5 December 2001

Letter Report of the Independent Scientific Advisory Group Regarding the B.C. Aquaculture Waste Control Regulation

Initial Review Comments on (I) Selection of Protection and Measurement Endpoints and (II) Methods for Establishing Environmentally Protective Thresholds, toward the Sustainable Management of Salmon Aquaculture Wastes

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1. Scope and Intent

The Scientific Advisory Group (SAG) is tasked with providing independent scientific advice to the British Columbia government toward regulations that enable the management of waste discharges, under the British Columbia Waste Management, in a scientifically defensible and environmentally sustainable manner.

The terms of reference for the SAG have been formally defined[1]. In general terms, the SAG has been asked by WLAP to "review and advise this Ministry on development of a new Aquaculture Waste Control Regulation". SAG's review will include the draft methods of analysis for a performance-based assessment of waste discharge, the proposed draft regulation itself, and stakeholder comments on the draft reg., including methodology. The scope of the SAG review is limited to marine and estuarine systems. It is understood that parallel activities will take place to address freshwater aquaculture operations.

This brief report, the first submission of the SAG, primarily provides preliminary comments on the draft methods of analysis that have been discussed. The methods aim to measure waste redistribution and possible biological impacts, and ensure that waste discharges from British Columbia coastal salmon aquaculture operations do not result in either poorly appreciated or societally unacceptable environmental costs relative to their known socioeconomic benefits.

Given the complexity and intensity of interest in salmon aquaculture, the SAG takes this opportunity to define the scope of our own deliberations and comments. First, the SAG has not been engaged to undertake a socioeconomic analysis of, nor provide policy directives on levels of environmental protection that balance economic needs and the broader interests of the general public in environmental protection, avoidance of resource conflicts, and overall environmental sustainability. Such decisions will need to be undertaken by senior MAFF and WLAP managers and their Ministers, but scientific knowledge is only one facet of the underlying issues.

Second, the SAG explicitly understands that some issues with regard to sustainable

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[1] "Terms of Reference for an Independent Scientific Advisory Group Regarding the Aquaculture Waste Control Regulation" WLAP, November, 2001.

coastal aquaculture (for finfish and shellfisheries) are amenable to being addressed under the Waste Management Act (and even share similarities with other types of organic waste discharges), while others are much better addressed using other management tools. We understand that separate specific regulations will be enacted to deal with escapements, and fish health; and that other initiatives are underway to facilitate relocations of currently poorly located sites, or to encourage the development and use of alternative technologies.

With regard to substances released from aquacultural operations, the SAG offers the view that the approaches being entertained and that seem feasible as part of a regulation under the Waste Management Act theoretically have the potential to address waste-related impacts of aquaculture on a site-by-site basis. The regulation, however, would lack the structure, the focused interest in, and the power to detect, monitor, and manage cumulative effects of multiple aquacultural operations in a larger defined coastal ecosystem. We were unable to envision mechanisms that would allow a performance-based, practical regulation aimed at individual operators that could address cumulative ecosystem effects of many aquaculture operations. This is an especially important issue if the geographic density of operations is likely to increase in the future.

The overall issue can be re-phrased as a question:

"Since the management of wastes from aquaculture relies on the assimilative capacity of the surrounding environment, how many operations can be sustained within a given area without adverse effects that extent beyond the actual footprint of the aquaculture site?"

The simple reality is that no one in the private sector, regulatory, or scientific community can answer this simple question at present with either certainty or objectivity. The SAG does not support the view that lack of scientific certainty should impede economic activities, since this occurs for any process or activity. However, allowance for uncertainties must be included in any realistic regulatory framework.

Overall , the SAG supports the development and implementation of an aquaculture waste control regulation under the British Columbia Waste Management Act PROVIDED that the government and industry are committed to an examination of the cumulative effects issues through other management tools and approaches.

The regulation is likely to be ill-equipped to address loss of assimilative capacity for the ecosystem based on the cumulative effects of many operations, but at least two major strategies could be employed to address this. The first, as discussed above, is to undertake the appropriate level of targeted investigation outside of the new regulation. The second is to ensure that the assimilative capacity in the environment around each operation is never exceeded, given our uncertainty about the larger implications of such exceedance for wastes and other effects. The latter approach is perhaps overly conservative, but justified if no level of effort is directed toward the first approach.

Waste management is only one aspect of achieving sustainable aquaculture. The SAG, therefore, hopes that the Provincial government will take a holistic view and examine interactive effects induced for example by the nature of feed and feeding practices. It is important to understand not only what is contributed via the food to the benthos, but what happens with time in terms of different feedstock materials.

SAG members briefly discussed the concept of maximum allowable loading for various substances (organic carbon, nutrients, metals) over a given time period as a regulatory endpoint, rather than the use of instantaneous chemical concentrations. The use of Total Maximum Daily Loads (TMDLs) has recently been adopted in the United States and some European countries. Such an approach takes a broader ecosystemic view of materials flux.

We understand that building in the concept of maximum allowable loading is too complex at this point, given the tight timeline for the development and adoption of an aquaculture waste management control regulation. [2]*. We nonetheless see this as a potentially useful approach for subsequent revisions of the regulation*, and briefly return to the concept in later sections.

Because of this issue and others discussed below (especially around the adequacy of current data to support the confident derivation of performance-based standards), *the SAG recommends that the triggers contained in the aquaculture waste control act, and the act itself, be introduced as an* ***interim*** *regulation. We recommend that the regulation be re-visited within three years following its first introduction and that it be based securely on the best scientific information and understanding available at the time.*

The limited efforts of SAG members are focused on ecological endpoints, sediment chemistry and diagenesis[3], and on appropriate measurement endpoints that do justice to the known science on the loading of organic matter and other substances to the seabed and associated biological communities. A regulation that requires operators to undertake potentially costly environmental monitoring, but does not provide information that allows them to make management adjustments internally (in the spirit of a pollution prevention or environmental management system) serves neither the interests of the regulated community or the public.

SAG recognizes the desire by MAFF, WLAP, and the current government to create and manage regulations that are simple and not onerous for either the regulated party or the regulator. We anticipate, however, that some very hard policy decisions will have to be

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[2] We note, however, comments by Brooks (2001, p. 3) that suggest a theoretical maximum *aerobic* carbon degradation rate of 4.0 g C/m2/day in low velocity areas (<0.1 cm/s) to a maximum of 22 gC/m2/day at velocities greater than 10-12 cm/s, based on Findlay and Watling (1994). Brooks (2001) estimated the average current velocities at B.C. salmon farms were in the range of 3.5 to 9 cm/s, with 2 h minimum surface speeds < 3 cm/s.

[3] **diagenesis [sedimentary]** *(di-a-gen'-e-sis)* All the chemical, physical, and biologic changes undergone by a sediment after it's initial deposition, and during and after its lithification, exclusive of surficial alteration (weathering) and metamorphism.

made about appropriate levels of environmental protection in the receiving environment around aquaculture operations. As a general rule, where there is uncertainty about the broader implications of some level of environmental response, both regulators and the public can tolerate higher potential environmental risk provided that there are subsequent tiers of observation and response that seek to better understand and address such risks. A simple YES – NO type dichotomization of potentially complex environmental issues (for example; compliant – non-compliant) does not set the stage for either the direction of future efforts to address issues or develop any consensus where there is a lot of grey area in the middle. This argues for a regulation that involves tiered triggers and responses, an approach that would be more congruent with self-regulation under an ISO14001 or other type of environmental management system.

In concrete terms, if the regulation is distilled down to sampling and analysis, self-reporting, and receipt of a simple pass-fail designation, the conditions for conflict will have been maximized. The regulated community will argue for an erosion of policy-based environmental protection levels, since the costs of failure are high. Any policy response to such pressure often erodes the scientific defensibility of the policy decisions and will be seen to undermine sustainability, even for the industry itself. Conversely, members of the public will see a passing grade as an abdication of any further need to address real and/or perceived environmental risks (an abdication of responsibility). Since the only chance to address such concerns is through failure, they will advocate for very highly protective and conservative endpoints indeed. The aquaculture issue certainly does not need any more polarization than is already evident.

One final note: Marine aquaculture falls squarely under the regulatory jurisdiction of both the provincial government (for example, under the Waste Management Act) and the federal government (for example, under the Federal Fisheries Act). There has been much discussion in recent years about the harmonization of regulations in areas where jurisdictions strongly overlap. There should be a desire by all to create the conditions wherein the aquaculture industry response to regulation serves the interests of all jurisdictions, without additional financial and other effort. Any redundancy or lack of coordination between the regulatory requirements for different jurisdictions should be avoided.

Federal Fisheries Minister Herb Dhaliwal publicly committed Canada recently to an endorsement of new recommendations that amend and would make more stringent controls on marine discharges from land-based activities that contribute sewage, sediment, and various other waste materials to the ocean. This is under the enabling framework of the 1995 United Nations "Global Program of Action for the Protection of the Marine Environment from Land-Based Activities".

The SAG therefore takes the view that a provincial aquaculture waste management regulation must be constructed with strong consideration for both emerging national and international trends with regard to the marine environment and environmentally sustainable economic development.

Having addressed the broader issues, the SAG now turns its attention to more concrete technical/scientific issues. The remainder of our initial comments are structured as follows:

- Section 2 provides a brief overview of our conceptual understanding of the issues around waste release from a coastal aquacultural operation;

- Based on this, Section 3 addresses whether the right monitoring endpoints have been identified;

- Section 4 provides SAG comments on sampling and chemical analysis methods that would maximize the interpretability of environmental data collected under the regulation.

- Section 5 discusses aspects of field sampling design, such as appropriate level of effort seasonally and spatially, and statistical consideration.

2. A Theoretical/Conceptual Model for Waste Releases to the Nearshore Environment

Brooks (2001)[4] undertook targeted investigations of the management of marine salmon aquaculture wastes based on surrogate measures in the underlying sediment. These included dissolved sulphide concentration, estimation of the oxidation-reduction potential (ORP), and assays of total volatile solids (TVS). These three parameters are easily measured in the field . Sulphides and ORP can be measured using hand-held electrodes (or colorimetrically) and TVS is routinely measured as percent of dry weight loss-on-ignition (LOI) at temperatures that oxidize the vast majority of organic matter present, but do not result is loss of mineral phases such as carbonates.

The rationale for the focus on these chemical surrogates is that excessive organic carbon additions to the seabed results in a much greater availability of easily degradable organic matter for sediment-dwelling heterotrophic bacteria. These bacteria increase their secondary productivity according to the extent of organic loading, and in doing so consume from the surrounding sediment pore water oxygen and other substances that can be used as terminal electron acceptors during cellular respiration. This in turn results in secondary effects on sediment and water chemistry, many of which can be deleterious to macroscopic fauna and flora living in or in close proximity to the seabed. These are discussed in the following section.

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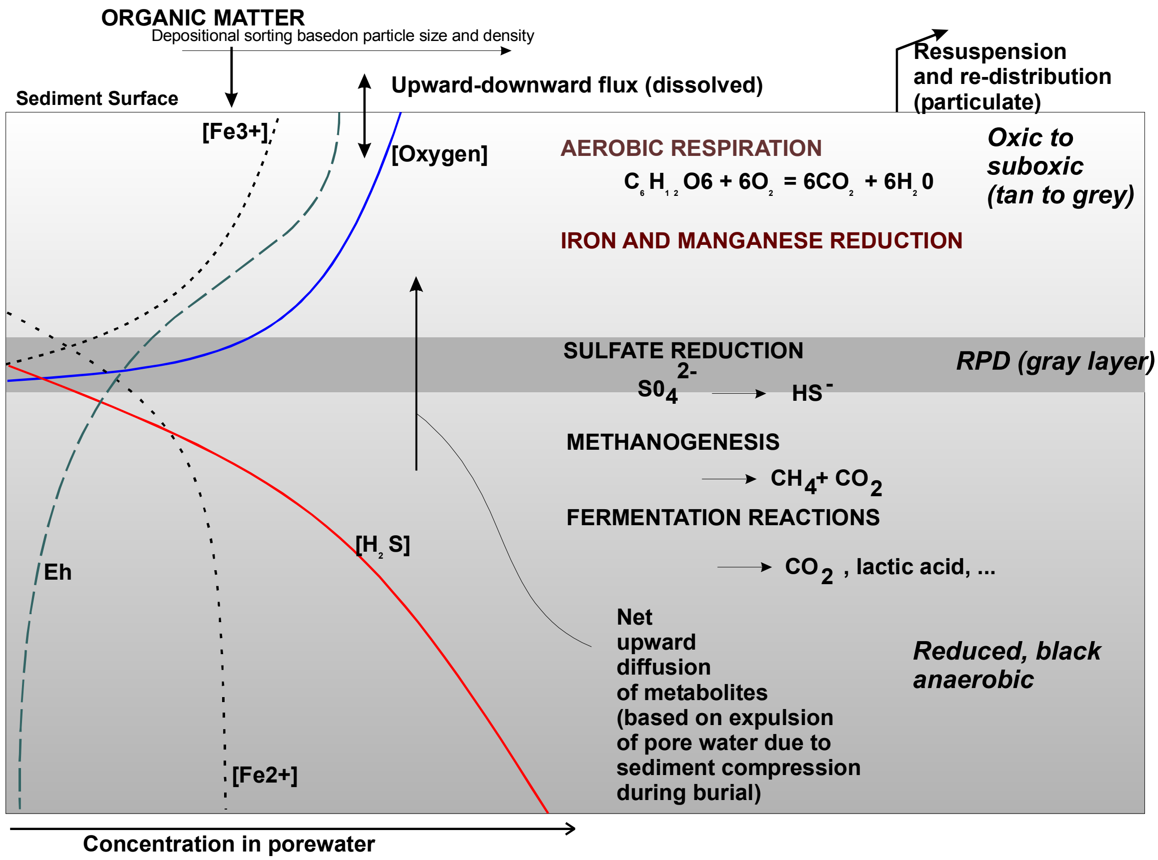
[4] Brooks, K.M., 2001. An evaluation of the relationship between salmon farm biomass, organic inputs to sediments, physicochemical changes associated with those inputs and the infaunal response – with emphasis on total sediment sulphides, total volatile solids, and oxidation reduction potential as surrogate endpoints for biological monitoring. Final Report to the Technical Advisory Group, Care of the British Columbia Ministry of Environment. 210 pp.

2.1 Anthropogenic perturbations of recent sedimentary environments based on heterotrophic microbial metabolism of organic carbon.

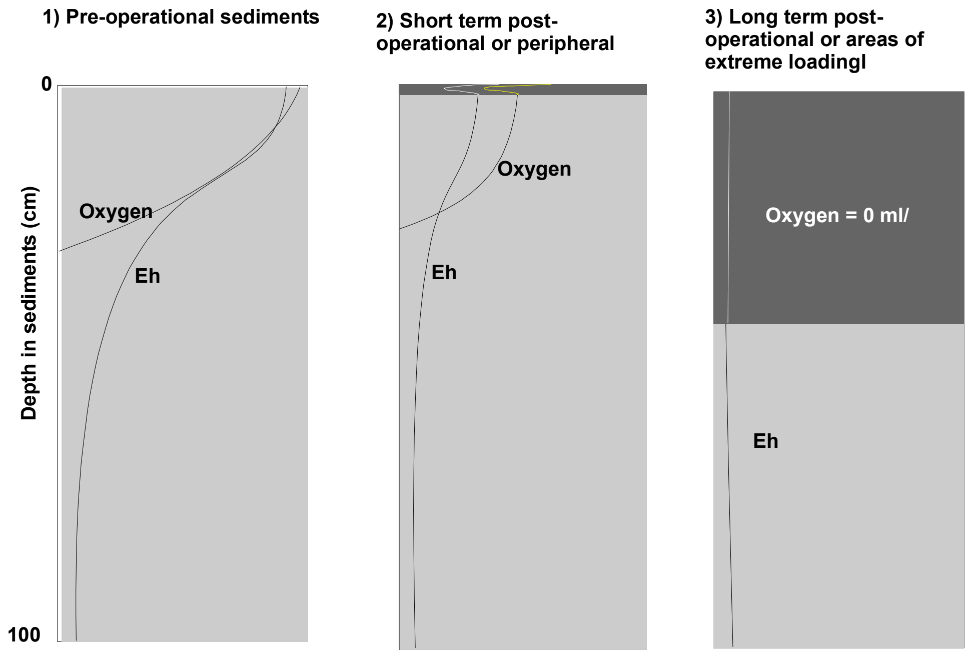


In natural marine sediments unaffected by human inputs of organic matter, there is generally an oxic to sub-oxic zone which may vary from a few millimeters thick in most coastal areas to tens of centimeters in thickness in deeper-water deposits on continental slopes. This zone is exceedingly important for the maintenance of infaunal macro-invertebrates and meiofauna, as oxygen is required for their respiration. The flux of degradable organic matter arriving at the sea floor along with the oxygen concentration in bottom water jointly control the sub-surface depth at which the dissolved oxygen content falls to zero.

When easily-degradable organic carbon of anthropogenic origin is added to sediments, increased heterotrophic bacterial activity results and this depletes oxygen much more rapidly in the surface layers of the deposits. When the rate of depletion exceeds the rate of re-supply via downward diffusion from the overlying bottom water, local loss of oxygen results. This has potential adverse environmental consequences. Degradable organic matter represents an attractive energy-yielding substrate for bacterial communities, so that even in the absence of oxygen, decomposition continues. Different substrates are used as oxidants following oxygen depletion, including dissolved nitrate, solid-phase manganese and iron oxides, and sulphate. The various reactions involved give rise to a series of zones organized by depth, where the aerobic zone (with O_2 present) is uppermost (at and just below the sediment-water interface). The nitrate reduction zone underlies the aerobic zone, and is in turn underlain by the manganese and iron oxide reduction zones, and finally the sulphate reduction zone. If the oxidant demand imposed on the sediments is extremely high, sulphate can be completely depleted, and organic matter will be decomposed by fermentation/disproportionation reactions where a principal product is methane. This zonation scheme is seen worldwide in natural sediments. In most natural coastal deposits, sulphate reduction commences at depths of a few centimetres or so. Methane production however is relatively rare because sulphate is an abundant ion in seawater and represents a large pool of "potential oxidant". Only where the organic load is very high is sulphate completed depleted from sedimentary pore waters within, say, a few 10s of centimetres of the sediment-water interface. Such a circumstance appears to be quite common in the sediments underlying salmon net pens.

A significant concern that results from imposed organic loading and progressive oxidant consumption is the production of noxious byproducts. Among these, two in particular are important: dissolved sulphide (H_2S), and ammonia. Ammonia is produced from the degradation of the nitrogen-bearing fraction of the substrate, whereas H_2S is a byproduct of the bacterially-mediated reduction of sulphate. Such products will diffuse from their zones of production toward regions of lower concentration. For this reason, the direction of diffusion is upward in most sediments. Because H_2S is toxic to most organisms, its production will be problematic where the sulphate reduction zone is near the sediment-water interface. In such a case, communities of burrowing infauna will be inhibited. Infauna may be similarly impacted if the upward diffusive flux of ammonia is large, as ammonium ion (the principal form of ammonia in seawater) can be toxic to organisms at high concentrations. When either the organic input is low or the lateral flux of dissolved oxygen to a given site is high, an aerobic zone is typically present and impacts on benthic fauna from noxious metabolic products like H_2S are minor. In such cases, the faunal communities are limited by food supply more than biochemical effects.

This overview leads to consideration of the following scenarios:



Episodic but limited loadings of organic matter would be expected to result in only a slight decline in the dissolved oxygen concentration of sedimentary pore waters. Under such circumstances, a diverse and 'healthy' benthic fauna assemblage could be expected. Longer term and massive loadings however could result in severe oxygen consumption and a sharp decline in the redox , as well as the presence of free sulphides in pore waters throughout the upper sediment strata. In such a case, there would be a high potential for impairment of the benthic community.

There is an apparent assumption that toxicity to macrobenthos as driven by excessive organic carbon deposition rates is due to sulphide toxicity and/or anoxia. There are a number of well-documented and poorly recognized metabolites of anaerobic decomposition that could adversely affect macrofauna and meiofauna, however. Among these are ammonia ion (and ammonium), carbon dioxide, methane, acetate, and fermentation products such as ethanol and formic acid.

With respect to net-pen farming, two major issues follow from this discussion. First, what is the ability of sediments underlying the pens to recover following the cessation of deposition? Second, what are the decomposition and oxidant consumption rates in proximal peripheral deposits, and do these allow benthic communities to persist in annuli around the pens?

Brooks (2001) concluded that the environmental assimilative capacity for TVS under typical bottom-current regimes in British Columbia is probably exceeded by rates of input of organic matter under net pens. This should give rise to shallow sub-surface anoxia and sulphate reduction. There appears to be considerably less potential for organic matter accumulation in areas peripheral to net pens (or sites of waste discharge) but this appears to be variable across sites and with distance. This conclusion was based on current-related estimates of *aerobic* degradation rates of non-refractory organic carbon provided by Findlay and Watling (1994). Under suboxic or anoxic conditions, decomposition rates for freshly-deposited organic matter in coastal deposits are as high, and sometimes even slightly higher (Canfield *et al*., 1989)[5], providing that sulphate is not depleted.

The quantitative depletion of sulphate from pore waters has an important negative feedback on decomposition rates — these will decline markedly where disproportionation becomes the primary decomposition pathway in the absence of sulphate.

Overall, this suggests a non-linear dynamic, wherein increased organic matter deposition is likely to further impair decomposition rates at some key loading rate, once sulphate is quantitatively depleted within the shallow depositional zone.

The conclusions by Brooks (2001) (see footnote 2) appear to be at odds with studies by Hargrave (1994), who investigated the relationship between sedimentation of organic carbon and a benthic enrichment index using data from fish farms and several embayments on the east coast of Canada. He concluded that when the sedimentation rate to benthic habitats exceeds 1 g C m^{-2} d^{-1} (grams of Carbon per square meter per day), enrichment effects begin to be imposed on organisms.

The depth of accumulation of organic matter that is freely accessible to heterotrophic microbial breakdown is likely to be a key determinant in both the potential for longer term recovery under a net pen and the immediate toxic effects of various heterotrophic microbial metabolites such as ammonia and sulphide.

Brooks (2001 – Figure 11) showed how the Redox Potential Discontinuity (RDP – the depth below which anoxic sediments occur) varied relative to the seabed surface away from the net pen at a poorly flushed, deep basin example site. The RDP occurred at a depth of around 5 cm at distances of ~40 m. or greater from the operation, while it decreased to zero at the edge of the net pen. These observations are consistent with the notion that the RDP depth has the potential to be an excellent surrogate of organic enrichment.

From this simple review emerge a few key questions:

Can a simple measure of chemistry in the top 2 cm of sediment adequately differentiate between risks from transient surficial loading and longer- term build-up?

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5 Canfield, D.E. Sulfate reduction and oxic respiration in marine sediments: Implications for organic carbon preservation in euxinic environments, *DSR*, *36*, 121-138, 1989.

At what point spatially and temporally would we expect to reach some critical threshold of build-up such that aerobic degradation rates become less dominant. Is the chemical monitoring regime up to the task?

How should sampling be conducted relative to annually variable loading rates of organic matter based an a typical aquacultural grow-out cycle?

2.2 Two-way Interactions Benthic Macroinvertebrate Communities and Sediment Chemistry

Sediment chemistry does not simply affect macrobenthos. Infaunal macrobenthos fundamentally alter the sediments in which they live by burrowing into, irrigating, re-working and stabilizing sediments. Rhoads and Boyer (1982)[6] provide a very useful conceptual model for successional sequences among marine infaunal benthos, which has major implications for the deposition of aquaculture wastes. This model is consistent with that of Pearson and Rosenberg (1978),[7] which is often used explicitly, or sometimes insidiously (with limited depth of understanding) in interpretations of marine benthic responses to stressors.

Either contaminants or physical and chemical stressors such as excessive organic carbon loading have the potential to eliminate later successional stage species. Such species tend to exhibit a higher individual biomass (and therefore tend to be more important food sources for higher order consumers such as bottom fish, seals, river otters, or diving birds) and tend to burrow deeper in the sediments, thus enhancing the downward penetration of oxygen-rich water from the overlying water column. Their role, therefore, in creating more aerobic conditions in sediments, to depths that extent locally well beyond a few millimeters, has a direct feedback effect on the sediment bacterial community, with a decrease in the amount of organic matter degradation through sulfate reduction, methanogenesis, and fermentation-type reactions relative to aerobic and iron- or manganese-reduction type metabolism. Sulphide

As in terrestrial communities, the presence of earlier successional species is often a pre-requisite for the later colonization of later successional species. The increase in the burrowing depth with succession is a clear example of this, since burrowing organisms progressively increase the depth of penetration of oxygenated water from the water column and dilution of potentially toxic microbial metabolites, enabling the recruitment into the community of progressively deeper burrowing, sulphide-intolerant taxa.

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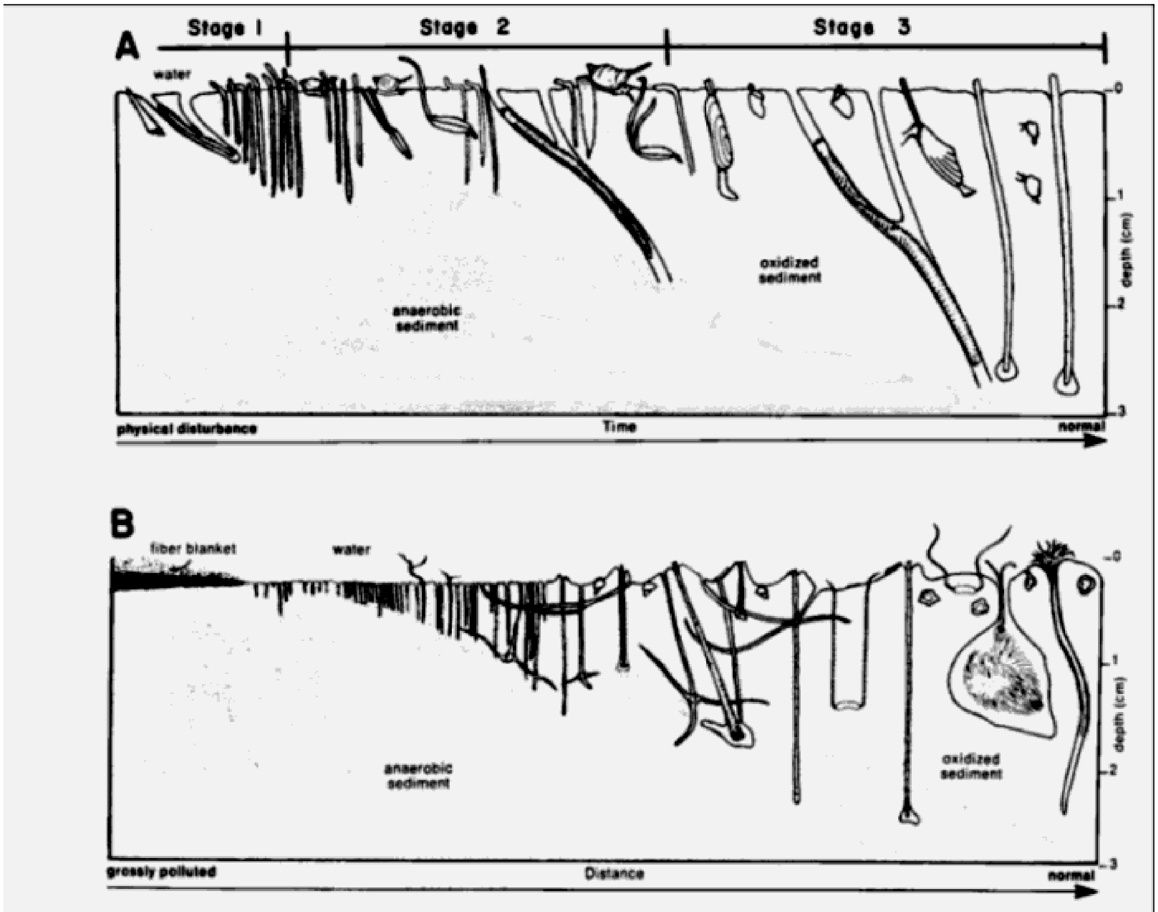
[6] Rhoads, D.C. & L.F. Boyer. 1982. The effects of marine benthos on physical properties of sediments: a successional perspective. pp 3-52 *in:* P.L. McCall & M.J.S. Tevesz, eds., Animal-Sediment Relations. Plenum Publishing, NY

[7] Pearson, T.H. & R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography Marine Biology Annual Review 16: 229-311.

The presence of late successional species can also exclude the presence of other species and inhibit recruitment to the sediment, through interference-type competition such as massive sediment reworking (e.g. for heart urchins *Brisaster latifolia* or sand dollars *Dendraster*) or filter-feeding of larvae from the water column, which would otherwise be recruited into the area.



Model of soft-bottom community responses to environmental stressors (top – physical disturbance and diagenesis; bottom – pollution induced stress). Stage 1 represents early successional communities following disturbance or communities in polluted habitats. Stage 3 communities represent successional endpoints or communities in non-polluted and more stable soft-bottom coastal habitats. Stage 2 are intermediate in both scenarios (from Rhoads and Boyer, 1982). Such successional sequences are evident both over time (following a limited duration event) and over space (away from the stressors).

The overall point of this discussion is that –

i) Later successional communities tend to have more value in foodwebs and overall ecological functioning than primary successional communities dominated by bacterial productivity and very small sized polychaetes such as the *Capitella* spp. complex or a few other organisms (several *Nephthys* spp. are other examples).

ii) Successional sequences in coastal infaunal environments in response to organic or contaminant loading are likely to involve non-linear, chaos type dynamics, where loss of some later successional species has the potential to provide a positive feedback that further exacerbates sediment pore water concentrations of facultative and obligate anaerobic microbial metabolites, including free sulphides.

Research by Warwick (1986)[8] and others demonstrated than benthic community impairment involves a loss of larger, higher biomass individuals and species (which tend to be longer lived) in favour of very small, shorter lived biota. Unfortunately, Warwick's technique involving "K-dominance curves" has not been widely adopted in the routine evaluation of benthic community impairment because it requires both the detailed sorting and enumeration of taxonomic groups and the determination of their biomass.

Brooks (2001) correctly describes hypothetical mechanisms that potentially can scavenge free sulphides from sediment pore water and/or otherwise render them less toxic to intolerant infauna and epifauna. His discussion, however, fails to address the conditions under which hypothetically communities can be driven over the short or longer term to more anoxic, earlier successional conditions.

We argue that the specific identity and physiological ecology of macrobenthic taxa is both important ecologically and useful for understanding the implications of anthropogenic activities. Neither total abundance nor measures of community biodiversity alone can account for the important, ecologically relevant changes in biological functioning.

For example, the presence of the bivalves *Alvania* sp., *Lucinoma annulata* and *Parvilucina tenuisculpta* found within 30 m of the majority of the net pens that Brooks (2001) studied is not surprising, since these clams often contain sulphur-oxidizing bacterial symbionts within their gills. They are well adapted to living in sulphide rich environments, although they can live under more oxic conditions as well – especially *P. tenuisculpta.* The observation of lucinacean bivalves either in terms of contribution to taxon richness or biomass cannot reasonably be considered as contributing to a less, as opposed to a more, perturbed community.

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[8] Warwick, R.M. 1986. A new way for detecting pollution effects on marine macrobenthic communities. Marine Biology 92:557-562.

From a policy perspective, and prior to envisioning measurement and regulatory endpoints, we ask the following questions:

Is there an explicit acceptance of the need for continued maintenance of the benthic infaunal community in an early successional sequence either below the net pens, in the nearfield environment, or beyond the lease?

If so, then this message should be communicated unambiguously and transparently to the public, as a matter of public record.

If not, then the regulatory measures considered, including species richness, total abundance, or chemical surrogates need to reflect a better understanding of community dynamics and the greater importance of some species from a successional perspective.

2.3 Deposition to Hard-substrate and Transitional Environments (such as Shell-Gravel Bottom Types)

The preceding comments relate to relatively uniform soft-bottom settings, and virtually all of the theoretical underpinnings and assumptions about the structuring of bottom communities (e.g. based on Rhoads and Boyer, 1982; Pearson and Rosenberg, 1978; Warwick, 1986) are based on such environments. Many of British Columbia's current salmon aquaculture operations or oyster grow-out rafts occur in areas where the instantaneous or average current velocities, and/or slope of the seabed, limit the net deposition of fine sediments.

Where current velocities or bottom slopes are high and fine-grained organic-rich detritus does not accumulate, the SAG does not envision that organic carbon loading will impose primary negative impacts on the benthic communities via the mechanisms discussed above. Rather, we hypothesize that changes might occur based on –

- Excessive accumulation or flow-through of suspended and ephemerally settling particulates, even for short durations, sufficient to clog filter feeding apparatus and/or otherwise overwhelm respiratory epithelia;

- Direct or indirect effects of specific toxicants in aquaculture wastes (other than organic carbon and heterotrophic microbial metabolites) such as metals, antibiotics (e.g. oxytetracycline), antifoulants (Flexgard XI, Flexgard VI and Aqua Net, which are copper and cuprous oxide formulations), sea-lice control agents (which might be used in the future, especially given recent events in Broughton Pensinsula), or other pharmaceuticals.

The SAG supports only as an interim measure the use of video-monitoring bottom-community census techniques as a simple way of detecting gross changes over space or *time in hard-bottom and shell-gravel communities where attached epifauna and epiflora are major community components. The video observations should also allow us to detect when the deposition of sediments begins to convert a primarily hard-substrate community to a more soft bottom community, a point at which many of the issues with regard to sulphide toxicity, anoxia and low redox would additionally apply.*

We recognize some limitations to this approach, which should be addressed when the interim regulation is re-visited three years after its adoption. First, such techniques will not address possible effects on non-attached, more mobile epifauna such as octopus, kelp greenling or rockfish populations. This might be especially important given the low intrinsic rate of population increase of many of our coastal bottomfish species, their tendency to adopt a territorial range which could create extremes of exposure where the size of the range is no larger than the size of the waste-affected area, and their longevity (several rock fish species can live to be more than 100 years old[9]).

Second, video observations tend to record conditions at the surface (on top of rocks and bottom growth) while much of the biological productivity occurs lower down in the three dimensional structure and is not easily observed. The void spaces in cobble and reef-like structures are likely to be important micro-environments that are especially vulnerable to deposition of aquaculture wastes and smothering. Such effects might not be evident in primarily filter-feeding or macrophytic larger organisms that project upward into the water column, since these organisms will be influenced more by pelagic water column conditions than by conditions in the very thin benthic boundary layer.

Third, abundance or relative density-type enumeration of community structure in non-depositional environments is difficult using video records, which are more amenable to presence-absence type observations. Presence-absence data are theoretically less powerful for detecting environmental change, especially where much of the biota are likely to be hidden from clear view, especially in reference communities.

Finally, we expect that specific toxicants, if present, could result in loss of more sensitive species and the increasing dominance of more tolerant species, since there might be a lessening of competitive pressure for space, less predatory or consumption pressure, and even a greater availability of food source (as organic carbon) for some species but not others – depending on their feeding ecology. Some species, therefore, are likely to serve as good sentinels of increased spatial or temporal scale of the influence of discharged wastes, either through their decline or proliferation. Unfortunately, the current scientific knowledge is too weak to allow us to know what the important sentinel species might be. Possible candidates would be soft and hard corals, especially those that contain symbiotic zooxanthellae, and sponges – which are likely to have little tolerance to clogging of pores from excessive sedimentation. The SAG hopes that a three-year interim period of video

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[9] DFO Pacific Region. DFO Science, Stock Status Report A6-15 (1999). Rougheye Rockfish – British Columbia Coast. http://www.pac.dfo-mpo.gc.ca/sci/psarc/SSRs/Gorundfish/A6-15rough1.pdf

data acquisition will allow us to begin to identify concretely specific suites of epifaunal sentinel species.

3. Data Needs for the Field Assessment of Impacts.

3.1 Redox-related community responses – E_h and S^{2-}

The SAG conditionally supports the use of sulphide concentrations as a chemical surrogate that correlates well with the tendency of benthic infaunal communities to exist in early (Stage 1) successional stages. It is important that sulphide measurements accurately capture the real pore water conditions of the receiving environment (including vertical extent of sediment effects), however, and we reserve the opportunity to comment on appropriate measurement techniques in the future.

There are a wide variety of other chemical measurements that correlate well with redox conditions, such as dissolved manganese and iron concentrations in porewater. Collection of unoxidized samples and their preservation and analysis for these metals is not trivial however. Anaerobic pore waters must be collected and processed under an anoxic environment (typically using a nitrogen-filled glove bag), and we do not feel that such an approach would be appropriate for net-pen staff. Further, Brooks (2001) and BC WLAP staff have noted that TVS concentrations and redox measurements (Eh) using an electrode tend to exhibit significant within-site variability, often greater than that seen for sulphide measurements made on grab samples using electrodes. Measurements of redox potential in sediments are particularly untrustworthy and we do not advocate that these be used on a routine basis.

Sulphide concentrations in shallow subsurface pore waters should be one of a few primary chemical surrogates used to detect and avoid benthic community impairment attributable to excessive microbial sulfate reduction in sediments that underlie net-pens. The measures need to accurately reflect ambient conditions with a minimum of sampling and analytical artifacts[10]*. Such artifacts are easily created when reduced sediments are crudely manipulated in highly oxygenated environments during and after collection. Sulphide should not be measured using an ion-selective electrode, since the sensitivity of these rapidly decreases during storage and use, and since such electrodes tend to drift, resulting in challenges in obtaining stable readings.*

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[10] Carr, R.S., and D.C. Chapman. 1995. Comparison of methods for conducting marine and estuarine sediment porewater toxicity tests-extraction, storage, and handling techniques. *Arch. Environ. Contam. Toxicol*. 28: 69-77.

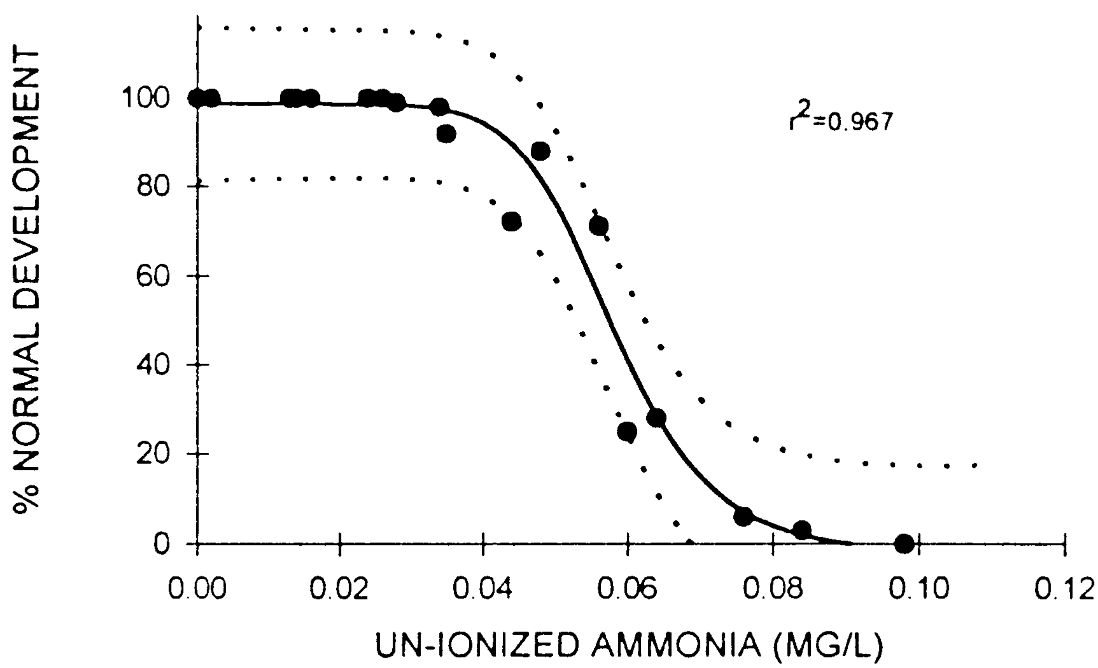
Eh too easily reflects poor electrode performance both TVS and Eh suffer from microscale sampling bias. These should not be the primary determinants of regulatory compliance.

We suggest that WLAP and interested parties give more thought to measures that better define the extent of waste materials build-up (i.e. depth of waste blanket and associated diffusive downward migration) as a very important aspect of biological effects. As noted earlier, the RPD depth – measurable in core profiles, is a potentially useful surrogate.

3.2 What's missing?

Ammonia as a major toxicant

We noted in Section 2.1 that sulphide is not the only microbial metabolite of organic matter decomposition in aquatic environments that can result in toxicity to macrobenthos. In fact, ammonia has been widely recognized to drive toxicity in sediment toxicity tests, and sometimes obscure an evaluation of other toxicants such as anthropogenic metals.[11]

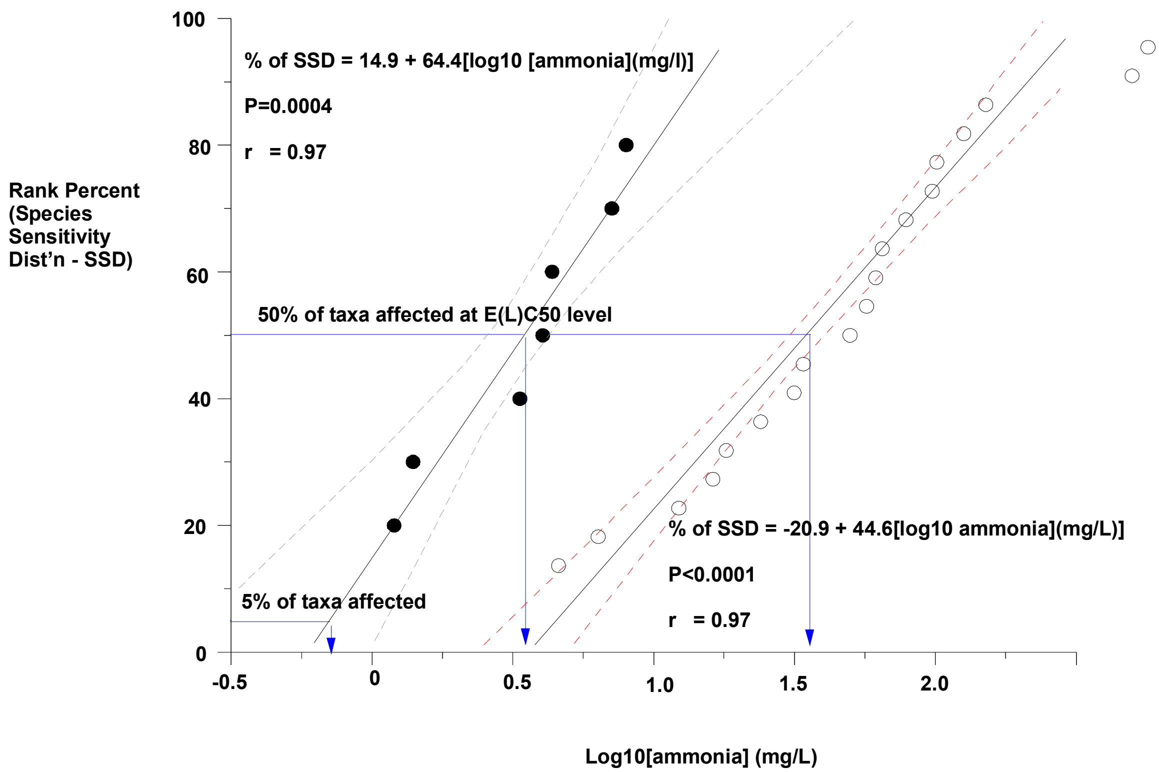


Response of sea urchins, *Strongylocentrotus purpuratus*, to ammonia (from Greenstein *et al.*, 1995)

[11] Greenstein, Darrin J., Said Alzadjali, Steven M. Bay, 1995. Toxicity of Ammonia to Purple Sea Urchin (*Strongylocentrotus purpuratus*) Embryos. *In* SCCWRP 1994-95 Annual Report. (http://www.sccwrp.org/pubs/annrpt/94-95/art-07.htm)

Other potentially high concentration metabolites underneath or adjacent to aquacultural operations could include methane and carbon dioxide. Unfortunately, little is known about these two. A scan of the "Aquire" aquatic ecotoxicity database did not render any records of published information on the toxicity of methane to aquatic organisms. Recent focus studies in British Columbia have not included information on ammonia or other metabolites, and the existing data therefore may be inadequate to define the relationships between aquaculture waste deposition, ammonia concentrations in sediment pore water, and biological effects.

A quick scan of the "Acquire" database, which summarizes aquatic toxicity studies published primarily in the peer-reviewed literature, shows considerable existing information on the toxicity of ammonia to marine aquatic organisms (an even larger data set for freshwater biota was excluded from further analysis). The following figures (next page) provide a graphical summary of the literature-derived toxicity data.



Each point on the graphs represents a toxicity endpoint from the literature, and the data were screened to include only EC_{50} and LC_{50} endpoints. An EC_{50} level is the ammonia "effects" concentration at which there was deemed to be an ecologically relevant, but non-lethal response (for example growth or reproduction) equal to a 50% impairment relative to controls (mostly in laboratory-based toxicity tests). An LC_{50} is the toxicant exposure level at which 50% mortality was noted at a specified time interval (interpolated concentration at which 50% of the test organisms died). The data are summarized as an

annex to our report. The graphs summarize this tabulated data. Each toxicity endpoint is ranked from lowest to highest (ranked from observed concentration at which most sensitive species responded adversely to the observed $E(L)C_{50}$ concentration for the least sensitive species). The percent rank was then plotted against the actual estimated ammonia concentration that induced the toxic effect.

This and similar techniques are commonly used to establish "Species Sensitivity Distributions" for individual toxicants, based on controlled experimental data[12]. As can be seen from the figure-

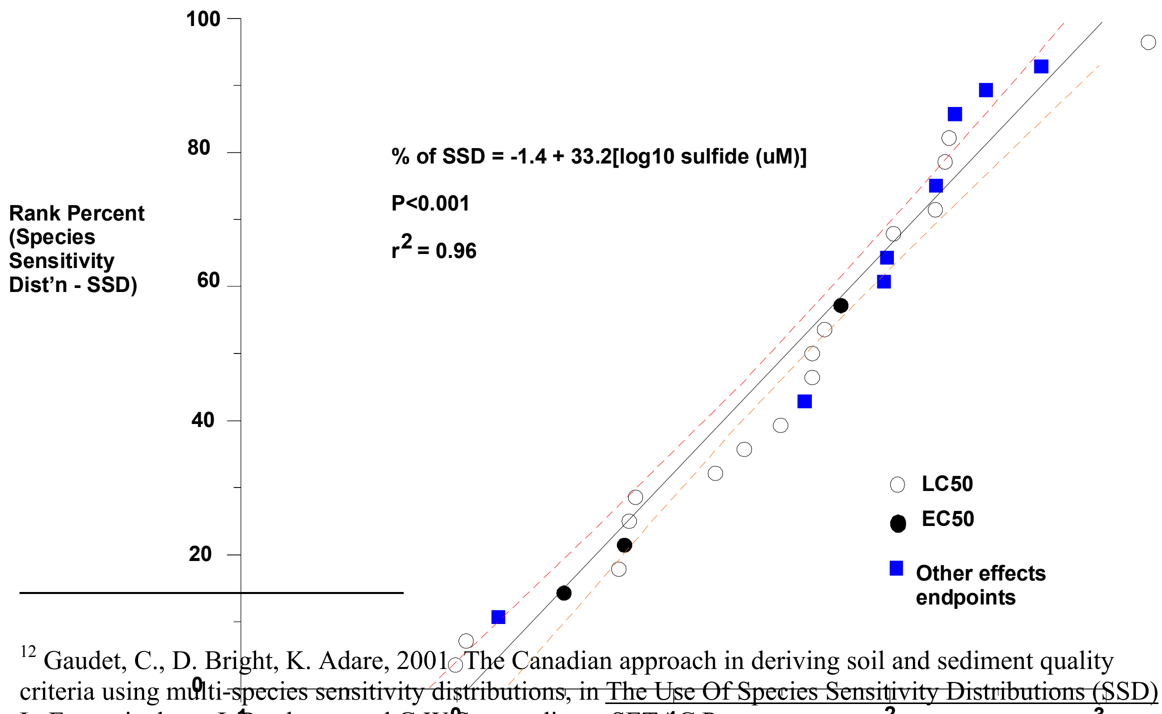
- non-lethal toxicity responses tend to occur at lower toxicant concentrations than lethal endpoints;

- the rank % of the distribution is related to the toxicant concentration as a straight line when the concentration is plotted on a $\log_{10}$ scale. This suggests that the species sensitivity distribution is log-normally distributed;

- tested marine organisms tend to respond negatively (at an EC_{50} or LC_{50} level) to ammonia concentrations in the exposure medium over the range of 0.3 to 100 mg/L ammonia (with a few outliers).

The above-mentioned observations tend to hold for the vast majority of aquatic life toxicants. The documented mortality response range of marine organisms to ammonia is actually quite small compared to some toxicants, spanning slightly more than two orders of magnitude of concentration.

If the EC_{50} endpoints are also taken into account, the estimated range over which toxic responses have been observed spans four orders of magnitude.



[12] Gaudet, C., D. Bright, K. Adare, 2001. The Canadian approach in deriving soil and sediment quality criteria using multi-species sensitivity distributions, in The Use Of Species Sensitivity Distributions (SSD) In Ecotoxicology. L. Posthuma and G.W. Suter, editors. SETAC Press.

A similar preliminary assessment is provided of the reconstructed species sensitivity distributions for sulphides, based on the available laboratory toxicity data; however, mortality and non-mortality type endpoints have been combined since there were too few of the latter found to allow separate analysis.

The construction of species sensitivity distributions are explicitly carried out in order to estimate the percent of species that might be affected at a given concentration based on direct toxicant effects.

The approximated species sensitivity distributions for ammonia and sulphide, as shown, are not overly conservative since most of the available data were for short-term (acute) laboratory exposure periods (96 h or less) as opposed to longer term, more chronic exposures. Furthermore, they are based on an effects level where either 50% of the exposed organisms died, or there was a 50% reduction in some ecologically relevant physiological function. Often, the evaluation of species sensitivity is carried out using information on chronic toxicity, and much lower response levels (LC_5, LOEC, NOEC).

In addition, the toxicity endpoints in the figures are expressed based primarily on a nominal (spiked) concentration of ammonia or sulphide. In reality, much of this spike is lost very rapidly in the test apparatus, such that the actual exposure concentration associated with a given toxicological response is much lower than this.

Based on the data as shown, a relatively non-conservative estimate of the number of species that might be affects at a given ammonia or sulphide concentration is as follows:

Est. % of spp. adversely affected	**Corresponding Exposure Concentration**			
	Ammonia (mg/L)	**Ammonia (μM)**	**Sulphide (mg/L)**	**Sulphide (μM)**
5%	3.8	210	0.053	1.6
25%	11	590	0.21	6.2
50%	39	2,200	1.2	35
75%	140	7,800	6.8	200
95%	400	22,000	27	800

The sulphide thresholds for effects on aquatic organisms, above, are useful for putting into context the July, 2001, values used by Erickson et al. to categorize current sites. The published literature also casts doubts on Brook's (2001) conclusion that a threshold for exclusion of some species was > 300 μM sulphide.

It should be noted that natural (pristine) sediments along the British Columbia coastline often have measurable concentrations of ammonia and sulphide in pore waters at shallow subsurface depths (a few cm) that are comparable to the values listed in the first two lines of the table above. The relevance of the above-described laboratory toxicity data to the real environment may be questionable, since their may be potential for acclimation and adaptation in field populations, as well as the potential for poorly appreciated chemical interactions that limit toxicity. One possible reason for discrepancies between an organism's presence in suboxic sediments the field and apparent sensitivity to ammonia or sulfide is the tendency of organisms to create relatively well flushed microenvironments in and immediately adjacent to their sediment burrows relative to sediments farther away. Direct field measurements of biological responses that are broadly representative of site conditions are clearly more appropriate for use in establishing response thresholds than laboratory toxicity data used in isolation.

It will be interesting to see how the species sensitivity distribution for sulphides, based on laboratory toxicity tests, compares with thresholds of effects based on field studies and benthic community data, based on the recent WLAP studies.

Overall, the SAG highlights the ammonia issue simply as a pre-requisite for further thought and discussion.

Metals and Persistent Organic Contaminants

Because the major portion of organic carbon released as aquacultural waste is amenable to rapid microbial break-down, and since some substances within the waste are expected to be moderately to highly recalcitrant to microbial biodegradation, there exists a potential for the build-up in sediments under and around net pens of various contaminants. These might include Cu, Zn, other metals, PCBs[13], PCDD/Fs, other organochlorines (substances that have a greater local persistence than the overall organic matter in which they are delivered, and – therefore – exhibit potential to accumulate over time based on chronic site use). We were unable to find any information on the concentrations of metals other than Cu and Zn, or organic contaminants under aquacultural operations within British Columbia.

Overall, concerns about such contaminants are plausible in the absence of any real data, in light of the chronic use of feeds with fish oils and meal from populations that have accumulated through biomagnification these globally redistributed persistent contaminants. Modern "high energy" extruded pellet diets for Atlantic salmon include around 22% or more (up to 30% in some cases) by weight of lipids. Many feed formulations are proprietary while content information for a few is openly available. High- energy growth diets rely extensively on fish meal (>90% of dietary protein) and marine oils (20-30% of dietary requirements).

[13] According to the Nov. 25, 2001 edition of the Sunday Times (London), " It was also revealed that the Scottish Environment Protection Agency had found high levels of polychlorinated biphenyls (PCBs) in sediments below or near many fish farms." ("'Safety threat' to salmon farmers". Mike Merritt)

Currently, there are initiatives within the international farming community to use fish meal and oil from areas of the world's oceans (e.g. southern hemisphere) that exhibit lower levels of accumulation of globally reidistributed POPs. In addition, replacement of fish-based oils with plant-derived oils is actively being investigated.

According to Nash (2001)[14] –

> "Data collected by the SCAN (2000) on basic feed ingredients (roughages, grains and cereals, vegetable oils, animal fat and other rendered by-products, fish meal and fish oil, as well as binders and trace element premixes) indicated that virtually all are contaminated with dioxin to varying degrees. Feedstuffs originating from plants generally contain low levels of dioxins (0.1–0.2 ng WHO-TEQ/kg dry matter), while fish meal and oil, particularly those originating from European sources are highly contaminated (fish meal 1.2 ng WHO-TEQ/kg dry matter, fish oil 4.8 ng WHO-TEQ/kg dry matter). European fish meals and oils are about 8-fold lower in total dioxin content than those produced from species caught in the coastal areas of less-industrialized regions of the world (Peru, Chile). Because of the high percentage of fish meal and oil in the diets of farmed carnivorous fish, such as salmon, the impact of using less contaminated feed materials of fish origin on whole diet dioxin burden is considerable. According to SCAN estimates, a typical diet for carnivorous fish containing 50% fish meal and 25% fish oil originating from Europe might contain 1.82 ng WHO-TEQ/kg dry matter, compared with 0.25 ng WHO-TEQ/kg dry matter if fish products from the south Pacific were used. Further reductions may be realized by partially replacing fish meal and oil with plant products, such as soybean meal and vegetable oils."

In recent years there has been controversy regarding mercury levels in tuna – especially with regard to canned tuna and risks for sensitive individuals within the human population. We do not suggest that tuna is used as a feedstock for food pellets used in aquaculture; nor that there is a risk of human mercury exposure through aquaculture output. Rather, the information reinforces the fact that fish oil and biomass from pelagic fish can contain sufficiently elevated levels of persistent, bioaccumulative contaminants, depending largely or trophic position.

This would not be an issue unless there was additional potential for a form geochemical focusing in sediments based on (i) long term cumulative loading, and (ii) much higher persistence than the organic matter in which it is delivered to the seabed.

The SAG was not unanimously convinced of the need for addressing possible introduction to the seabed of metals other than Zn and Cu, or other persistent, bioaccumulative substances within an aquaculture waste control regulation. We find, however, some public controversy around this issue and little real data. Rather than provisionally building a requirement to routinely monitor persistent, bioaccumulative contaminants in sediments, we recommend that targeted studies be undertaken to provide a much more concrete picture of the hypothetical risks, so that adequate revisions can be built into the interim regulation, if required.

4. Chemical Methods under the Reg. – getting meaningful, interpretable data from simple field measurements of effects.

The SAG will provide future detailed comments on appropriate field sampling and analytical techniques for the chosen chemical surrogates and other measures. For the time being, we generally emphasize the need for appropriate versus inappropriate techniques for measuring S^{2-} and ammonia concentrations.

5. The Metrics for Defining Environmentally Protective Thresholds.

The regulation will likely include one or more trigger or compliance concentrations of surrogate variables for biological impact of aquacultural wastes. The value of such numbers depends on –

> (i) the confidence in the interrelationships between predicted and realized community impairment for a given concentration,
>
> (ii) scientific/technical estimates of the larger ecological significance of various predicted biological changes, and
>
> (iii) policy decisions about what level of ecological impairment is acceptable or appropriate below, near, or farther removed from one or more aquacultural operations.

Many regulatory jurisdictions around the world set soil, sediment or water effects thresholds for contaminants at concentrations resulting in effects on 5% of the potentially present species or fewer (e.g. Australia, New Zealand, Netherlands, U.S. for some environmental quality guidelines). Water quality guidelines in Canada and British Columbia are by policy even more stringent, being set a levels at or below the threshold for effects on the most sensitive organism (below the lowest observed effet level). Often, where the data are inadequate for the task of establishing credible dose-response relationships, some estimate of the lower effects threshold is further adjusted (divided) using a 10- to 1000-fold uncertainty factor. Such practices have become routine throughout North America and Europe.

There are mandatory requirements to derive matrix soil quality standards under the BC Contaminated Sites Regulation and the Waste Management Act that minimize risks to either soil invertebrates living in or plants rooted in contaminated soils. This is analogous to benthic biota in aquatic communities. As a matter of policy, the soil standard for the protection of soil invertebrates and plants is established as the lesser of an EC_{50} (non-mortality endpoints) or LC_{20} response level for agricultural, residential, or urban parkland areas, and the greater of the two for commercial and industrial sites.

The CCME sediment quality guidelines specify a threshold biological response level (Threshold Effects Level – TEL) which combines the ranked 15th %ile concentration for all studies where effects were noted, regardless of the response level, and the 50th ile concentration of the data set for which no effect was noted at a given exposure concentration, using a geometric mean of the two. A probable effects level (PEL) is defined as the square root of the product of 50th %ile of the effects data set and the 85th %ile of the no-effects data set. Subsequent to the development of the CCME sediment quality guidelines, several studies have shown that adverse effects are routinely observable in marine field populations when one or more contaminants in sediment exceed the PEL, a target value that is generally much lower than the 50th%ile of effects data. Rarely is there a lack of adverse impact when the PEL is exceeded.

This brief discussion is included simply to illustrate that a 50% species reduction would be very much at the upper extremes in policy decisions undertaken within the international regulatory community.

The recent focused studies by WLAP and Brooks (2001) begin to provide detailed information on the relationships between chemical substances in sediments in the vicinity of current aquaculture sites and benthic community structure. Brooks, however, appeared to use whatever curvilinear fit for the data that provided the best fit for the actual data irrespective of whether it made sense theoretically (e.g. see Brooks, 2001, Figures 52, 85). A polynomial fit can easily be calculated for any bivariate relationship, but more often than not results in an over-specification of the fit to the data in hand, such that the relationship is clearly not generalizable to other independent data sets, and has little if any predictive value.

In contrast, the use of an exponential model is often appropriate for variables that correlate well with the distance from a point source discharge and are causally related to the re-distribution of stressors.

As noted above, species sensitivity distributions to toxicants are considered by many current researchers to be best described as log-normal distributions. For defining predictive relationships between chemical surrogates of toxicity and loss of taxa, a log-normal regression model can be supported based on theoretical considerations. This assumes, however, that benthic community responses are the simple result of the progressive loss of taxa with increasing toxicant concentrations, based on their individual physiological susceptibility. Such an assumption is consistent with a larger conceptual model of soft-bottom marine benthic communities simply as species assemblages, where there is little interaction between or co-dependency of the members of the larger community.

Many ecologists hold that the species assemblage model of community structure is inappropriate. Rather, they prefer to think of communities as mutually interacting groups of organisms, where the activities of one organism facilitate the presence (or absence sometimes) of others. The model of Rhoads and Boyer (1982) as discussed above, clearly adheres to the view that benthic communities and community succession involves emergent properties that are not adequately encompassed using a species assemblage construct.

We argue that two-way biota-sediment interactions has the potential to drive community succession either toward very shallow Stage 1 type successional communities or towards later successional Stage 3 communities, with larger deeper burrowing fauna. There exists the possibility, therefore, that beyond critical sulphide, redox, and/or other chemical thresholds there is a possibility that near-surface anoxic conditions would be reinforced – in part through a further erosion of organic carbon assimilative capacity once sulfate levels were sufficiently depleted.

In simple terms, the dynamics of community succession near an aquacultural operation might be explained by the dominance of one of two states (surface anoxic/surface oxic) with the actual conditions bifurcating over space or time at some critical threshold. If this is the case, then the accurate identification of the appropriate threshold becomes very important. One statistical model that accommodates the identification of critical breakpoints within non-linear, potentially chaotic systems is the “hockey stick” or “broken stick” regression models. The model assumes that the overall bivariate relationship is described by two separate relationships, each of which can be described using a linear or log-linear fit. The intersection of the two lines on the independent variable (toxicant or surrogate concentration) represents the threshold between two states, each controlled by different driving forces. In terms of mechanics, various points near the visually identified break-point are assigned to one side or other of the two bivariate relationships until the least-squares fit of both produces the least additive variance.

The SAG, WLAP and MAFF staff engaged in discussions about species richness as a regulatory endpoint, in the specific context of natural variation versus loss of biodiversity beyond normal variation. The following table summarizes the variability of species richness in the detailed study sites included in the Brooks (2001) study.

Site	**Reference Sites**		**Farm sites (0, 15 and 30 m from operation)**	
	Mean No. of Taxa	**±95% C.I.**	**Mean No. of Taxa**	**±95% C.I.**
A	60.3	3.8	28.7	5.5
B	64.1	6.4	18.4	5.6
C	33.3	3.4	13.7	2.5
D	38.0	8.0	14.0	6.5
E	60.7	14.7	9.7	11.8
F	42.3	17.4	8.3	20.8
G	56.7	32.0	10.3	7.6

Based on this and previously stated considerations, there may be reasonable justification for targeting aquaculture waste discharge regulations at a level that represents less than a 50% loss of species richness. The natural variability at most sites is perhaps not so high that more modest perturbations cannot be identified as being statistically significant. In addition, it is not clear that a 50% loss in species richness does not represent an appreciable ecological impairment.

The study by Brooks (2001) and the WLAP focus study placed considerable emphasis on appropriate sampling design, especially with regard to the choice of reference locations. Clearly, there will be a need to ensure that reference and aquaculture sites are comparable with regard to bottom conditions, depth, current velocities, local relief, ventilation rates, and oxygen conditions (based on pre-operational conditions).