certified farms or to collaborate on modelling using proxy estimates e.g., based on cage numbers/ area.

Notes:

- Data collection formats should be developed to support more standardised data collection, i.e., including data validation measures -- although a degree of flexibility is needed to accommodate site-specific sample design requirements.
- The TWG recommends collection of background data, not directly linked to prevailing standard requirements, should be based on an *a priori* analytical model, i.e., greater advance consideration should be given as to how it could practicably be used for future standard-setting and data collection formats optimised to this. These formats should ideally also lend themselves to more real-time data visualisation. The TWG noted the desirability of more sophisticated data management approaches (e.g., online relational database management systems etc) but acknowledged that despite their limitations Excel spreadsheets still constitute the most universally adoptable medium.

Area-based management

Area based-management requirements could logically be incorporated in the input-output management criteria – or developed as a separate criterion. This topic will be addressed in greater depth by the TWG as part of the remaining discussions. However, within its current deliberations, the TWG has discussed the following:

- Setting absolute limits on nutrient concentrations in receiving waters effectively incentivises certified farms to actively cooperate with other water users to manage eutrophication[28]. However, the TWG felt that the standard should specifically state that if there are unacceptable impacts from a number of farms, then certified farms have responsibility for collective action and stewardship.
- This merits further development and adoption of area-based management approaches at appropriate 'landscape' scales e.g., around data collection, sharing/ transparency and mitigation feedback loops linked to farm-siting, feed input quality and prudent farm husbandry and management measures.
- The design of such requirements also consider incentives (e.g., cost-sharing, biological performance and strategic opportunities around social licence for large companies/ partner support) and barriers to participation
- The TWG recommends that ABM requirements for WQ management should be coordinated in conjunction with similar ongoing discussions in other standards areas.
- Requirements for sea-lice management in the Salmon Standard were cited as an ABM example in existing standards. The TWG noted a major deficiency around ABM requirements for WQ management in current standards. The TWG also felt this is a highly challenging area particular in fragmented production contexts including many small-scale enterprises. This merit involvement in other areas of specialist expertise/experience in natural resource management (e.g., in consensus building, conflict mitigation, stakeholder/ institutional analysis etc.). Ideally, this would be part of a wider consideration of ABM requirements across multiple aligned standard criteria.

Recommendation for a revised indicator:

[28] This was endorsed by one of the panels observations on cooperation and support provided by an ASC certified farm (a large-scale tilapia cage farmer) to smaller farmers on Lake Toba, Indonesia, in response to eutrophication trends.

1) Additional ABM indicator(s) should be developed requiring certified farms to coordinate feed-input and output requirements with other farms/ water users at an appropriate landscape scale (i.e. linked to siting/ classification requirements under WQ Criteria 1). These could be incorporated within criteria 3 (IOM) or within a dedicated criterion -- potentially covering ABM measures under different standard principles.

Annex 1 Proposal for a (relatively) simple tool for assessing farm impacts on water quality

This document presents a procedure for conducting an auditable, minimally complex assessments of possible **farm impacts on water quality based on predicted increase in surface water nutrient concentrations** associated with increased nutrient loading from aquaculture activities. The assessment procedure is based on published steady state water quality models, data collected elsewhere in the certification process, other publicly available geographic data about the waterbody in which a farm is seeking certification and **the emerging scientific consensus that both nitrogen (N) and phosphorus (P) can limit primary productivity in freshwaters (Paerl et al. 2016).**

It must be stressed that the procedure described here is presented as a possible minimum threshold for an auditable assessment of potential farm impacts on water quality. **The proposal remains neutral as to what criteria should be used to decide on acceptable levels of impact (e.g., percent increase from baseline, trophic status threshold boundaries, etc.)** as these are dealt with elsewhere in the standard. Furthermore, the proposal assumes that **an applicant for certification will have conducted sufficient monitoring of both total N and total P in the candidate waterbody so as to produce meaningful estimates (e.g., geometric mean) of concentrations prior to new farm operations.** Thus, there is no discussion of appropriate water quality sampling strategies.

There is increasing scientific evidence suggesting a need to consider both N and P inputs when assessing eutrophication effects. The P limitation paradigm has historically been a useful guide for lake management but it may not provide a complete picture of possible effects on water quality outside temperate regions in proximity to significant atmospheric N emission sources (i.e., the Tropics, but also increasingly the Sub-Arctic). Many freshwater lakes and reservoirs are N limited or both N and P limited, and additional inputs of either nutrient can lead to deterioration of water quality (Paerl et al. 2016 and references therein).

The most likely limiting nutrient (N or P) for increased primary productivity and subsequent water quality impairment can be determined based on the mass ratios to total N and total P inputs to a water body (N_{In} and P_{In}; Paerl et al. 2016). Based on a large amount of data synthesized by Guilford and Hecky (2000), **Paerl et al. (2016) suggest that for mass ratios of N_{In}:P_{In}≥23, lakes are generally P limited. When N_{In}:P_{In}≤9, lakes are generally N limited. At intermediate N_{In}:P_{In} ratios (between 9 and 23), lakes are both N and P limited.** The actual scheme presented by Paerl et al. (2016) is slightly more complicated as they note that **relative rates of N fixation and denitrification are also important (i.e., when denitrification greatly exceeds fixation, N limitation is more likely). This additional complexity could be incorporated into the protocol. For example, waterbodies with hypolimnetic anoxia are likely to have higher rates of denitrification while any evidence of cyanobacterial blooms would suggest higher rates of N fixation.**

Both the most likely limiting nutrient (N, P or both) and the change in nutrient concentrations can be estimated using published steady state water quality models, **the monitoring needed to assess reference (pre-farm) conditions and assessments of N and P inputs associated with farm operations.**

Steady state water quality models (e.g., Vollenweider, Dillon-Rigler, etc.) typically assume that **the mass of P leaving a lake (P_{Out}; kg/yr) can be estimated based on the mass of P (P_{In} kg/yr) entering the lake and an empirical retention coefficient, r_P** (see, e.g., Brett and Benjamin 2008).

$$P_{Out} = (1 - r_P)P_{In} \quad (1)$$

Similar steady state models predicting outputs as a function of inputs and retention also exist for N (e.g., Harrison et al. 2009):

$$N_{Out} = (1 - r_N)N_{In} \quad (2)$$

Concentrations of nutrients in water leaving a lake ([N_{Out}] and [P_{Out}]; mg/l) can be assumed to be the same as concentrations in the epilimnion ([N_{Epi}] and [P_{Epi}]; mg/l). In turn, **epilimnetic concentrations**

can be assumed to be a reasonable proxy for the average concentration in the water column ($[N_{Lake}]$ and $[P_{Lake}]$; mg/l).

$$[N_{Out}] = [N_{Epi}] = [N_{Lake}] \quad (3)$$

$$[P_{Out}] = [P_{Epi}] = [P_{Lake}] \quad (4)$$

Using the annual flow of water through the waterbody outflow (Q_{Out}; m^3/yr) we can estimate the mass of nutrients leaving the system:

$$P_{Out} = 0.001\ [P_{Out}]\ Q_{Out} \quad (5)$$

$$N_{Out} = 0.001\ [N_{Out}]\ Q_{Out} \quad (6)$$

Annual outflow may either be obtained directly or from annual unit runoff (mm/yr) and catchment area. Typically, such measurements are made by national hydrographic agencies. If measurements are not available, there are a number of publicly accessible global databases that can be used to obtain annual outflow estimates (e.g., Ghiggi et al. 2019, Linke et al. 2019).

Rearranging (1) and (2), using the equivalencies in (3) and (4) and substituting in (5) and (6), it is possible to **estimate present day nutrient inputs based on epilimnetic (measured) water chemistry**:

$$P_{In} = (0.001\ [P_{Out}]\ Q_{Out})/(1 - r_P) \quad (7)$$

$$N_{In} = (0.001\ [N_{Out}]\ Q_{Out})/(1 - r_N) \quad (8)$$

Both r_P (summarized in Brett and Benjamin 2008) and r_N (summarized in Harrison et al. 2008) can be estimated based on annual inflow (Q_{In}; m^3/yr), water body area (A; km^2) and mean depth ($\bar{z}$; m).

Table 2 of Harrison et al. (2008) provides a series of N retention coefficients for lakes and reservoirs in temperate and tropical regions. These are all of the form

$$r_N = 1 - \exp(-v_f/H_l) \quad (9)$$

where v_f (m/yr) is an empirically determined apparent settling velocity and Hl (m/yr) is the hydraulic load determined from A and Q_{In}

$$H_l = Q_{In}/(1000000\ A) \quad (10)$$

Due to evaporation from waterbody surfaces, Q_{In} will almost always be greater than Q_{Out}. **Wang et al. (2018) provide estimates of lake evaporation (E; mm/yr) as a function of latitude that could be implemented as a lookup function and McMahon et al. (2013) provide guidance for modelling evaporation when measurements are not available**.

$$Q_{In} = Q_{Out} + 1000\ EA \quad (11)$$

Brett and Benjamin (2008) present a statistical reassessment of a large published dataset of lake P inputs and output measurements using a range of variants on the Vollenweider approach. They conclude that **the best model simulated P retention coefficients based on mean hydraulic residence time (τ_w; yr) raised to an exponent x= -0.53 and an empirical coefficient, k ($yr^{-0.47}$)**

$$r_P = k\ \tau_w^x/(1 + k\ \tau_w^x) \quad (12)$$

Mean hydraulic residence time (τ_w; yr) can be determined from mean depth and hydraulic load:

$$\tau_w = \bar{z}\ /\ H_l \quad (13)$$

By substituting equations for r_P and r_N into equations (7) and (8), it is possible to estimate nutrient loading to a water body based on current measurements of water quality:

$$P_{In} = (0.001\ [P_{Out}]\ Q_{Out})(1 + k\ \tau_w^x) \quad (14)$$

$$N_{In} = (0.001\ [N_{Out}]\ Q_{Out})\ \exp(v_f/H_l) \qquad (15)$$

Equations (14) and (15) provide simple and auditable estimates of nutrient inputs prior to the start of farm operations, which can then be used with the Paerl et al. (2016) typology to assess whether a system is N limited, P limited or co-limited. The effects of any increased N or P inputs associated with new farm operations can be simulated using equations (1) and (2) with the present-day nutrient inputs provided by equations (14) and (15) and an assessment of farm inputs.

The approach presented here could be readily implemented as a spreadsheet. This would then allow applicants to make their own calculations without needing access to specialized tools or software.

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