4 Review of overseas practices for marine water quality objectives

4.0.1 This section describes and compares the rationales, scientific basis and methodologies adopted by various jurisdictions in deriving WQOs. Particular emphasis is given to the pros and cons, as well as limitations and constraints of the different approaches used.

4.0.2 The term “Water Quality Objectives” (WQOs) used in Hong Kong is termed “water quality criteria” (WQC: USA), “water quality guidelines” (WQG: Singapore, Australia, and Canada), “water quality standards” (WQS: PR China, Malaysia), and “environmental quality standards” (EQS: Japan, EU and European countries) in other jurisdictions. WQOs are always formulated in terms of what functions the water is being used for (i.e., “beneficial uses”, “intended uses” or “environmental values”) and hence what uses are being protected by the WQOs.

4.0.3 The most common and traditional approach for deriving WQOs has been measurements of physical and chemical parameters, and assuming that if these physical and chemical parameters can be maintained at certain level, the aquatic environment will be protected. However in more recent years it has been recognized that these are largely indirect measures of the state or health of the environment, and the alternative way is to monitor the biology of the environments directly (e.g., ANZECC and ARMCANZ, 2000a). Nevertheless, the WQOs still play an essential role in preserving the health of aquatic ecosystems, as the parameters concerned are easier to measure and monitor than most bioindicators. The review will cover at least over 40 water quality parameters or indicators which are listed in Appendix A1.

4.0.4 Water quality objectives can be an important component of any framework for water resources management. In very broad terms, there are three different approaches to water resources management adopted by overseas jurisdictions (CCME, 2003):

1) **The technology-based approach**: where limits on the release of chemicals are based on some definition of what can reasonably be achieved technically/economically. As such the standard for discharge into the receiving waters primarily depends on the effectiveness of the treatment technology and the dilution capacity available, whilst little or no consideration is given to establish WQOs. This approach is generally adopted by jurisdictions such as Germany, Japan, Malaysia, etc.

2) **The use-protection approach**: that essentially involves the designation of beneficial uses/ environmental values to a water body, and an appropriate mix of management options are applied to ensure these uses/values are not compromised. In this approach WQOs are the basis for assessing whether the designated uses/values are being adversely affected. They can also be used to back calculate to a corresponding effluent concentration. This approach is commonly adopted by jurisdictions such as Australia, Canada, Europe, and US.

3) **The non-degradation approach**: where discharge limits are established based on the natural background levels of substances of concern at the site. This approach is in fact the strictest form of the “use-protection approach”, and has generally been restricted to waters of high environmental value.

4.0.5 Most jurisdictions defined beneficial uses (or intended uses or functional uses or environmental values) to some extent and established associated WQOs for a number of parameters. This highlights the importance and prominence of the use-protection approach and justifies in particular the setting of WQOs.

4.0.6 In practice, a mix of management approaches is usually used. WQOs can be of assistance in benchmarking individual technology-based approaches, and technology-based approaches are likely to be included in the mix of management strategies used to achieve WQOs. WQOs can also form part of the non–degradation approach if the framework for establishing WQOs is broad and flexible, as in the case of Australia (ANZECC and ARMCANZ, 2000a) and the EU (European Commission, 2000). In the Australian state of NSW, and many other jurisdictions, all three approaches can be in use at the same time depending on the location and situation.

4.0.7 Throughout all of the jurisdictions, different types of methodologies are used for deriving WQOs, for each of the three kinds of parameters of interest: toxic chemicals, physico-chemical characteristics (including nutrients), and microbiological indicators. While there appears to be a consensus in the methods used for derivation of WQOs for the latter parameters (physical, nutrients and microbiological), there are more disparate opinions and ways of estimating WQOs for toxic substances. This fact reflects the many gaps in knowledge about ecotoxicology, and consequently translates in many uncertainties that make the task of regulation and water policy quite difficult.

4.1 Nutrients and physical characteristics

4.1.1 The most common method, and the one adopted by a majority of countries around the world, is to establish a "**baseline**" of values for reference sites against which to compare the quality of the waters. The resulting WQOs usually consist of upper and lower limits within the natural range of variation for some parameters, or just upper limits for others. Separate WQOs may be established to cater for the seasonal variations.

4.1.2 The sources of information that can be used to establish the baseline or reference condition can include: historical data collected from sites of interest; spatial data collected from sites or areas nearby that are uninfluenced (or not as influenced) by the disturbance being assessed; or data derived from other sources (ANZECC and ARMCANZ, 2000a). The latter might be considered for instance as an interim measure if there are neither suitable historical data nor comparable reference sites available. It could include identifying the reference condition from published literature, from modelling or from expert opinion. For modified ecosystems, use of the "best available" reference sites may be the only option available.

4.1.3 At its simplest, the approach involves taking a percentile of the reference distribution as the WQO. The approach can, however, also include or lead to consideration of site specific modifying factors, development of empirical relationships between parameters, use of predictive modelling and assessment of sustainable loads.

4.1.4 In theory, an advantage of Hong Kong over many other countries is the uniformity of climatic conditions throughout its relatively small area (1,651 km^2 of seawater), which minimises the variability between sites and may make it possible to establish the same baseline dataset for the entire territory. Different percentiles of this data set, for example, could then be used to establish WQOs for different levels of ecosystem protection. In this way, the 80th percentile, as recommended in Australia (ANZECC and ARMCANZ, 2000a), could be used slightly to moderately disturbed zones whereas a less stringent percentile, perhaps the 90th or 95th percentile, could be used for highly disturbed water bodies. The different seasons will probably need to be treated separately.

4.1.5 In practice, the main difficulty with the approach outlined above is the size, the proximity and the influence of the Pearl River. Natural conditions in the western approaches will be more variable and superimposed on this variability is the pattern of contamination from the PRD and local sources.

4.2 Toxic substances

4.2.1 The European countries and Canada (and to some extent the USA) are applying a mix of methods to deal with persistent chemicals (all of which are organic chemicals or POPs) that show bioaccumulation in organisms. Those methods that specifically address bioaccumulation are appropriate for setting WQOs for protection of wildlife predators, human consumers of seafood and aquaculture products. For non-persistent toxic chemicals, metals and persistent chemicals not related to bioaccumulation, there is still debate in scientific circles as to the best way to set quality guidelines or standards.

4.2.2 There are differences in the statistical methods adopted by various jurisdictions used to estimate the protective thresholds for all species. A common problem is that the toxicological data available have been derived for a few species tested under laboratory conditions and the bulk of the data are acute toxicity (LC_{50} and EC_{50}) values, rather than chronic no-observable-effect concentrations (NOECs). This variability in sensitivities is accounted for in part in the statistical procedures used to estimate the thresholds from the laboratory data. These are as follows:

- The **Assessment Factor** (AF) / Safety Factor (SF) approach was the traditional method used to derive early WQOs by the USEPA and others. The factors were usually applied to the most sensitive data point. The factors were applied to convert from acute or LOEC data to chronic NOEC figures (10 – 100, as in Canada), and also to account for variations in quantity and quality of data (between 10 and 10,000, as in Europe). This approach is still used to derive WQGs in some jurisdictions, even in those that prefer the SSD approach (EU, Australia), usually to adjust the SSD figure for further uncertainties.

- The **Triangular distribution** approach, which is only used by the USEPA and is an early approximation of later SSD approaches; it uses all available NOEC data for a chemical and fits a triangular distribution to the data to protect a nominated percentage of species.

- The **Species Sensitivity Distribution (SSD)** approach, which is the preferred method by many developed countries (e.g., The Netherlands, EU, Australia), although a variety of SSD curves have been used. Newman et al. (2000) reported that many toxicity datasets do not fit the typical sigmoid curve of the SSD, but the Australian use of the Burr distribution curves largely overcomes this problem (ANZECC and ARMCANZ, 2000a; Campbell et al., 2000; Shao et al., 2000).

- **Bootstrapping** is a universal method, applicable to any distribution of toxicity data (Grist et al., 2002), and developed to address some of the limitations mentioned by Newman et al. (2000); European countries can use it in addition to the SSD method. It is simpler mathematically than the early SSD curves, although it requires computing power. Again, the Burr distribution software (ANZECC and ARMCANZ, 2000a) may be simpler to use and do the same job.

4.2.3 Warne (1998) and ANZECC and ARMCANZ (2000a) have reviewed the first three methods, and determined that the SSD approach was more consistent with risk principles, particularly that of more data giving greater confidence in the WQO figure. Both the SSD and bootstrapping statistical methods are scientifically sound and produce similar results. The AF method has been criticised for being too subjective (Chapman et al., 1998; Warne, 1998). Indeed, the factors used are based on limited scientific evidence, while large factors may generate threshold values lower than the standard analytical capabilities of most laboratories, causing problems with compliance. Kwok et al. (2007) recommended an additional factor of 10 when applying temperate data to tropical systems, when data are limited.

4.2.4 The critical process is to determine what degree of protection from chemical pollution the threshold values would provide to an ecosystem. The aim would be to ensure that any concentration of toxicants in water and sediment do not reduce the populations of most or all the species that form an integral part of a particular ecosystem and do not impair the overall structure or function of the ecosystem. For instance, Canadian guidelines aim at protecting 100% of all species everywhere from long-term exposure, whereas European countries, Australia and USA aim at protecting a percentage of species, usually 95%, sometimes 99% (pristine areas) or 80% (heavily modified ecosystems).

4.2.5 In addition to all the above, the European countries and the USA have two sets of thresholds: one for chronic effects (called AA-EQS in Europe and CCC in the USA) and another one for acute effects (called MAC-EQS and CMC respectively). It is debatable whether such distinction may be practical in terms of protection to the ecosystem, but it may help regulatory authorities in their monitoring since no-compliance with the acute thresholds is often indicative of accidental spills or misuse of toxic chemicals (pesticides, waste discharges, etc.), which are likely to be temporary and relatively easy to deal with, whereas no-compliance with chronic thresholds may be indicative of deeply entrenched contamination problems which require an investigation and tough decisions. It should be noted that the methodologies for deriving the short-term exposure protection figures are not as robust as for long-term exposure, the protection levels are less certain, and there are monitoring difficulties to consider.

4.3 Biological criteria

4.3.1 The overseas practices of development and use of biological criteria are summarized below:

Australia

4.3.2 The Australian national guidelines (ANZECC and ARMCANZ, 2000a) put an emphasis on assessing aquatic biological communities. ANZECC and ARMCANZ (2000b) view bioassessment "*as a vital part of assessing changes in aquatic ecosystems, and as a tool in* *assessing achievement of environmental values and attainment of the associated water quality objectives*". The biological information is an integration of many natural and human disturbances. Guidelines for biological assessment, outlined in the Australian document, are intended to determine substantial ecosystem effects, including: "*changes to species richness, community composition and/or structure; changes in abundance and distribution of species of high conservation value or species important to the integrity of ecosystems; and physical, chemical or biological changes to ecosystem processes*".

4.3.3 Protocols are provided for some bioassessment methods and new ones are being developed with improved indicators, experimental design, sampling approaches, decision criteria (i.e., acceptable level of change) and analysis. Biological assessment can have several different aims (with accompanying different experimental designs). Common among these are: broad-scale assessment (e.g., for rapid screening or coverage on broad geographical scales); early detection of changes; and assessment of biodiversity to determine adverse effects at population, community and ecosystem levels (e.g., determining ecological sustainability of human activity).

European Union

4.3.4 The European Union has established environmental objectives that are defined to a large extent in terms of biological quality elements (see Article 4 and Annex V of European Commission, 2000). To assess compliance, member states need, among other things, to establish type-specific biological reference conditions that represent high ecological status. For coastal waters this needs to be done for three biological quality elements:

- Composition, abundance and biomass of phytoplankton;
- Composition and abundance of other aquatic flora (macroalgae and seagrasses); and
- Composition and abundance of benthic invertebrate fauna.

The reference conditions for each then form the basis of a classification system that is used to assess whether surface water status is high, good, moderate or poor. For most water bodies member states are required to "*protect, enhance and restore (...)with the aim of achieving good surface water status at the latest 15 years after the date of entry into force of this Directive*" (i.e., 2015). Progress on adoption of this Directive is illustrated in the UK's Technical Advisory Group (UKTAG, 2005), where the biological parameters are being integrated with the water quality and chemical pollutant standards.

Canada

4.3.5 The concept of ecosystem-based management is basic to the Canadian approach to site-specific application of environmental quality guidelines (EQGs) (CCME, 2003). This concept incorporates integrated management of natural landscapes, ecological processes, physical and biological components, and human activities. These 2003 guidelines form the "*scientific basis for developing site-specific environmental quality objectives*", which in turn influence water management strategies that incorporate social and economic factors. The site-specific WQOs incorporate measurement of biological indicators of ecosystem health and integrity, along with traditional physical and chemical indicators. CCME (2003) recognises that no single method can adequately address all requirements for developing WQOs but information on "resident species" is an important component of overall evaluation, identifying their range of sensitivity and factors that influence chemical bioavailability.

US

4.3.6 Biological criteria, or biocriteria, in the USA are "*numerical measures or narrative descriptions of biological integrity*", which set the biological quality that must be present to support a desired condition in a water body. Biocriteria are derived from biological assessments of reference waterbodies and involve integrated measures (indices) of the composition, diversity, and functional organization of a reference aquatic community. Physico-chemical water quality data and biological data are collected to define the baseline conditions of each type of water body, and numerical biological criteria are then developed. This reference condition approach is a similar approach to the EU system described above. Five approaches for estuaries and coastal marine waters are available, which depend on the degradation status of the water body and the amount and quality of historical data. The criteria are estimated using the indicator variables as follows:

- For waters in excellent condition, the median values of the indicator variables are used;
- For degraded sites, criteria are defined by the upper quartile;
- For significantly degraded areas with reference sites, the intercept value on a regression or distribution curve is used;
- For degraded areas with insufficient historical records, the intercept is used as above but a model is required to extrapolate back; and
- For coastal waters in general, an index site approach and models are required.

4.3.7 Biological criteria are adopted on a State-by-State basis to protect aquatic life uses of the waterway. Chemical, physical and biological integrity, as defined by the Clean Waters Act (Section 101(a)), is taken to "*define the overall ecological integrity of an aquatic ecosystem*". Biological assessment results from State surveys on the presence, condition and numbers of types of fish, insects, algae, plants, and other organisms are compared with the biocriteria established for that water body. State biological assessment data are used to evaluate the effectiveness of management actions, as reflected in the responses and improved conditions of biological communities.

4.3.8 USEPA (2000a) has provided technical guidance for development of biocriteria and bioassessment programs by the States. The main measures of ecosystem condition are benthic infauna (macroinvertebrates), fish, aquatic macrophytes and phytoplankton, while measures for zooplankton, epibenthos and preserved remains (palaeoenvironmental reconstruction) are being developed and draft methods are provided. Data can be developed into indices of condition or indicator taxa evaluated.

Hong Kong

4.3.9 At present, there is absence of biological WQO for Hong Kong. In light of the international trend to employ biological criteria and indicators to enhance conservation of aquatic community, the review will explore the feasibility of developing biological WQOs for Hong Kong waters. The criteria formulation needs local data of aquatic organisms' responses to inhabiting water conditions and pollution levels. EPD has implemented a biological indicator monitoring programme to collect the required data.

4.4 Microbiological WQOs for bathing waters

4.4.1 Expert review of existing epidemiological studies is the underlying method used to derive WQOs (and guidelines and standards) for recreational waters. Jurisdictions either derive their own WQOs directly, or after considering what recent developments have been made in the field, decide to adopt or adapt the WQOs developed by others. The World Health Organization (WHO) and the European Commission (EC) are the jurisdictions that have most recently developed their own WQOs (WHO, 2003; European Commission, 2006b). The respective approaches (and outcomes) are similar and differ from what was common practice previously. Both have taken a risk based approach and provide a series of numerical values that are used to classify water quality status. Their respective approaches also include a requirement to evaluate the likelihood of faecal contamination. This is to be done by means of a sanitary inspection or "*bathing water profile*". Australia has adopted the World Health Organization approach with minimal changes (NHMRC, 2008). Singapore has also used the World Health Organization approach but they have simplified it by selecting one value from the series as a threshold value.

4.4.2 WHO (2003) provides a review and assessment of the health hazards encountered during recreational use of coastal and fresh water environments. Guidelines for recreational use of marine waters are provided in WHO publication *Guidelines for safe recreational water environments* (WHO, 2003). The primary aim of the guidelines is the protection of public health. The specific guidelines that are given to address microbiological quality relate to activities where whole-body contact takes place (i.e., those in which there is a meaningful risk of swallowing some water). There are no corresponding guidelines covering secondary contact recreation which would involve, for example, wading, boating and fishing.

4.4.3 Of the water quality aspects, the main concern (and most specific guidance) relates to faecal pollution. Compared with earlier guideline approaches to faecal pollution the principle focus has expanded from retrospective numerical compliance assessment to include real-time management and public health protection. One of the main outcomes of this is a classification system for recreational waters based on a combination of sanitary inspections and microbial measurements.

4.4.4 An important consideration was the "Annapolis Protocol" (WHO, 1999) which recommended:

- A move away from reliance on numerical values of faecal indicator bacteria as the sole compliance criterion to the use of a two component qualitative ranking of faecal loading supported by direct measurement of appropriate faecal indices.

- Allowance for the impact of actions to discourage water use during periods, or in areas, of higher risk (e.g., advice to avoid swimming after rainfall events).

These recommendations that led to a classification of recreational water environments recommended in WHO (2003). This classification is essentially a matrix formed by the outputs of a sanitary inspection (the "ranking of faecal loading") and the results of microbiological monitoring (the "direct measurement of appropriate faecal indices").

4.4.5 A series of randomised controlled trials performed in the temperate northern European waters (e.g. United Kingdom) formed the key studies for the derivation of numerical guideline values for the microbiological quality of bathing waters (WHO 2003). The study revealed that for marine waters, intestinal enterococci (faecal streptococci) showed a dose-response relationship in bathers for both gastrointestinal illness and acute febrile respiratory illness (AFRI).

4.4.6 WHO Guidelines for the selected contaminants are included in Appendix A5. Numerical guideline values for numbers of intestinal enterococci per 100mL are expressed in terms of the 95th percentile and represent readily understood levels of risk based on the exposure conditions of key studies. WHO advised that the guidelines are flexible, and in devising local bathing standards, the country/region should take into consideration account socio-cultural (immunity, illness rates of population), environmental (climate, hydrology) and economic conditions to suit regional, national and/or local circumstances.

4.4.7 A summary of bacterial water guideline/standards for bathing waters adopted by some overseas jurisdictions is given in Table 4.1. The existing WQOs (≤ 180 count/100ml *E. coli*, calculated as the geometric mean for all samples collected during the bathing season) and grading system for Hong Kong beaches were based on the results of the epidemiological studies jointly conducted in late 80's and early 90's by EPD and the University of Hong Kong, making reference to the then WHO's report. This benchmark corresponds to a swimming-associated illness rate of 10 cases per 1000 swimmers. In addition, the beach rating system sets an upper limit of 610 *E. coli*/100 mL corresponding to a swimming-associated illness rate (i.e. health risk) of 15 cases per 1000 swimmers, which is comparable to overseas practices, such as those adopted by the USEPA (19 cases per 1000 swimmers)[3] and WHO (50 cases per 1,000 swimmers)[4]. The health risk approach for deriving the current WQOs is similar to overseas practices.

4.4.8 Nevertheless in light of the international trend to adopt the WHO guidelines and the application of enterococci as the bacterial indicator, the review will revisit if the existing WQOs and bacteria indicators (*E. coli*) are still fit for long-term protection of bathing beaches in Hong Kong. It will also examine whether the variation of enterococci level in water would correlate well with the changes in pollution levels normally encountered in local beach water (which is sub-tropical in nature), and the acceptable risk level for human health, as well as the need and feasibility of adopting enterococci as an alternative/supplementary bacteria indicator.

---

[3] The risk rate (19 cases per 1000 swimmers) is used in the USEPA's Final Rule on "Water Quality Standards for Coastal and Great Lakes Recreation Waters", 16 November 2004.

[4] The WHO guidelines classify the microbial water quality into four categories: A, B, C and D. Category B (corresponding to a gastrointestinal illness rate of 50 cases per 1000 swimmers) is generally taken as the acceptable swimming-associated health risk level.

Table 4.1 Summary of bacterial water guidelines/standards for bathing waters

Country/Region/Organization	**Water quality guidelines/standards for bathing water (per 100 mL) for bathing water**
WHO (Note: Four categories of microbial quality, A, B, C and D)	For marine waters (Category B) Intestinal enterococci ≤ *200* (95th percentile)
	For fresh waters *E. coli* (guideline value not yet derived)
E.C. Directive (Note: Four classes of beaches: Excellent, Good, Sufficient and Poor)	For marine and transitional waters (for "Good" beach) *E. coli* ≤ *500* (95th percentile) Intestinal enterococci ≤ *200* (95th percentile)
	For fresh waters (for "Good" beach) *E. coli* ≤ *1000* (95th percentile) Intestinal enterococci ≤ *400* (95th percentile)
Hong Kong	For marine waters ("Fair" beach) *E. coli* ≤ *180* (geometric mean of all samples collected in a bathing season)
USEPA	For marine waters Enterococci ≤ *35* (geometric mean of at least five samples) For fresh waters *E. coli* ≤ *126* (geometric mean of at least five samples) Enterococci ≤ *33* (geometric mean of at least five samples)
Australia	For Marine and Fresh waters Faecal coliforms ≤ 150 (median over the whole bathing season) or Enterococci ≤ *35* (median over the whole bathing season)

4.5 Water quality guidelines relating to bioaccumulation and mariculture

4.5.1 Some countries have developed guidelines for chemical substances that persist for long periods and bioaccumulate in organisms. It is only in recent years that approaches to developing guidelines or standards for these types of chemicals have become available and only a few countries have developed guidelines using these approaches.

4.5.2 Some guidelines are based on residues in tissues that would protect predators from secondary poisoning or humans from eating fishery products. In other instances such tissue residue guidelines have been related back to concentrations in water. The WHO/FAO food standards setting agency, Codex Alimentarius Commission, has established a code of practice for fish and fisheries products, stating the importance of site selection and growing water quality in the production of farmed fishery products that are safe for human consumption (Section 6.1). Information on these latter aspects are not dealt with here but can be accessed through the Codex Alimentarius website <http://www.codexalimentarius.net/download/standards/10273/CXP_052e.pdf>. The overseas practices for the WQOs relating to bioaccumulation and mariculture are summarized below.

Australia and New Zealand

4.5.3 The assumption for potentially bioaccumulating chemicals in ANZECC and ARMCANZ (2000a) is that at the low concentrations of the trigger values, significant magnification is unlikely to occur. In the absence of comprehensive guidance at the time, the protection level for such compounds was increased to 99% from the normal default of 95%. Australia and New Zealand have developed water quality guidelines (including toxicant parameters) for the protection of aquaculture species as well as human consumers of aquatic food.

4.5.4 The limits for chemicals in foods are set by the Food Safety Authority of Australia and New Zealand (ANZFA, 2000). However such limits bear no relation to concentrations in water. Given the uncertainty and limitations of adopting solely water quality standards for human health protection, the Australian and New Zealand official guidelines make it clear that the water quality guidelines mentioned in Section 4.5.3 above are not to guarantee the achievement of relevant food standards and they are to be used in conjunction with the food safety standards to protect the health of human consumers of aquaculture products (ANZECC and ARMCANZ, 2000a).

European Union (EU)

4.5.5 The European Union requires the evaluation of all available data to derive the environmental quality standards (European Commission, 2001). This includes protection of top predators and human health, and accounts for all direct and indirect exposure routes in aquatic systems including bioaccumulation. Earlier European Commission (2003) technical guidance follows the route from a Predicted Environmental Concentration in water towards a risk quotient for a top predator (bird or mammal), but the most recent guidance (Lepper, 2005) requires that a safe water concentration is calculated from a determined safe level for a predator.

EU approach for protection of predators

4.5.6 The source document for the EU methodology for protection against secondary poisoning from substances with a potential to bioaccumulate is Lepper (2005). The pathway for secondary poisoning is by uptake through the food chain, hence long-term feeding studies are appropriate. The results of these studies may be expressed as concentration in food (NOEC; mg/kg_{food}) or as dose (No Observed Adverse Effect Level; NOAEL; mg/kg body weight/day) causing no effect. For further calculation, NOEC oral is needed, and this can be derived by multiplying the NOAEL by a conversion factor, based on body weight/daily food intake. Lepper (2005; Table 7) provides a table of conversion factors for eight common test species, ranging from 8 to 40.

4.5.7 A quality standard (an interim figure in developing EQSs) is given as the concentration in food of the predator (QS_{spb} where "spb" = "secondary poisoning of biota"; this approximates the $PNEC_{oral}$, which is the "Predicted No Effect Concentration" for oral intake). QS_{spb} is derived from the $NOEC_{oral}$ by dividing by an assessment factor. Again, Lepper (2005; Table 8) provides assessment factors for extrapolation of mammalian and bird toxicity data that range from 30 to 3,000, depending on the type of test. If several NOEC oral for bird or mammal species are available, the lowest of the resulting QS_{spb} is used as the standard. As toxicity data for wildlife birds or mammals are generally not available, extrapolation from laboratory test species is often necessary.

4.5.8 It is feasible, for example for planning or design purposes, to transform quality standards in prey-tissue (QS_{spb}) to corresponding concentrations in water, which can be regarded as a surrogate standard ($EQS_{sp\ water}$). This can be done if there are reliable data on partition coefficients (K_{ow}; octanol-water partition coefficient), bioaccumulation factors (BCF) or biomagnification factors (BMF) of the chemical. Lepper (2005) accounts for the longer food chains in the marine environment by incorporating biomagnification in both the prey of predators (BMF_1) and in the prey of top predators (BMF_2). For marine water:

$$EQS_{sp\ water}\ [\mu g/L] = \frac{QS_{spb}\ [\mu g/kg]}{BCF\ [L/kg] * BMF_1 * BMF_2} \qquad \text{(Equation A)}$$

4.5.9 There are few measured BMF data available, so Lepper (2005; Table 9) provided default BMF values, which vary according to log K_{ow} of the organic chemical (up to 10 for log K_{ow} of 5 – 8). Lepper expressed caution at using the water quality standard because of uncertainties associated with both default and experimental bioaccumulation data, and hence expert judgment is required.

4.5.10 The same approach applies for metals but one must take into account that inverse relationships have been observed for metals where the highest BCF values were found in waters with the lowest metal concentrations, and vice-versa. Hence, BCFs should be calculated from studies conducted with environmentally relevant metal concentrations in the test media or by using bioaccumulation factors (BAFs) observed in the field (Lepper, 2005).

4.5.11 The European Commission (2006a; Annex I, parts A and B) provides EQSs for 41 nominated chemicals in water. Generally, these EQSs are thought to provide a sufficient basis to ensure comprehensive protection and effective pollution control. There are at this stage, however, three chemicals for which the EQSs for water may not be sufficiently protective and EQS for biota have been developed as well. For these chemicals, the residue concentration in prey tissue to protect predators against secondary poisoning should not exceed the following levels (all in wet weight) of fish, molluscs, crustaceans and other biota:

- 10 µg/kg for hexachlorobenzene
- 55 µg/kg for hexachlorobutadiene
- 20 µg/kg for methyl-mercury

<u>EU human health guidelines for consumption of fisheries/mariculture products</u>

4.5.12 The EU framework also provides for derivation of human health related quality standards for consumption of fisheries/mariculture products. The technical guidance, which is provided by Lepper (2005) and summarised below, recommends a "simple but practicable approach", given that there is currently no standard approach or protocol. By convention, the amount of chemical consumed in fishery/mariculture products should not exceed 10% of the relevant threshold level for humans (e.g. the acceptable/ tolerable daily intake [ADI / TDI] or NOAEL for oral intake).

4.5.13 This human health EQS, expressed as µg/kg fishery product, is calculated using the standard human body weight (bw) of 70 kg, and standard rate of consumption of fishery products of 115 g/day:

$$EQS_{hh\ food} = \frac{0.1 * \text{threshold level } [\mu g/kg\ bw] * 70\ kg\ (\text{human bw})}{0.115\ kg\ \text{seafood consumption}} \qquad \text{(Equation B)}$$

4.5.14 This $EQS_{hh.food}$ can be transformed to the corresponding concentration in water ($EQS_{hh.food.\ water}$) by applying the same approach for transforming QS_{spb} above:

$$EQS_{hh.food.\ water}\ [\mu g/L] = \frac{QS_{hh.food}\ [\mu g/kg]}{BCF\ [L/kg] * BMF} \qquad \text{(Equation C)}$$

4.5.15 For example, the European Commission (2001; amended 2002 and 2005) has set maximum levels of some metals in seafood for human consumption. These are:

- Lead: between 200 – 1,500 µg/kg, depending on the species
- Cadmium: 50 – 500 µg/kg
- Mercury: 500 – 1000 µg/kg

These human consumption figures do not directly relate to concentrations in water, due to human risk calculations that consider body weight and individual consumption of seafood. It is at least theoretically possible to convert them to water concentrations using Equation C in Section 4.5.14 above.

4.5.16 These approaches are the same for metals as for organic chemicals. The approach does not specifically consider possible high risk groups (which may be a relevant consideration in Hong Kong where seafood consumption per capita is amongst the highest in the world), although limiting the acceptable uptake of a substance to 10% of its threshold value gives an additional margin of safety.

Canada

4.5.17 The primary route of exposure for terrestrial mammals and birds that prey on aquatic life is through consumption of contaminated aquatic organisms such as fish, invertebrates, and aquatic plants. Canada (CCME, 1998b; 1999b) has developed tissue residue guidelines (TRGs) for protection of such wildlife, particularly mammals and birds. These are for highly persistent, bioaccumulative compounds and have been developed for DDT methyl mercury, polychlorinated biphenyls (PCBs), polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) and toxaphene. Such compounds tend to accumulate to a greater extent in organisms higher up the food chain and hence water quality guidelines may not be applicable.

4.5.18 These TRGs refer to the "*maximum concentration of a chemical substance in the tissue of aquatic biota that is not expected to result in adverse effects in wildlife*" (CCME, 1999b). TRGs can apply to any aquatic species consumed by wildlife, such as fish, shellfish, other invertebrates, or aquatic plants. To protect all wildlife, the guidelines should be applied to the aquatic species at the highest end of the trophic level. However, CCME (1999b) encourages the use of species-specific or site-specific objectives wherever possible. TRGs in Canada are used, for example, in contaminated sites remediation to help interpret biological monitoring data, and can be useful screening tools to assess the potential risk of exposure through consumption of contaminated prey items (CCME, 1999b).

4.5.19 The general method for TRG derivation (CCME, 1998b) is based on Newell et al. (1987), with significant modifications. TRGs are derived from evaluating physical properties of specific chemicals, such as fate, persistence, and their environmental concentrations, as well as toxicity studies that examine ecologically important adverse effects on the wildlife (e.g., reduced reproductive capabilities) from consumption of a contaminated diet. There are pre-set minimum data requirements, but interim TRGs may be derived. To calculate TRGs, a lowest effect threshold is calculated for each mammal and bird species. Reference concentration (RC) values (contaminant levels in prey items that are considered to be protective of predators) are then calculated from body size and food intake data for typical Canadian wildlife species. The lowest RC is recommended as the TRG.

US

4.5.20 The USEPA (1995) derived fish tissue residue guidelines (TRGs) to protect predatory wildlife from bioaccumulating chemicals by first calculating dietary threshold body burdens then back-calculating to fish tissue levels using food and chemical assimilation efficiencies. They proceeded to derive ambient water-based criteria by dividing the TRG value for the chemical by its bioaccumulation factor (BAF) or bioconcentration factor (BCF), recognising that BAFs and BCFs vary among different species. The USEPA (1995) provides a method for deriving an average BAF for each trophic level.

4.5.21 The more recent revisions (USEPA, 2000a) incorporate a better consideration of non-water sources of chemical exposure, including the preference for use of a bioaccumulation factor (BAF) over a bioconcentration factor (BCF); the BAF better reflects contaminant uptake from all sources (e.g., ingestion, sediment) by fish and shellfish, rather than just from the water column as for BCF. USEPA (2000a) provides detailed procedures and guidelines for estimating BAF values (based on field-measured data from local or regional fish), and equations and background support for calculating exposures and criteria.

4.5.22 USEPA is continuing to evaluate the feasibility of developing and implementing national criteria for highly bioaccumulative chemicals that are expressed as concentrations in tissues of aquatic organisms (tissue residue criteria). Again, a BAF or bioaccumulation model would be useful to relate chemical concentrations and loads in water and sediment to concentrations in tissues of fish and shellfish.

US human health methodology

4.5.23 The USEPA criteria for the protection of human health are designed to minimise the risk of adverse effects occurring to humans from lifetime exposure to chemicals through two exposure routes; both the ingestion of drinking water and consumption of fish obtained from surface waters. The former is not relevant to marine waters and, although it is assumed that "surface waters" refers mainly to freshwater, the criteria based on fish consumption should be broadly transferable to seafood generally.

4.5.24 The calculation of criteria to protect human consumption of fish involves a different procedure to that of Canada; it involves calculation of a threshold contaminant body burden in wildlife, then back-calculation to an equivalent concentration in fish or water using a complex combination of food and chemical assimilation efficiencies and bioconcentration/ bioaccumulation factors (CCME, 1999b).

4.5.25 USEPA (2000a) has developed similar methodology for deriving water-based criteria (Section 304(a) of the Clean Waters Act) to protect human health when consuming seafood. The initial focus was to develop criteria for chemicals that bioaccumulate, such as mercury, arsenic, PCBs, and dioxin. As at 2006, the fish consumption values for arsenic, PCBs, and dioxin were still based on the water column exposure. The optional methodology is intended to guide States and Tribes to derive their own site-specific ambient criteria and standards but will be also used to calculate national water quality criteria and as default factors to evaluate State standards.

4.5.26 The revised guidelines build on the methodology of the original guidelines (USEPA, 1980) for the protection of human health, which addressed three types of endpoints: cancer, non-cancer, and taste and odour (organoleptic) effects, producing guidelines for 64 pollutants or pollutant classes. USEPA (2000a) adopted a default fish consumption value for the general adult population of 17.5 g of fish/d, and for subsistence fishers, 142.4 g/d.

4.5.27 The criteria derived from non-cancer data were based on the Acceptable Daily Intake (ADI) (otherwise known as the reference dose [RfD]). ADI values were generally derived using a no-observed-adverse-effect level (NOAEL) from animal studies, although human data were used whenever available. The ADI was calculated by dividing the NOAEL by an uncertainty factor of 10, 100, or 1,000 (depending on the quality of the data) to account for uncertainties in extrapolating limited toxicological data to humans.

Microbiological WQOs for mariculture

4.5.28 Shellfish are able to accumulate viruses or pathogens in their gastrointestinal tracts, digestive glands and other tissues. The rate of accumulation is dependent on the microbiological species and the shellfish species. Most waterborne pathogens originate in human and animal faeces, and include a wide variety of viruses, bacteria, and protozoa.

The transmission of viral disease is a key health concern associated with consumption of shellfish. Pathogens or viruses that infect humans following consumption of aquatic food are mostly of human origin, having entered aquatic ecosystems via sewage effluent discharges. These enteric viruses or pathogens are able to remain viable in the aquatic environment for certain period of time. Because of the difficulty and expense associated with the direct detection of pathogens, bacterial indicator organisms (e.g. *E. coli*, faecal coliforms) are widely used as indicator organisms to signal the extent of faecal pollution and possible presence of pathogenic organisms.

4.5.29 The microbiological WQOs for protecting mariculture are generally developed based on the Quantitative Risk Assessment Approach (Ministry for the Environment, 2003). In brief, the process would start with the definition of acceptable health risk by consuming fish and/or shellfish which are contaminated with a certain degree of faecal indicator bacterial species. Subsequently, the threshold ambient concentration of the indicator bacteria organisms in seawater can be estimated through empirical relationships which consider information such as the accumulation and depuration rates of bacteria, natural range of bacteria concentrations and pathogen concentrations in the fish/shellfish as well as safety factors. Some of the Mainland and overseas microbiological guidelines/standards for mariculture are given in Appendices A2 to A7.

<u>Complementary measures to enhance protection of human consumers of seafood</u>

4.5.30 The sole relying of water quality criteria for protection of human consumers of mariculture products have the following limitations :

- Difficulty in ascertaining the level of bioaccumulation and bioavailability of toxicants in cultured species, and the mechanisms of the chronic effects of the toxicants and carcinogens.

- Great uncertainty and limitation of extrapolating laboratory toxicity data to aquaculture environment under a wide range of environmental conditions.

- Tolerance to individual toxicants is highly species specific and variable among different aquaculture species, and usually only a few representative species can be selected for assessment.

- In addition to the culturing water, the harvested mariculture products are generally subject to various sources of contamination such as the sources of species fries, feeding materials, etc.

4.5.31 Given the uncertainty and limitations of adopting solely water quality standards for human health protection, a two-tier approach to enhance protection of human health associated with seafood consumption, through direct control on the relevant food safety standards that specify indicator levels in product flesh, and monitoring of the quality of culturing waters, is commonly adopted by jurisdictions such as Australia, New Zealand, the US, and the EU. In particular, this approach applies to the control of shellfish[5] products which are of greater health risk concern because: (i) sshellfishes are filter feeders which may

[5] Shellfish is a fishery term of aquatic invertebrate used as food, including various species of molluscs (clams, mussels, oysters, winkles, and scallops) and crustaceans (shrimp, prawn, lobster, crayfish, and crabs).

accumulate pathogens, toxic organics and metals from water; and (ii) it is popular for some of the shellfish products (e.g. oysters) used for direct raw consumption.

4.5.32 In addition to the two-tier approach of applying water quality and food safety standards mentioned above, some jurisdictions further implement complementary measures to enhance the safety of shellfish products, and reliance is generally on preventative and management approaches. These include control of the siting and classifications of shellfish growing waters according to sanitary surveys and bacteriological monitoring, requiring further treatment (e.g. depuration[6] or relaying[7]) of shellfishes harvested from certain classes of culturing waters, shellfish quality assurance programmes which cover control of the harvesting, processing, marketing or export of shellfish products.

4.6 Summary of the Mainland and overseas practices for WQOs

4.6.1 The amount and type of information available varied from jurisdiction to jurisdiction and, even when comprehensive information was available, it was still difficult to make simple and direct comparison on the basic approaches used, legislative and policy framework, level of protection, derivation methods, periodic review, and compliance monitoring and reporting. Table 4.1 summarizes these aspects for each jurisdiction studied. It is advisable that this table only be used in conjunction with the text of this review and the original source documents as appropriate.

[6] Depuration means the process of reducing the pathogenic organisms that may be present in shellstock by using a controlled aquatic environment as the treatment process.

[7] Relay means to transfer shellstock from a growing area classified as restricted or conditionally restricted to a growing area classified as approved or conditionally approved for the purpose of reducing pathogens as measured by the coliform indicator group or poisonous or deleterious substances that may be present in the shellstock by using the ambient environment as the treatment process.

Table 4.2 Summary of approaches, legislative framework and policies, methods of derivation of WQOs, practices of review and compliance for marine waters amongst the Mainland and overseas jurisdictions.

	Approach	Legislative framework	Protection[1] and derivation	Periodic review	Monitoring & reporting
PR China	Uses given values.	Apparently a mix of mandatory and non-mandatory. Implemented by Provinces, Autonomous Regions, Municipalities and Fisheries Authorities.	Ecosystems, aquaculture (production aspects) and human consumers *No information on derivation*	5 year plan	National and local; annual statistics reported on MEP website.
Australia	Uses mix of given values, site specific/ reference site data, and biological assessment. Flexible and risk based.	Not mandatory. Resource for and implemented by States & Territories. Basis for policy.	Ecosystems (various levels) *Toxicity data (statistical distribution-SSD), reference site data* Recreation (whole body/primary contact and aesthetics) *Primarily WHO* Aquaculture (production aspects) *Expert review* Human consumers *Food standards (tissue concentrations) are largely used.*	Informal ~ 10 years	Site-specific guidance given; risk based approach; implemented by States.
EU	Uses mix of given values[2], site specific/ reference site data, and biological assessment. Flexible and risk based.	Mandatory. Certain water quality classifications must be achieved by specified dates. Implemented by Member States.	Ecosystems (various levels) *Toxicity data (AF/SF3+ statistical distribution - SSD), reference site data* Recreation (primarily whole body/primary contact) *Expert review* Human consumers *Largely provided by ecosystem protection approach*	Directives in 2019/2020; some aspects every 4 – 6 years	Monitoring by Member States; annual summary.

	Approach	Legislative framework	Protection[1] and derivation	Periodic review	Monitoring & reporting
Canada	Uses mix of given values, site specific/ reference site data, and biological assessment. Flexible.	Not mandatory. Resource for and implemented by Provinces & Territories. Basis for policy.	Ecosystems *Toxicity data (AF/SF3), reference site data, and fate and persistence* Recreation *Expert review(and implied future move to WHO)*	~ 3 – 4 years; On-going updates	Site-specific guidance given; Provinces and Territories apply.
USA	Uses mix of given values[2], site specific/ reference site data, and biological assessment. Flexible.	Not mandatory. Resource for and largely implemented by States and Tribes. Basis for policy and enforcement.	Ecosystems *Toxicity data (triangular distribution) and reference site data* *Recreation (primarily whole body/primary contact)* *Expert review* Aquaculture *Provided by ecosystem protection above* Human consumers *See Section 4.2.8*	Rolling review; US EPA reviews State WQSs	States implement and report to USEPA.
WHO	Only recreation. Uses mix of given values and site specific information. Flexible and risk based.	Not mandatory. Resource for and implemented by any interested jurisdictions.	Recreation (primarily whole body/primary contact) *Expert review*	Review as new data available	Up to local jurisdiction. Regular reporting to public important. Suggestion that there be 100 samples over 5 year rolling period.

1. Approximates underlying “protection intended to offer” in the context of this table, it focuses on beneficial uses to be protected. Individual jurisdictions may use different terminology and compartmentalization.
2. Addressing both short duration events and long term condition.
3. Application factors/safety factors.