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The dependability of fishery monitoring programs: Harmonising the quality of estimates with the risks to the conservation of aquatic populations

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Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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ABSTRACT

Fisheries and Oceans Canada (DFO) is finalizing a national fishery monitoring policy to ensure that it has dependable, timely and accessible information for fisheries to manage them sustainably and to minimize harm to non-harvested incidentally captured taxa and to habitats. The policy seeks to implement an objective and consistent approach for setting the type and degree of monitoring employed across fisheries managed nationwide by DFO under the Fisheries Act. Implementation of the policy will involve evaluating the degree to which data on removals in individual Canadian fisheries are appropriate for determining whether fishery removals are sustainable for target and incidentally captured stocks / populations. An important consideration for policy implementation is the quality of estimates and decisions on compliance to limits produced using data from fishery monitoring programs. Quality describes the ability of an estimation process to produce a valid estimation or to reach a correct decision on compliance to a limit (e.g. evaluation of whether the quota has been respected). The quality of an estimate depends on the variability and bias of an estimation resulting from the randomness inherent in the data collection process and from the implementation of the sampling protocol. Under the policy, required or desired levels of quality should be commensurate to the degree of risk posed by fisheries to the conservation of aquatic populations. The present research document builds on previous DFO scientific advice as well as a draft policy implementation tool used to screen risks to conservation, and addresses three main objectives. First, we review and revise descriptors used to classify risk to target catch and bycatch species through the prosecution of Canadian fisheries. Second, we review and slightly revise approaches to quantifying estimation quality and propose an approach to harmonise the quality of estimates with the risks to the conservation of aquatic populations. Third, we outline options for modifying catch monitoring programs and/or fishery management measures to ensure that realised estimation quality levels are commensurate with conservation risk. This document supports the conclusions and advice from a DFO National Science Advisory meeting of May 14-16, 2019 for science advice on a catch monitoring risk assessment tool for a national policy on fishery monitoring.

1. INTRODUCTION

Fisheries and Oceans Canada (DFO) is finalizing a national fishery monitoring policy to ensure that it has dependable, timely and accessible information for fisheries to manage them sustainably and to minimize harm to non-harvested incidentally captured taxa and to habitats. The policy seeks to implement an objective and consistent approach for setting the type and degree of monitoring employed across fisheries managed nationwide by DFO under the *Fisheries Act*. There are several different types fishery monitoring tools used in Canada (see summary in Table A1 of Beauchamp et al. 2019). These include those for which data are reported by resource users such as fisher questionnaires, pre-departure and pre-arrival notifications (hails), commercial sales slips, creel surveys, and logbooks, and those for which data are reported by independent monitors such as dockside monitoring, at-sea observers and video monitoring systems (for a review see Beauchamp et al. 2019). These tools can be implemented as sampling surveys, whereby a random subset of fishery activities is monitored, or as censuses.

A key consideration for the policy is the quality of estimates and decisions on compliance to limits produced using data from fishery monitoring programs. Quality describes the ability of an estimation process (e.g. estimation of the total landing for a given stock) to produce a valid estimation or to reach a correct decision on compliance to a limit (e.g. evaluation of whether the quota has been respected). It measures the validity of the estimation, i.e. how close to the true value the estimate is likely to be. The quality of an estimate depends on the variability and bias of an estimation resulting from the randomness inherent in the data collection process and from the implementation of the sampling protocol. The concept of quality is discussed in detail with respect to fishery monitoring in Allard and Benoît (2019) and is reviewed briefly later in this report.

In the present document, dependability is defined as the adequacy of the quality of an estimate or a decision on compliance to a limit to reach a correct conclusion about that estimate or decision in light of the level of conservation risk associated with it. For example, a given degree of quality associated with a monitoring program designed to verify compliance with a total allowable catch (TAC) could be undependable if the population is in the critical zone of the Precautionary Approach and is at risk of overfishing, but dependable if the population abundance is at a historically high level. Uncertainty in estimated removals in the former case can to lead to unintended overfishing detrimental to stock rebuilding objectives, while such harm is not expected in the latter case.

There are three guiding principles in DFO’s fishery monitoring policy with respect to the design of monitoring programs and its implementation to specific fisheries:

- dependability,
- cost-effectiveness, and
- the principle of shared accountability.

The first two principles comprise statistical and scientific concepts and are of interest for the present document.

The first principle seeks to harmonise the level, frequency and type of fishery monitoring with the degree of risk associated with the fishery and the complexity of the fishery (e.g. the number of fleets involved and the nature of in-season management measures). Two classes of risk are considered by the policy: risks to the conservation of aquatic populations, species, biotic communities and habitat, and risks associated with compliance of fishers to fishery regulations.

While the two are not independent in practice, this report focuses on the former and in particular to conservation risks for populations and species. The policy aims to take a precautionary yet pragmatic approach to establishing monitoring needs by recognizing that the quality of estimates should be commensurate with conservation risks posed by the fishery (Babcock et al. 2003; Martin et al. 2015). A fishery that poses heighted risk to the conservation of one or more populations should be monitored in such a way to ensure a high likelihood of deriving catch estimates of sufficient quality to support high quality science and effective management actions. In contrast, the sustainable management of a fishery that poses a low risk to conservation can be supported by estimates of lower quality.

The second principle recognizes the need to consider cost-effectiveness, in addition to quality and risk, and provides flexibility in defining a suitable monitoring program A monitoring program with lower cost monitoring options that result in lower quality estimates may remain dependable provided that conservation risks are also lower. Therefore, monitoring costs can be reduced by reducing risks to conservation (e.g. by decreasing allowable removals or decreasing the likelihood and magnitude of incidental capture of non-target species via avoidance or enhanced selectivity of fishing gear).

Implementation of the policy will follow several steps to ensure consistent application and a high likelihood of achieving conservation, monitoring and compliance goals. Three key steps are:

- the screening of conservation risks and the quality assessment of the existing monitoring programs, which comprise one or more monitoring tools;
- the determination of monitoring objectives related to conservation, compliance and other factors to identify and address gaps where they exist in the assessment; and,
- the specification of monitoring requirements, i.e. the cost-effective combination of monitoring tools and sampling (coverage) levels that will provide estimates that are dependable.

Scientific analysis and advice are integral to implementing these steps as they relate to conservation objectives and risks. The assessment involves an evaluation of conservation risks, an assessment of the quality of catch estimates required for the scientific evaluation of population/species status and for management (e.g. removals with respect to allowable catch), and an analysis of the extent to which quality is commensurate with risk (gap analysis). Two methodologies with corresponding implementation tools are required to aid in the retrospective assessment and, if required following the gap analysis, the determination and specification of new or additional monitoring requirements. The first methodology, implemented in the Quality Assessment Tool (QAT), provides means to assess the quality of estimations obtained from monitoring programs. The QAT has undergone national peer review under the Canadian Science Advisory Secretariat (CSAS) (Allard and Benoît 2019; DFO 2019) as well as some testing and validation in a workshop setting. The second methodology, implemented in the Risk Screening Tool (RST), provides means to characterize risk in a semi-quantitative manner, and specifies estimation quality requirements according to risk levels. Herein, QAT and RST will refer to the methodology or the tool, according to the context.

The main objectives of this research document are to provide the background material and the scientific advice required as a basis of the RST, to determine modifications required to the QAT to harmonise it with the RST and to define dependability in the context of the RST and the QAT.

The present document proposes modifications and improvements to the existing draft RST (Appendix I), and to a lesser extent, to the QAT to allow their application in the implementation of the fishery monitoring policy. First, the considerations (descriptors) used to characterize conservation risk consequences in the draft RST require some updating and, most importantly, need to be subjected to peer review to ensure their relevance, completeness and clarity.

Second, the target quality characteristics in the draft RST require refinement and specific linking to the QAT. Third, slight modifications to the QAT are required to provide measures of quality specifically designed for the context of conservation risk.

DFO's Fisheries and Harbour Management Sector requested advice from DFO Science to address these gaps. A national CSAS advisory process took place from May 14-16, 2019 to address the following three objectives:

- Review the descriptors and methods within the draft RST for assessing and categorizing the risk to target catch and bycatch species through the prosecution of Canadian fisheries. This risk analysis needs to provide categories of risk for target catch, and bycatch (both landed and discarded);
- Provide guidance on quality thresholds appropriate for each of the risk categories used in the RST; and
- Provide advice on how the RST and the QAT can be used to inform decisions on modifying catch monitoring programs so that the required dependability can be achieved.

These objectives respectively relate to characterizing risks to conservations, harmonizing quality and conservation risk and outlining options to ensure consistency in the rigour of catch monitoring programs.

This research document provides the background material necessary for the advisory meeting and the scientific advice that resulted. The objectives above are addressed individually, in turn. In addressing the objectives, we summarize the existing draft RST, as well as the QAT, which is documented in detail in Allard and Benoît (2019). As indicated above we do not address objectives of fishery monitoring that are non-statistical or not directly associated with conservation risks. These include deterrence (e.g. deploying at-sea fishery observers on vessels that are likely to violate regulations or conditions of licence) and regulatory enforcement objectives (e.g. site visits and at-sea boarding by fisheries officers), and factors associated with equitability and shared accountability. Furthermore, we center our review on the monitoring of catch and associated characteristics (e.g. catch ratio limits), and do not explicitly address the use of catch monitoring with respect to the effects of fishing on habitat and communities which is part of the fishery monitoring policy. We note however, that the monitoring of catch of certain benthic habitat-forming species is covered, and therefore impacts on habitats and communities are indirectly covered, at least in part.

2. CHARACTERIZING RISKS TO CONSERVATION

DFO developed the draft RST to harmonise monitoring program requirements with conservation objectives (sustainability of fishery impacts on populations, species, habitats and ecological communities) and regulatory compliance objectives (see summary of the existing RST in Appendix I). In both cases, the RST aims to provide a methodology to evaluate the risk posed by fisheries to meeting the objectives. Risk is evaluated semi-quantitatively as the product of a range of consequences and their associated likelihood. Here we review key aspects of risk determination for conservation objectives related to populations and species (target catch ad bycatch) in the RST for which some modifications would be beneficial. We begin by discussing proposed changes to the general application of the RST and then discuss potential modifications to consequence descriptors.

2.1 THE FOCUS OF THE RST: A FISHERY OR A POPULATION

The draft RST's approach is for a consideration of a set of risks on a fishery-by-fishery basis, while considering cumulative effects. The intended focus is therefore the fishery. This approach is natural given that monitoring programs are often established and operated at the fishery level. It is also appropriate for risk to regulatory compliance and enforcement that are set at the fishery level. Nonetheless, exceptions may exist (e.g. a limit on total removals across all fisheries specified by an international treaty).

For risk to a conservation objective, the cumulative impact from all fisheries relevant to that objective must be assessed. For example, for a population conservation objective, all fisheries that capture individuals from that population as target catch or bycatch or that otherwise meaningfully interact with that population must be included. Correctly gauging the risk posed by fishing on the sustainability of populations requires an assessment of the potential impact of all removals. This is consistent with the practice in stock assessment, where it would not be possible to correctly apportion mortality into fishing mortality and that caused by other sources if an assessment were undertaken fishery-by-fishery and then somehow combined. Evaluating sustainability of fishing would be very difficult.

The QAT was designed to assess quality of estimations or compliance with limits from the population perspective whether for a single fishery or for several fisheries impacting the population. In particular, it can assess the contributions of several monitoring programs to the quality of an estimation process or a compliance with limits for a single population and, therefore, allows a determination of the joint dependability of several monitoring programs with respect to conservation risk concerning that population. This avoids the difficulties of the inverse process, i.e. assessing the risk posed by each fishery affecting a population and then cumulating those risks to obtain the risk to the population. For example, what is the risk due to each fishery in a group of fisheries that each account for 10% of captures of an at-risk population? Most importantly, the QAT facilitates the evaluation of trade-offs in estimation process quality among fisheries and monitoring tools. One can therefore more objectively evaluate, for example, whether a fishery capturing relatively few individuals of a population could be allowed a lower standard of monitoring.

Treating the fishery as the subject of fishery monitoring program reviews is particularly problematic for bycatch. Incidental catch in one fishery is often composed of numerous species, captured in varying numbers. It is not clear how one proceeds with the evaluation of risk from this perspective, as it will be clearly impractical or impossible to assess risk for all incidentally captured species in most fisheries. Should risk be evaluated with respect to the most common incidentally captured species in the fishery, which may not experience an elevated fishing mortality due to that fishery, or with respect to the most biologically and ecologically vulnerable species? With species or populations as the subject of monitoring program reviews, it will be much easier to prioritize species for assessment based on conservation risk.

2.2 CONSEQUENCE DESCRIPTORS

The consequence descriptors for conservation factors related to catch in the draft RST are based on management frameworks under the Precautionary Approach (reference points and harvest control rules), when they are in place, or risk-based considerations of potential impact of fisheries otherwise. The latter draw heavily on DFO scientific advice on risk-based methods for determining sustainable mortality levels of bycatch species (DFO 2012; Pardo et al. 2012). The considerations in that advice are relevant not just to bycatch, but to any population for which there is incomplete data and knowledge on population dynamics and demographic parameters.

2.2.1 Productivity and susceptibility

DFO (2012) considers that the vulnerability of populations to overexploitation can be assessed as a function of their productivity (resilience defined as the capacity to withstand overexploitation or to recover if depleted) and susceptibility to capture and mortality.

The advice identifies natural mortality (M) as a key parameter in developing benchmarks for management of bycatch and other data poor species because it can be used as a proxy for productivity. Species with higher M are likely to be more productive and hence able to sustain higher exploitation rates. Proxies for F_{MSY} (fishing mortality producing maximum sustainable yield) reference points have been proposed as 0.87*M for teleost fishes and 0.41*M for elasmobranchs, and for the fishing limit reference point as F_{lim}=1.5* F_{MSY} (Zhou et al. 2012). These F_{MSY} proxies are used in the RST. Numerous methods are available to estimate M (reviewed in Pardo et al. 2012; Zhou et al. 2012; Kenchington 2014). Equivalent proxies are not available for invertebrates to our knowledge.

Inherent in the use of M for sustainable fishing proxies is the assumption that M represents values for populations at non-depleted equilibrium. However, M may vary temporally and in particular can vary inversely with abundance resulting in depensation at low abundance, as can occur if predation rates on the population increase as abundance declines (Gascoigne and Lipcius 2004; Swain and Benoît 2015; Forrest et al. 2018). Maintaining exploitation rates when M increases, without corresponding increases in recruitment, increases the risk of overexploitation (Legault and Palmer 2016). For populations for which this is suspected to be the case, benchmark values smaller than 0.87*M and 0.41*M should be used in the RST.

The RST presently assesses susceptibility to capture and mortality based only on the relative distributions of the population and the fishery, i.e. availability. It assumes the level of consequence to be low when a substantial portion of the population is not exposed to fishing mortality. However, consequences may be low even if relative spatial overlap is not and the presently used metric may overstate consequence. First, relative distributions need to be considered with respect to time (diurnally, seasonally, etc.) to correctly describe availability. Short durations of overlap, provided they are not during a sensitive biological stage (e.g. during spawning) or a period of aggregation, may not present an elevated consequence unless fishing effort and catchability are high. Second, catchability and selectivity should be considered. Even at high availability, there may be little catch if the catchability to the gear is low. Similarly, consequences of catch may be low for example if the gear selectively catches abundant early life stages. Third, even if animals are captured, the consequence may be low if post-capture mortality is low (i.e. there is a high likelihood of successful live release). Trait and indicator-based approaches are available to gauge the likelihood of post-capture mortality for species that are not retained and landed (Benoît et al. 2010, 2013).

Notwithstanding accounting for availability, aggregative behaviour in a population can contribute to risk of overexploitation. Even if availability is low, small changes in the relative distribution of the population and fishery could result in occasional elevated likelihood of high exploitation if the population is highly aggregated. This is of particular concern as the distribution of most fish species tends to contract as abundance declines, increasing catchability and therefore vulnerability to overfishing (Paloheimo and Dickie 1964; MacCall 1990; Swain and Sinclair 1994).

There are other commonly used risk-based semi-quantitative frameworks to evaluate vulnerability, notably Productivity and Susceptibility Analysis (PSA) and its variants (Stobutzki et al. 2001; Hobday et al. 2007; Patrick et al. 2009; Micheli et al. 2014). These involve a large number of life history traits (e.g. age and size at maturity, fecundity, reproductive strategy) and susceptibility measures, which are combined in a simple ordinal index. However, recent studies have shown that these methods overstate vulnerability (Zhou et al. 2016) and are associated with a high prediction error rate concerning vulnerability when tested in simulations (Hordyk and Carruthers 2019). Instead, vulnerability appears to be most accurately predicted with few characteristics: the intrinsic rate of population increase, availability, selectivity and discard mortality (Hordyk and Carruthers 2019). Noting that the intrinsic rate of population increase is a function of age-dependent M and reproductive schedule (McAllister et al. 2001), the productivity and susceptibility traits in the current draft RST are likely (largely) sufficient to characterize vulnerability for the purposes of establishing the consequences of catch to long-term sustainability.

2.2.2 Other indicators or approaches

DFO (2012) indicates that trends in post-recruitment abundance can provide an indication about whether present catch levels might be impairing the productivity of the population. This indicator is employed in the RST. However, it is important to note that trends should be reviewed with respect to changes in fishery management and objectives, markets, or other factors that may bias the indicator if they are based on fishery-dependent information (e.g. catch-per-unit effort).

The vulnerability of a population can be established via generic age-structured population simulations, rather than by using proxies or indicators. Simulation-based approaches more accurately reflect the interplay of life-history characteristics affecting productivity, and the effects of susceptibility characteristics, compared to PSA approaches and are straightforward to implement (Hordyk and Carruthers 2018). Simulation-based methods should therefore be considered as part of the RST toolbox.

2.2.3 Objectives need to be defined clearly

When considering impact descriptors for catch, it is critical to take into account how catch is used in fishery management. The foregoing assumes that catch is monitored to ensure that removals are sustainable in the short and long term (e.g. that they are below a certain limit). Removals that are high relative to the abundance and productivity of the stock would suggest a need for high quality fishery monitoring. However, there are instances in which catch is monitored to ensure that management procedures that are not necessarily based on catch remain sustainable. For example, input-managed fisheries (based on fishing effort, not removals) do not require catch monitoring for tactical management decisions, but long-term trends in catch or catch properties can inform on whether the input controls lead to sustainable outcomes. In such a case the risks to conservation posed by errors in catch monitoring may be smaller than in output-controlled fisheries, all else being equal. As such, the rigour of monitoring required could be less.

2.2.4 Proposal of new conservation risk consequence descriptors for the RST

Based on the foregoing considerations, the consequence descriptors in the RST were revised and are presented in Appendix II Table 2.1. Attention was paid to highlighting the key elements that distinguish the descriptors among risk categories.

3. ASSESSMENT OF THE QUALITY OF MONITORING PROGRAM ESTIMATIONS

Allard and Benoît (2019) separated the statistical objectives of fishery monitoring programs into two classes requiring different approaches for assessing quality: estimation and compliance with limits. Estimation objectives relate to scientific (e.g. stock assessment and recovery potential assessment) and administrative activities (e.g. reporting on removals and economic value). Compliance objectives are relevant when the management scheme involves some sort of limit (e.g. total allowable catch, the allowable percentage of undersized catch) and the estimate of the parameter is used to determine if the limit has been respected or not.

Allard and Benoît (2019) identified two classes of characteristics that affect the quality of an estimation process: statistical and operational characteristics. We first describe those characteristics and then proceed to define measures of quality.

Allard and Benoît (2019) also propose some measures of dependability independent of the risk to population. These measures are not relevant in the current context where dependability refers to the relationship between the quality of the parameter estimation process or compliance-to-limit decision process with the risk to the population.

3.1 STATISTICAL CHARACTERISTICS

Statistical characteristics describe the impact of the randomness in the random sampling protocol and the impact of the properties of the statistical estimator used to obtain the estimate. In sample surveys, statistical characteristics have an important impact on variability but, in most cases, a small or correctible impact on bias. In censuses, statistical characteristics have no impact.

3.2 OPERATIONAL CHARACTERISTICS

Operational characteristics are related to the implementation of the monitoring program and properties of the estimator or of model-derived estimates. They include either deliberate or unintentional differences between the actual sampling protocol and the sampling protocol assumed in the statistical analysis. Operational characteristics also include factors such as measurement errors and errors associated with calculating the estimates (details below). Operational characteristics can impact both sample surveys and censuses (Allard and Benoît 2019). Impact on bias can be very important but difficult to assess. Impacts on variability are often small and, in some cases, such as with measurement error, are partially accounted for in the statistical characteristics. However, in other instances involving differences in the actual sampling scheme from the one unwittingly assumed by the analyst, there may be a non-negligible under or over-estimation of variability.

It is notable that all past attempts at establishing quality goals for fishery monitoring programs have focussed narrowly on statistical characteristics, typically only with respect to variability or its inverse, precision (e.g. Agnew et al. 2010; Hanke et al. 2012), but in some cases also with statistical bias (e.g. Amandè et al. 2012). Some of these cases involved detailed accounting for variability due to multistage sampling (i.e. sampling with respect to trips, hauls within trips, fish within hauls, etc.; Voldstad et al. 1997; Cotter et al. 2002; Agnew et al. 2010). Other authors have recognized the importance of operational characteristics in affecting quality (e.g. Babcock et al. 2003; NMFS 2004), but were unable to explicitly incorporate these effects in setting objectives for quality as the task is not trivial. Instead, some have advocated testing for or quantifying the contribution of operational characteristics and using those results to plan follow-up actions (e.g. enforcement or modifications to the programs) and to temper the interpretation of the estimates with respect to risks to the resource or fishery in a somewhat ad hoc manner, i.e. be more precautionary (NMFS 2005; Rago et al. 2005; Volstad and Fogarty 2006). In contrast, Babcock et al. (2003) have argued that the influence of certain important operational factors, such as biases (and variability) resulting from non-representative sampling of the fishery and observer effects (a difference in fishing behaviour between observed and unobserved vessels) can be attenuated by increasing sampling rates (coverage levels). This approach has been criticized in part because the contribution of operational factors is unlikely to be directly inversely related to coverage levels (Rago et al. 2005). Furthermore, some operational characteristics can remain relevant even in a census (100% coverage), such as non-reporting bias in logbooks and the variability resulting from visual estimation of catch (Allard and Benoît 2019).

Based on general survey methodology (Groves et al. 2009) and considerations specific to fisheries (Babcock et al. 2003), Allard and Benoît (2019) identified fifteen operational characteristics that can affect the quality of estimates from fishery monitoring programs (for definitions and details see Allard and Benoît 2019).

The first two characteristics, undercoverage (1) and overcoverage (2), address the relationship between the sampling frame used to plan the monitoring and the target population. These characteristics can affect bias in both sample surveys and censuses but are not expected to affect variability.

The next three characteristics apply to sample surveys and relate to how samples are chosen compared to how they are assumed to have been chosen in the statistical calculations and include unintended clustering of samples (3), unintended sampling stratification (4), and other irregular selection probabilities (5; including vessel targeting). These characteristics can have an important effect on variability and potentially also bias.

The observer effect (6), addresses a bias induced by the act of making the observation. There should be no observer effect in a census.

The next two characteristics relate to the influence of missing values, due to unintentional factors (7), including unintentional non-response, and intentional factors (8) including intentional non-response. If missing values are random across samples then there should be no effect on bias. However, if the missing values are systematic in some respect, such as in deliberate under-reporting of catch, then a bias will result. The variability caused by missing values is accounted for in the statistical calculation of the standard error (SE) but reducing the number of missing values does increase quality.

The remaining characteristics relate to unplanned errors and include: errors in data reported by resource users (9; e.g. fishers, fish plants and buyers), errors reported by independent observers (10), equipment error (11; e.g. measuring tool bias and/or imprecision), data handling errors (12), adjustment errors (13; e.g. converting landings from dressed to fresh weights), imputation error (14), and modelling error (15). An overall unaccounted bias will occur if these errors are biased, i.e. they tend to systematically under or overstate the true values. The contribution of these characteristics to variability is accounted for in part or in whole in the statistical SE (see Allard and Benoît 2019).

3.3 MEASURING QUALITY OF A PARAMETER ESTIMATION PROCESS

Allard and Benoît (2019) measure quality by first measuring the contribution of statistical characteristics and a suite of operational characteristics to each of the two components of quality: bias and variability. These calculations ideally involve existing data but can also involve expert opinion for typical values.

Biases associated with statistical and operational factors are considered additive. The result of these computations is the estimation process bias denoted by b_{ep}.

The standard error (SE) is the basic measure of variability employed. The SE obtained from statistical analysis reflects mostly the variability of the estimator due to the randomness of the sampling protocol. The impact of departures from the sampling protocol (operational characteristics) on variability not otherwise reflected in the SE are described by multiplicative corrections to the SE. For example, operational factors that cause the SE to be underestimated are associated with a correction factor greater than one. Impacts of operational characteristics representing added variability in the data are added quadratically to the SE. The result of these computations is the estimation process variability denoted by s_{ep}.

The measure of the quality of the estimation process proposed by Allard and Benoît (2019), termed estimation process error (e_{ep}), combines the bias and the variability using a formula heuristically based on the root mean-square error and is defined as:

$$e_{ep} = \sqrt{b_{ep}^2 + s_{ep}^2}.$$

Subsequent to the finalization of Allard and Benoît (2019) and the related science advisory report (DFO 2019), a workshop was held in December 2018 to test and apply the QAT. An important conclusion of that workshop was that quality should be assessed with respect to variability and bias separately, rather than jointly in the estimation process error. This conclusion was motivated by the fact that estimation process errors caused by variability and bias have different consequences to conservation risk. Variability comprises random errors such that parameters like total catch are equally likely to be underestimated or overestimated in a given year. In contrast, bias comprises systematic errors resulting in a repeated under or overestimation. Over time the errors caused by bias will compound, leading either to systematic loss of fishing opportunities in the case of positive bias, or to systematic over-fishing in the case of negative bias. The present document reflects this change.

We will use the following notation.

- θ: the true value of the parameter being estimated by the parameter estimation process, for example, the total catch;
- b_{ep}: the estimation process bias as defined above;
- s_{ep}: the estimation process variability as defined above;
- $\hat{\theta}$: the typical value of the parameter, as obtained from the estimation process; and
- $\theta_{anticipated} \approx \hat{\theta} - b_{ep}$: the anticipated true value of the parameter.

The two measures of quality for an estimation are the relative estimation process bias and the estimation process variability:

- $rb_{ep} = b_{ep}/\theta_{anticipated}$
- $rs_{ep} = s_{ep}/\theta_{anticipated}$

Note that:

- When $rb_{ep} = 0$ and $rs_{ep} = 0$, this indicates perfect information.
- High absolute values of rb_{ep} and rs_{ep} correspond to low quality.
- rb_{ep} is a signed value; a negative value indicates that the estimation process tends to underestimate the value while a positive value indicates the opposite.
- rs_{ep} is similar to the coefficient of variation.

3.4 MEASURING QUALITY FOR A DECISION ON COMPLIANCE TO A LIMIT

In the following, we suppose that the limit is an upper limit on a quantity (e.g. a total allowable catch) which is the most common case; the lower limit case is symmetrical. The case for a limit based on a proportion is similar if the limit is away from 0% or 100%. Different formulae are required for limits on proportions close to 0% or 100% or for limits on the numbers of rare events.

Heuristically, this approach is based on a statistical test to reject the hypothesis that the limit has been reached or breached, assuming that the error of the estimate process follows a normal distribution. The concepts of statistical tests of significance are only used as a guide. The development depends on verifying only roughly the assumption required to apply the significance tests (e.g. that the distribution of the estimator is approximately symmetrical with a single mode). Special cases for other distributions, such as the Poisson, can be elaborated (Allard and Benoît 2019), as discussed below for rare event cases.

We use the following vocabulary based on detection of non-compliance with the limit (not to be confused with non-compliance to regulations):

- A "false positive" refers to concluding that the limit has been exceeded when in fact it was not.
- A "false negative" refers to concluding that the limit has been respected (not exceeded) when in fact it was exceeded.

False negatives are the focus of the following discussion given that their conclusions are detrimental to the conservation of aquatic populations, the primary concern addressed in this report.

We use the following notation concerning the limit.

- L: the true upper limit required to meet the objective, e.g. the total removals that will respect the objective for the population, such as allowing the stock to grow out of the critical zone within a stated timeframe and at a stated probability;
- $\hat{L}$: the estimate of the limit, obtained from the scientific process, i.e. the estimated removals that are expected to respect the objective with a stated probability and within a stated timeframe, based on stock assessments, population models, etc. ;
- $b_{\hat{L}}$: the bias of the limit obtained from the scientific process;
- $\varepsilon_{\hat{L}} = \hat{L} - (L - b_{\hat{L}})$: the error on the limit obtained from the scientific process; and
- $\sigma_{\hat{L}} = \sigma_{\varepsilon_{\hat{L}}}$: the uncertainty of the limit obtained from the scientific process.

Currently, the uncertainty on a limit is rarely included in the decision making process, i.e. $b_{\hat{L}} = \sigma_{\hat{L}} = 0$. However, it is generally agreed that such uncertainty exists and that it is desirable to include it in risk-based decision making. This uncertainty can be estimated within a single assessment model as well as with respect to several competing plausible models for a stock. We propose that $\sigma_{\hat{L}}$ be estimated by taking the median absolute deviation (MAD) of the values $\hat{L}$ obtained by competing models and, possibly, different inputs to those models.

Estimating $b_{\hat{L}}$ may be difficult and as such $b_{\hat{L}}$ should be set to 0 unless a formal retrospective review of predictions versus actual results produces a different value.

We expand the notation defined in the previous section:

- θ: the true value of the parameter being estimated by the parameter estimation process, for example, the total catch;
- b_{ep} and s_{ep} are respectively the estimation process bias and estimation process variability;

- $\hat{\theta}$: the typical value of the parameter, as obtained from the estimation process;
- $\theta_{anticipated} \approx \hat{\theta} - b_{ep}$: the anticipated true value of the parameter;
- $b_{\hat{\theta}} \approx b_{ep}$: the bias of the estimation process;
- $\varepsilon_{\hat{\theta}} = \hat{\theta} - (\theta + b_{\hat{\theta}})$: the error of the estimation process around its expected value; and
- $\sigma_{\hat{\theta}} = \sigma_{\varepsilon_{\hat{\theta}}} \approx s_{ep}$: the variability of the estimation process.

Values for the estimation process bias $b_{\hat{\theta}} \approx b_{ep}$ and the estimation process variability $\sigma_{\hat{\theta}} \approx s_{ep}$ are be obtained by applying the results of Allard and Benoît (2019), i.e. by calculating the values that result from the statistical and operational considerations as described above.

Consider the one-sided statistical hypothesis test H_0: $\theta \geq L$ vs H_1: $\theta < L$. The limit will be considered satisfied if the null hypothesis H_0 is rejected.

A false negative as defined above (i.e. concluding that the limit has been respected when in fact it was exceeded) means to reject H_0: $\theta \geq L$ when $\theta \geq L$, i.e. a type I error in statistical vocabulary.

Notice that $\theta = \hat{\theta} - b_{\hat{\theta}} - \varepsilon_{\hat{\theta}}$ and $L = \hat{L} - b_{\hat{L}} - \varepsilon_{\hat{L}}$.

Let φ be the cumulative distribution function of the standard normal distribution.

Adapting the approach in Allard and Benoît (2019) to the objective of conservation, we define the following measure of quality of an estimation process for compliance applications.

We consider heuristically the probability that the statistical test H_0: $\theta \geq L$ vs H_1: $\theta < L$ yields the correct conclusion, for $\theta = \theta_{anticipated} = \hat{\theta} - b_{ep}$ and $\sigma_{\hat{\theta}} = s_{ep}$ and $L = \hat{L} - b_{\hat{L}}$ and $\sigma_{\hat{L}}$.

We define the measure of quality of the parameter estimation process for compliance applications heuristically as:

$$\textit{The probability of avoiding a decision detrimental to conservation}$$

This probability is computed as follows:

- If $\theta_{anticipated} < L$, the quality of the parameter estimation process is equal to 1.
- If $\theta_{anticipated} \geq L$, the quality of the parameter estimation process is:

$$1 - \left[\textit{the power of the test at } \theta = \theta_{anticipated} \textit{ and } L\right] = 1 - \varphi\left(\frac{L + b_{\hat{L}} - (\theta + b_{\hat{\theta}})}{\sqrt{\sigma_{\hat{L}}^{2} + \sigma_{\hat{\theta}}^{2}}}\right).$$

The definition for $\theta_{anticipated} < L$ differs from that of Allard and Benoît (2019) to take into account the fact that conservation is the unique objective.

3.5 QUALITY ASSESSMENT BASED ON SEVERAL MONITORING PROGRAMS

The framework developed by Allard and Benoît (2019) recognizes that several monitoring programs may be implicated in estimating a parameter of interest.

In some instances, parameters from several estimation processes are summed to estimate a final parameter of interest such as total catch of a given population, across fleets or fisheries. Assuming that the total is obtained as a sum of the individual estimates, then:

- the bias of the overall estimation process for the total is the sum of individual estimation process biases; and,
- the variability of the overall estimation process for the total is the square root of the weighted sum of the squared individual estimation process variability.

Weights may be required to reflect the relative contribution of each estimation process to estimating the parameter of interest. For example, if bycatch of a species in several fisheries is estimated as a ratio to the total catch of the target species in each of these fisheries, the weight of the catch of the target species must be used as mathematical weights in the addition of the ratios.

When a parameter is estimated by summing the contribution of several fisheries, it may be useful to include the contribution of fisheries for which there is no monitoring program. In such situations, the QAT can used to assess the impact of the absence of a monitoring program to the quality of the parameter estimation process as long as some minimal information is available. For example, an estimate of removals by a non-monitored fishery may be available from past monitoring programs, from a similar fishery that is monitored, or from people familiar with the fishery. Such an estimate can be used as an imputation in the QAT, with a suitable assessment of the imprecision of this imputation. When summing this estimate of removals with the estimates of removals by the monitored fisheries, an assessment of the impact of the non-monitoring on the quality of the overall estimation will become available.

In other instances, a final parameter of interest is a product of two or more estimation processes, such as the estimation of catch from separate monitoring programs that estimate fishing effort and catch-per-unit effort. In these instances, the overall quality of the estimation process is obtained by applying, heuristically, the formulae for the bias and variance of a product of two independent random variables (for details see Allard and Benoît 2019).

3.6 ASYMMETRICAL SAMPLING DISTRIBUTION, RARE EVENTS, PRESENCE-ABSENCE

In the following discussion, we neglect the impact of operational characteristics on the bias and the variability of the estimator. In actual applications, they must be included.

For situations where the sampling distribution of the estimator is not symmetrical, the requirements on the variability should be stated in terms of the confidence interval (c.i.) with confidence level 0.683 (reflecting one unit of standard error for the normal distribution), for compatibility with the requirement for the symmetrical case above and with different requirements for the left-hand-side and the right-hand-side depending on the relevant direction of the error (see Allard and Benoît (2019) for the justification for this choice). Such a situation will occur in rare-event cases. In some cases, the confidence interval can be computed using analytical or numerical methods whereas in other cases it can be estimated using simulation. Furthermore, in rare-event cases, it may be more appropriate to use absolute values (e.g. the number of events or animals) than the relative values to describe precision and bias.

The following examples illustrate these concepts.

- Consider a situation where the objective is to estimate the number of trips for which a certain rare event has occurred. Suppose that the constant probability of the event occurring on a given trip is p. If n trips are observed, then p is the parameter of the binomial distribution $B(n,p)$. Then, the confidence interval can be computed using one of several methods (all giving similar c.i.'s).

Suppose that a rare event occurred on 2 trips out of 1,000 observed trips. The unbiased estimator of p is $2/1000 = 0.002$ and the confidence interval (c.i.) with confidence level 0.683 is $[0.001, 0.004]$ and the left and right-hand side relative errors are $(0.001 - 0.002)/0.002 = -50\%$ and $(0.004 - 0.002)/0.002 = 100\%$.

- Consider a situation where the objective is to estimate average catch numbers per trip for a rarely caught but at-risk species. Suppose that the average catch number is λ. Then λ is the parameter of the Poisson distribution (if the data is not under- or over-dispersed). Again, the confidence interval can be computed as follows:

  Suppose that 2 animals were caught during 1,000 observed trips. The unbiased estimator of λ is $2/1000 = 0.002$ and the c.i. with confidence level 0.683 is $[0.0007,\ 0.0046]$ and the left and right-hand side relative errors are $(0.0007 - 0.002)/0.002 = -64.6\ \%$ and $(0.0046 - 0.002)/0.002 = 132.0\%$.

- A small sample occurs when the sample is less than approximately 20 and the population distribution asymmetrical and/or multimodal.

  Suppose that the following size 10 sample was obtained: $[0\ \ 2\ \ 3\ \ 3\ \ 4\ \ 88\ \ 96\ 103\ 104\ 113]$. The sample is highly bi-modal, and the central limit theorem does not apply. The estimate of the mean is 51.6. Using a bootstrap method, the 0.683 c.i. is [37.9, 70.6]. The left and right-hand side relative errors are -26.6% and 36.8%. The bootstrap can also be used to estimate the statistical bias which is -1.8% for this sample.

- Another asymmetrical situation occurs when it is only desired to know whether individuals from a population are captured, while the specific amounts are of lesser concern. In this case, the quality is better stated in terms of test on probability p that any individual be captured with $H_0: p = 0$ and $H_1: p \geq p_1$ where p_1 is the smallest probability that is meaningful for conservation decisions (e.g. $p_1 = 0.001$). The treatment in this case is therefore similar to the compliance applications discussed below.

3.7 NOTE: QUALITY VS MONITORING TOOL COVERAGE

Monitoring requirements for programs involving sample surveys, such as at-sea observer programs, are typically specified in terms of target sampling or coverage rates, i.e. the percentage of trips for which an observer will be deployed. However, it is incorrect and misleading to assess quality from coverage rates alone (Haigh et al. 2002). Even in the absence of operational characteristics affecting quality, and ignoring bias, the link between coverage rate and variability is not linear. This can be illustrated by considering the equation for the SE of the estimator of the population mean under simple random sampling:

$$\sqrt{1 - n/N}\, \sigma/\sqrt{n}$$

where *n* is the sample size (e.g. the number of observed trips), *N* is the population size (e.g. the total number of trips for a fishery and year), σ is the population standard deviation, and $\sqrt{1 - n/N}$ is the finite population correction factor. From this equation it is evident that the SE is a function of σ, *n* and the coverage rate (*n/N*). However, the factor $\sqrt{1 - n/N}$ is negligible (close to 1) unless the coverage rate is greater than around 30%. Therefore, in a large fishery involving many trips (high *N*), quality may be considered high (i.e. SE small) if there is little variability in catch characteristics among trips (small σ) or if the sample size is large (noting that the estimator for σ is also an inverse function of sample size), even if the coverage rate is small. Conversely, an elevated coverage rate (e.g. 50% coverage giving $\sqrt{1 - n/N} = 0.71$) does not guarantee elevated quality since σ can be large.

4. DEPENDABILITY: HARMONISING QUALITY REQUIREMENTS AND RISK

The dependability of a parameter estimation process supported by one or more monitoring programs depends on its quality with respect to estimation or to compliance-with-a-limit decision and the levels of acceptable conservation risks and risks to the fishery in terms of possible foregone opportunities (Rago et al. 2005). With respect to conservation risk, this amounts to the risk of population collapse if management actions are compromised by invalid (inaccurate and/or imprecise) estimates of catch. In estimation applications, the assessment of dependability relies in effect on specifying objectives for risk-specific acceptable quality. For applications to decisions on compliance with limits, the assessment relies on specifying a risk-dependent minimum on the probability of avoiding a decision detrimental to conservation (e.g. deciding that a TAC was respected when it was not). In turn, the limit may also be set in such a manner as to account for risks to conservation (presented later in the document). For example the total allowable catch (TAC) for a population that is in the critical zone of the Precautionary Approach will likely be set such that it allows for a high probability of stock rebuilding to a level above the limit reference point in a defined (short) period of time, while the TAC for a stock in the healthy zone may be associated with a neutral probability (0.5) of decline (DFO 2009).

In the context of the RST and QAT, we define a parameter estimation process to be dependable if its quality is appropriate for the level of risk associated with the parameter. For example, the quality of a monitoring program designed to verify compliance with a TAC should be high if the population is in the critical zone of the Precautionary Approach but can be low if the population abundance is at a historically high level.

The purpose of this section is to propose minimum quality requirements (thresholds) as a function of the risk level assessed by the RST, where quality is defined by the QAT. The choice of thresholds cannot be entirely objective. Consequently, threshold values for dependability are provided as guidance rather than absolute values. The choice of threshold values may change as new information is obtained on performance and improvements are made to monitoring programs. The values presented below are based on a mixture of considerations related to effect or signal detection and achievability.

The following quality requirements apply not only to situation for which the QAT has been developed but also to situations not covered by the QAT, e.g. estimation of proportions close to 0 or 1, of number of rare events, etc. or compliance-to-limits decision about such quantities. In those situations, the quality requirements proposed below are applicable, but the mathematics required to assess the quality is outside the scope of the QAT.

4.1 ESTIMATION APPLICATIONS

To simplify the link between quality and risk, we propose risk-specific thresholds for rb_{ep} and rs_{ep}, the relative bias and the relative variability of the estimation process, respectively. Because rb_{ep} and rs_{ep} are relative measures, a single set of thresholds with respect to conservation risk is sufficient.

4.1.1 Variability

Errors due to variability can be detected (e.g. by repeating a survey) and, by definition, should average to 0 over time. At least over the long run, they should be less detrimental to conservation than errors due to bias.

The QAT measure of variability is heuristically similar to the coefficient of variation.

For censuses and for sampling surveys where the sampling distribution of the estimator is approximately symmetrical and the sample size is above approximately 20, we propose the requirements on estimation process variability obtained by QAT as laid out in Table 1.

Table 1. Parameter estimation process variability (rs_{ep}) thresholds according to conservation risk categories, for situations where the sampling distribution of the estimator is approximately symmetrical.

Component	High conservation risk	Medium conservation risk	Low conservation risk
RST statement	High likelihood of determining if objective is met	Reasonable likelihood of determining if objective is met	Adequate to determine if objective is met
Threshold values	$rs_{ep} \leq 15\%$	$rs_{ep} \leq 30\%$	$rs_{ep} < 50\%$

Parameter estimation variability thresholds were chosen to match the expectations stated qualitatively in the RST (Appendix I) that monitoring programs should provide adequate information to estimate catch when risk is low, a reasonable likelihood of ‘correctly’ estimating catch when risk is medium, and a high likelihood when risk is high. Parameter estimation variability comes from operational characteristics that create measurement imprecisions and, in sampling surveys, from the sampling randomness. In sampling surveys, parameter estimation variability can be reduced by increasing sample size, even though it may be more cost efficient to use other approaches aimed at addressing operational characteristics (e.g. strict adherence to pre-specified sampling plans, increase observer training or the use more precise scales).

The variability threshold for high conservation risk ($rs_{ep} \leq 15\%$) should be generally achievable in monitoring programs based on censuses or from a sampling survey with very large sample size. It may also be achievable in programs with smaller sample sizes but for which there is little potential for added variability resulting from operational characteristics. For example, parameter estimation variability values at or just above the threshold (~20%) were obtained in the Pacific groundfish video-monitoring program with sample sizes in the hundreds (Stanley et al. 2009).

The variability threshold value for intermediate risk was set in light of the target 20-30% coefficient variation (CV) value used for US fisheries at-sea observer programs based on extensive review (NMFS 2004). CV values consistent with those thresholds are achievable for many species and fisheries in US observer programs (Wigley et al. 2007) and should likewise be achievable in Canada given similarity in the programs in terms of sample sizes and sampling schemes. Nonetheless, there are two considerations that argue for a target value that might be different from the US-NMFS values. First, the US-NMFS threshold values are for individual fisheries, unlike the ones for the present framework which are for populations, potentially caught by many fisheries. The realized CVs in the present framework, for the sum of the catches, could be smaller. Second, the US-NMFS threshold values do not account for the contribution of operational characteristics to variability, unlike the DFO framework. Based on the Gulf of St. Lawrence skate discard case study presented in Allard and Benoît (2019), this contribution could increase variability by 35% or more, resulting in an overall parameter estimation variability of 33% in that specific case. A threshold value of 30% was therefore chosen in light of these considerations.

The variability threshold value for the low risk case was chosen to bracket estimates by plus or minus 100% error (or a bracket of –50% to 200% if a log scale is used). Such values should be achievable for many populations sampled even in pilot (e.g. Benoît 2011; Pezzack et al. 2009) or small scale at-sea observer programs (Rutherford et al. 2009).

Finally, in contrast with what was proposed in the present context for removal estimates by Allard and Benoît (2009), we now propose to simplify the dependability assessment for rare events by considering only the length of the upper confidence interval, as it is the one that most concerns conservation risk. This is consistent with the practice in the US (Appendix 4 in NMFS 2004).

4.1.2 Bias

Errors due to bias may be very difficult to detect and their impact will accumulate over time. They could be much more detrimental to conservation than errors due to variability. Therefore, we propose that requirements on bias be stricter than those on variability.

With respect to bias, we propose the following maximum relative values, in the relevant direction (Table 2).

Table 2. Parameter estimation process bias (rb_{ep}) thresholds according to conservation risk categories, for situations where the sampling distribution of the estimator is approximately symmetrical.

Component	High conservation risk	Medium conservation risk	Low conservation risk
RST statement	Theoretically unbiased	Bias should be limited	Not specified
Direction of bias relevant to conservation risk			
Negative: Underestimating the parameter can be detrimental to conservations objectives	$0\% \leq rb_{ep}$	$-10\% \leq rb_{ep}$	$-25\% \leq rb_{ep}$
Positive: Overestimating the parameter can be detrimental to conservations objectives	$rb_{ep} \leq 0\%$	$rb_{ep} \leq 10\%$	$rb_{ep} \leq 25\%$
Both: Overestimating or underestimating the parameter can be detrimental to conservations objectives	$rb_{ep} = 0\%$	$-10\% < rb_{ep} \leq 10\%$	$-25\% < rb_{ep} \leq 25\%$

These values reflect proposed acceptable bias following any bias corrections applied during the parameter estimation procedures. The values are also in-line with the draft RST, which indicates that at high risk, a monitoring program should have a design that is theoretically unbiased (and presumably also in practice) and at medium risk bias should be limited. For catch estimates for a stock at medium risk, a 10% negative bias means that actual annual removals will be 11.1% higher than estimated; during a period of low productivity, underestimating removals by 11.1% annually over several years may have an important cumulative impact. For catch estimates for a stock at low risk, a 25% negative bias means that actual annual removals will be 33.3% higher than estimated. Even in a low risk situation, such an error may become consequential over time.

Most of the parameter estimation bias is expected to come from operational characteristics of the monitoring programs, as opposed to the statistical characteristics. Most biases cannot be corrected by increasing sample size even up to 100% (i.e. to a census – details in Allard and Benoît 2019). For high conservation risk cases, parameter estimates that are unbiased or biased in a direction not detrimental to conservation (e.g. a monitoring program overestimating removals) should be generally achievable in monitoring programs in which sources of bias from operational consideration are minimal such as in many video-based monitoring and well audited programs (Beauchamp et al. 2019). In contrast, biases are expected in at-sea observer surveys and often in programs that depend on data provided by the resource user (Beauchamp et al. 2019). For example, comparisons of landed and observed catches from at-sea observer programs suggest that biases on the order of 5-25% can be occur for taxa for which there are incentives for fishers to misreport or misrepresent catches (Benoît and Allard 2009; Faunce and Barbeaux 2011), while there may be no biases when there are no such incentives (Benoît 2013).

4.1.3 Compliance-to-a-limit applications

In many situations, a limit to the catch or to some fishery characteristic is implemented in order to meet a conservation goal. The limit can be on target species catch, on the bycatch of specific species, on the proportion of specific individuals (e.g. softshell crabs in the snow crab fishery), etc. While this section is worded in this context, the methodology is applicable to limits related to other objectives (e.g. respect of resource sharing agreements or of international treaties).

In many cases, a monitoring program is a component of a decision making process that involves several elements, often including a limit on a parameter (e.g. the TAC) established following a stock assessment to estimate parameters and reference points (e.g. FMSY) and of some rules (e.g. a harvest control rule) or procedure to obtain the limit. Finally, there is a decision rule to manage the fishery (e.g. close the fishery if the estimate of the total catch given by the monitoring program has reached the TAC).

The quality of a parameter estimation process in the context of a decision on compliance to a limit is measured as the (heuristic) probability of avoiding a decision detrimental to conservation, i.e. concluding that the limit has not been breached when in fact is has. Therefore, values closer to 1 are preferable. If the typical true values (i.e. after correction for statistical and operation biases) of the parameter are well below the limit, the quality is deemed to be 1. However, the quality will tend to be low if the typical true value of the parameter is close to the limit. For fisheries fishing to the TAC, a common case, high quality will be reached only if the parameter estimation process has nearly zero bias and variability.

Limits are a convenient and necessary tool in fishery management. However, the consequences of a wrong decision on compliance to a limit are impacted by the quality of the estimation process. If the estimation process is very accurate and precise, a wrong decision may mean that the limit was breached by only a small percentage and inconsequential, while if it is very inaccurate and/or imprecise, the breach may be very consequential.

Thresholds for the quality of compliance to a limit (Table 3) were based on general risk-based considerations. We propose the required quality of the parameter estimation process for the operational limit shown in Table 3.

Table 3. Thresholds for quality (0 to 1) of compliance to a limit for three conservation risk levels defined from the RST.

Risk	High conservation risk	Medium conservation risk	Low conservation risk
Required quality	≥ 0.95	≥ 0.75	≥ 0.50

For low conservation risk cases, accepting a 50% (or risk neutral) probability of an incorrect decision is reasonable. This is consistent with the current approach for fisheries fishing up to the established TAC or catch limit, assuming that the parameter estimation process is not biased. In contrast, for high risk cases, a high probability of reaching the correct decision is appropriate, and a value of 0.95 is consistent with what is routinely considered high probability in statistics and risk-based decision making. A threshold value approximately intermediate to those for low and high risk was chosen for medium risk.

4.2 DEPENDABILITY WHEN MULTIPLE MONITORING PROGRAMS ARE INVOLVED

Individuals from a single population or stock are often captured by more than one fleet or in more than one fishery. Estimating total catch therefore requires summing estimates from multiple monitoring programs. Even abundant populations that are commonly caught in fisheries will be captured infrequently or in small amounts in some fisheries because their availability and/or catchability to those fisheries are low. Setting uniform monitoring variability and bias objectives for such populations across fisheries and fleets would be cost-ineffective and would confer little conservation benefit. For a given level of desired variability, required sample size for species that are rare in the catch will typically be much higher than for common species (e.g. Babcock et al. 2003; Gilman 2012; Wakefield et al. 2018). Monitoring costs would therefore also likely be much higher. Meanwhile the incremental risk to conservation posed by these fisheries may be small or negligible. It is therefore important that monitoring quality objectives remain commensurate with the contribution to the conservation risk of each individual fishery.

5. THE ASSESSMENT PROCESS

The process of determining, evaluating, and revising fishery monitoring programs involves the use of the risk screening tool (RST) and the quality assessment tool (QAT) in a recursive process aimed at harmonising conservation risk and estimation process quality. In this section we outline this process as illustrated in Figure 1.

The process will normally begin with an assessment of risk for a specific conservation objective (e.g. a population biomass declining below a certain critical level). The RST will be used to assess the impact of all fisheries that capture individuals from that population as target catch or bycatch or that otherwise meaningfully interact with that population (e.g. capture of prey, destruction of spawning grounds). In many cases, only one or a small number of fisheries will be relevant while, in some cases, many fisheries could be relevant. To establish a population level risk, the RST will take into account the contribution of each relevant fishery to the risk for the specific conservation objective. If a fishery specific risk score is desired, the fishery contribution to the population level risk must be calculated.

When one or only a few fisheries are relevant, the required quality can be compared directly to the results of the QAT for the monitoring of these fisheries. When many fisheries are relevant, it may be necessary to use the QAT to assess the quality of the monitoring tools jointly, using the QAT's facility to evaluate quality with respect to one or more monitoring programs across all fisheries that capture a species (Allard and Benoît 2019).

If the quality of some of the monitoring programs are found to be insufficient, the monitoring programs may be improved. However, in some cases, improvement to the monitoring programs may be impossible or excessively costly or other sources of uncertainty too large to allow a reduction of the conservation risk. In those cases, the fishery management decision process may have to be revised (e.g. reduce the TAC). Possible interventions are described in the next section of the document.

For a risk in a compliance-with-a-limit application, in many cases, only one fishery will be relevant (the fishery subject to those regulations). The assessment for a single fishery is based on the anticipated value for the catch and the limit set for that fishery, or the fraction of the overall limit that is expected for that fishery.

The assessment across several fisheries is similar, but also involves an assessment of the quality across fisheries. In risk to a conservation objective and risk to compliance with a limit when more than one fishery is relevant, the advantage to considering dependability across fisheries is that it allows for the possibility of trading-off requirements for quality between fisheries, such that for example, monitoring for one fishery that produces estimates with high variability may be deemed acceptable if monitoring in other fisheries produces precise estimates. Such tradeoffs are also possible for estimation applications. The calculation would involve weighting fishery-specific acceptable parameter estimation bias and variability values by fishery-specific anticipated catch and ensuring that the average corresponds with overall variability and bias requirements for the assessed level of risk.

When a single fishery is relevant to several conservation objectives and/or compliance limit objectives, the required quality of the monitoring program would normally be determined by the most demanding among them in the risk assessment.

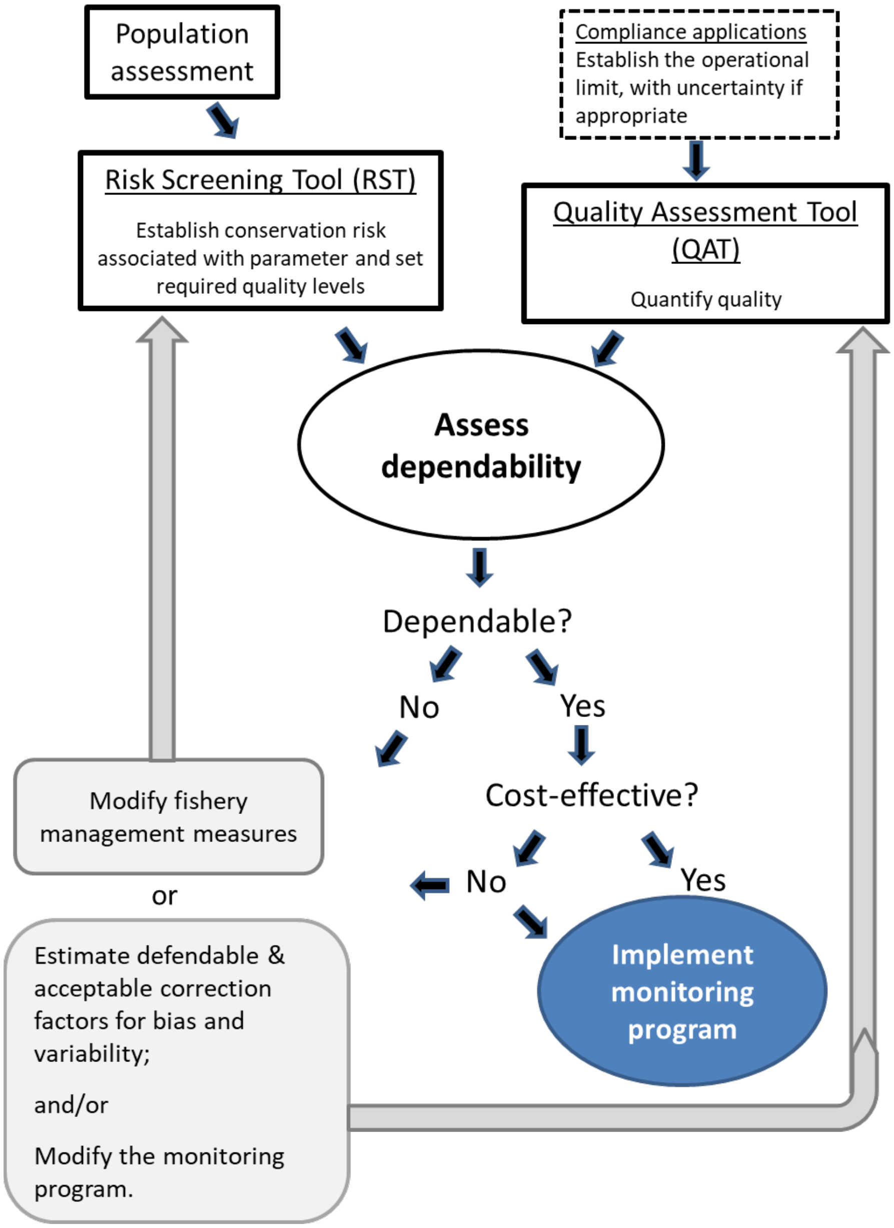

Figure 1. Flow diagram for the implementation of the Fishery Monitoring Policy with respect to conservation risk. The diagram illustrates the principal flows, however in practice many of the processes will be more integrated. The process begins with the establishment of conservation risk using the revised Risk Screening Tool, followed by the application of the revised Quality Assessment Tool. Results from the dependability analysis may motivate modifications to the fishery management plan and/or the monitoring program, followed by a re-assessment of dependability and possibly quality or conservation risk.

5.1 DEPENDABILITY OUTCOMES

There are three possible determinations for monitoring programs that result from assessing dependability via the QAT: dependable and cost efficient, dependable but cost inefficient, and not dependable.

5.1.1 Dependable and cost-efficient

A monitoring program that is found to be dependable and cost efficient can be implemented as designed. A future reassessment of dependability is only warranted if:

- there is a change in the assessed status of the population or fishery management plan that result in a change in conservation risk;
- there is a desire or need to modify the type or sampling intensity of one or more monitoring program; or,
- there is new information to inform the assessment of estimation process quality, for example to specify the effects of operational characteristics on bias and variability.

5.1.2 Dependable but cost-inefficient

Monitoring programs that exceed quality requirements for dependability may be cost-inefficient from the perspective of conservation risk. Of course, the monitoring programs may be required to address complexities in the fishery, such as monitoring of catch for individual transferable multi-species quotas. In the absence of such constraints, fishery managers and stakeholders may consider alterations to the monitoring programs that reduce cost while not undermining estimation process quality requirements such that dependability is compromised. This could involve reducing sampling (coverage) rates or changing the monitoring tools employed to favour lower cost options that do not unduly compromise quality. For reviews of monitoring tool options with respect to quality and cost-effectiveness see Mangi et al. (2015) and Beauchamp et al. (2019).

5.1.3 Not dependable

There are two principal options for the fishery monitoring policy when a monitoring program is deemed not dependable: modify the monitoring program(s) to improve quality (NMFS 2004) or decrease the conservation risk and therefore lower the quality requirements for the estimation process. We explore these options in the next section.

6. IMPROVING DEPENDABILITY

When an estimation process is deemed not dependable, either the monitoring program or the risk must be modified to achieve dependability.

In this section, we present some approaches that can be used to increase the quality of a monitoring program. We also present some fishery management approaches to reducing risk since in some cases, reducing risk may be more cost-effective than increasing the quality of monitoring programs.

6.1 IMPROVING THE QUALITY OF THE PARAMETER ESTIMATIMATION PROCESS

There are several options available for modifying a monitoring program to improve quality. Some of these are not specific to the ‘not dependable’ case and may also constitute options for reducing monitoring costs in fisheries for which monitoring is otherwise deemed dependable.

The evaluation of monitoring programs in the QAT provides a detailed accounting of the contribution of statistical and operational factors affecting the variability and bias of an estimation process. This accounting constitutes a key tool with which to determine the factors that contribute most to variability and bias, and that are most amenable to change in a cost-effective manner. Results from the QAT can therefore guide the selection of options from the non-exhaustive choices and examples that follow.

6.1.1 Improving coverage

Coverage here refers to the relationship between the target population and the sampling frame, the list of units available for sampling of the target population. Undercoverage occurs when the sampling frame excludes sampling units that are otherwise members of the target population, while overcoverage occurs when the frame includes units from another population. Both can contribute to bias.

Undercoverage, which is likely to be the most frequent of the two in fishery monitoring, can occur for example, if:

- a survey is based on an incomplete list of fishers;
- sampling units are excluded from monitoring due to remoteness or an incapacity to accommodate the monitoring (e.g. at-sea observers with respect to vessel size or safety standards); or
- a fishing activity (the sampling unit) is not recognized as being part of the frame, e.g. trips for which there is no pre-departure hail-out causing them to be excluded from the trip selection process for at-sea observer deployment.

Bias occurs when excluded sampling units have characteristics (e.g. catch rates) that are different from the population. The implementation of procedures that reduce the likelihood of poor coverage will also reduce the potential for bias. These include:

- administrative procedures, such as mandatory licensing for recreational fisheries, that ensure that sampling frames are accurate;
- operational procedures that provide suitable alternative monitoring for units that are otherwise excluded (e.g. motion sensing cameras for monitoring recreational fishing activity in remote locations);
- the application of standards to minimize the number of units that cannot accommodate monitoring (e.g. vessel safety standards with regards to at-sea observer safety);
- technology to ensure that the sampling frame is correctly specified (e.g. mandatory hails) and to identify sampling units that were missed for follow-up actions and for deterrence (e.g. vessel monitoring systems); and
- increased deterrence and enforcement activities to minimize or eliminate cases of unreported or hidden fishing activity.

6.1.2 Increasing sample sizes

Increasing sample sizes is effective in reducing statistical variability.

Sampling rate (% coverage) close to 100% or equal to 100% (i.e. censuses) may also attenuate some of the variability cause by operational characteristics. The influence of unplanned sample selection issues (e.g. clustering, unplanned stratification) will attenuate, as will errors from imputation. However, certain other types of errors, which occur also in censuses (e.g. missing values, measurement and data handling errors), will be unaffected.

Increasing the sample size generally does not reduce bias, other than certain statistical biases. For example, biases in data reported by resource users are present in both sample surveys and censuses (e.g. underreporting bycatch of a species in mandatory logbooks). While biases associated with observer effects or the unrepresentative sampling of units for observation (e.g. vessel selection effects for at-sea observer programs) should generally attenuate monotonically as coverage increases to 100%, the form of the decrease is far from certain (Rago et. al. 2005).

6.1.3 Improving sampling design and sample selection

Using sampling designs that recognize that variability can be generated at different scales can improve the ratio of precision to cost; for fixed sampling costs, the precision of an estimate can be improved or, inversely, for a fixed precision of an estimate, the sampling costs can be reduced (Cochrane 1977; Cotter et al. 2002; Volstad and Fogarty 2006; Cotter and Pilling 2007).

Cluster sampling can improve the precision/cost ratio especially when clusters are similar to the whole population (e.g. same variability) and clustering reduces effort (e.g. for monitors travelling from dock to dock). Stratified sampling can improve the precision/cost ratio when within-stratum variability is smaller than among stratum variability. Stratification can be established with respect to a single objective, or optimized if there are multiple objectives for a monitoring program (e.g. Miller et al. 2007). In monitoring programs aimed at estimating rare catches that are highly clustered in space and time, the optimal solution may be an adaptive sampling scheme by which once an event is detected, additional samples are rapidly allocated in the region surrounding the event.

In all instances, it is critical that the design, implementation, and analysis methods all correspond, failing which errors that are difficult to estimate may be generated.

Irregular sampling unit selection probabilities can generate bias and may cause variability to be incorrectly estimated (Allard and Benoît 2019). Procedures and tools that identify units for sampling with sufficient lead time to deploy a monitor will allow for a closer adherence to the sampling plan and therefore a lesser risk of generating bias and variability. Examples of such procedures and tools are pre-departure and pre-arrival hails, automated sample selection programs and fishing effort monitoring (Benoît and Allard 2009; Palmer et al. 2016; Beauchamp et al. 2019).

The targeted deployment of a monitor to a sampling unit for enforcement and deterrence purposes results in a forced inclusion of that unit with a probability of one. The sample is not representative of other samples taken following the monitoring scheme and should not be treated equivalently. However, data from at-sea observer surveys presently do not distinguish targeted and non-targeted samples for legitimate privacy and enforcement integrity reasons. The analysist is therefore unable to treat the data accordingly. Measures to correct the situation, such as having distinct enforcement and standard monitoring databases would improve the quality of estimation process that include monitoring tools that serve to estimate one or more parameters as well as for enforcement.

6.1.4 Addressing other key sources of bias

Monitoring tools that depend on data supplied by resource users are susceptible to biases resulting from biased reporting and missing values due to intentional factors (Allard and Benoît 2019). Similarly, certain independent monitoring tools, most notably at-sea observer surveys, can be associated with biases resulting from observer effects (Benoît and Allard 2009; Faunce and Barbeaux 2011). In both cases, the direction and magnitude of biases are likely to be a function of incentives for misreporting or altering behaviour, such as target catch and bycatch limits, stigma associated with the bycatch of certain species and a desire to hide illegal activities (Beauchamp et al. 2019).

Bias in monitoring tools that rely on data supplied by resource users may be reduced or eliminated by switching to accredited independent monitoring. However, if these tools are implemented as sample surveys and the proper controls are not in place, biases can remain if there are observer effects. For both resource-user dependent and independent monitoring tools, biases are likely to be most effectively addressed by addressing incentives for compliant behavior, possibly including routine auditing. For example, video monitoring has been very effectively used to create an invigilation effect and a means for routine auditing (compliance monitoring). As a result, catch reporting in west coast groundfish fisheries results in very precise and unbiased estimates derived from harvester logbooks (Stanley et al. 2011). Compliance monitoring (e.g. via overflights, vessel monitoring systems, random boardings by officers), combined with strong disincentive for non-compliance (e.g. stiff fines), can help to reduce bias.

The structure of monitoring programs themselves can generate strong incentives for non-compliance (misreporting or observer effects). At-sea observer programs are probably the most notable example. In Canada, these programs have three competing and arguably largely incompatible goals: monitoring for estimation purposes (often mainly for bycatch), monitoring for compliance purposes (bycatch limits) and enforcement/deterrence. Monitoring for compliance, particularly if it means closing a fishery once the limit is reached, creates a powerful incentive to alter behavior to minimize bycatch that would otherwise be caught when an observer is absent. The same is true given enforcement/deterrence objectives. As long as it remains profitable for a fisher to alter behavior when an observer is present, biases will remain. It is for this reason that an optimal observer coverage level below 100% cannot be specified as it relates to bias (Rago et al. 2005).

6.1.5 Decrease controllable sources of error

There are several operational characteristics identified by Allard and Benoît (2019) that relate to unplanned errors that contribute to the statistical variability of estimates or constitute an additional source of variability. Reducing these errors will decrease variability and improve quality, though perhaps not sufficiently to render a monitoring program dependable. Table 4 summarizes possible solutions for improving quality for some of the operational characteristics.

6.1.6 The case of multiple fisheries

When the RST establishes risk and quality requirements at a population level and several fisheries impact this population, multiple options to improve dependability and/or cost-effectiveness will be available.

In such situations, a careful analysis of the contribution of each fishery is recommended. For example, it may be difficult to improve the contribution of a fishery observed by a census, even if it takes a large part of catch but it may be cost-effective to reduce the coverage to sample survey. On the other hand, high quality monitoring (and potentially costly) programs may be required for each of several small fisheries jointly catching a large part of the population.

Table 4. Possible actions to improve quality for some operational characteristics associated with unplanned errors.

Operational characteristic	Examples of possible actions to improve quality
Errors reported by independent observers	Improve catch sub-sampling methods or increase sub-sampling rates; provide means to improve the precision of catch estimates, for example by reducing the reliance on visual estimation or providing stricter guidelines for visual estimation.
Equipment error	Mandate the use of more reliable equipment or implement a protocol and schedule for equipment standardization.
Data handling errors	Remove intermediate data handling steps, for example using electronic log-books or other electronic data capture methods. Increase quality assurance and control using double entry of data from paper forms.
Adjustment errors	Routinely revisit the empirical relationships used to make adjustments; minimize the number of adjustments required by standardizing requirements for the landed form of catches.
Imputation errors	Minimize the need for imputations by adjusting sampling stratification schemes; undertake research to identify covariates and models that can improve imputation.

6.2 MODIFYING FISHERY MANAGEMENT RULES

Decreasing conservation risk to lower quality requirements is the most likely option when resource users are unwilling to undertake monitoring program changes that would lead to sufficient improvements in quality. This will most often result from a lack of desire to increase the costs or the intrusiveness of fishery monitoring.

6.2.1 Reducing conservation risk

Conservation risk can be reduced by accepting removal limits that are lower than would otherwise be proposed based on stock status and the Precautionary Approach for the stock, or in the case of bycatch, by reducing catches by increasing gear selectivity, by avoiding locations and times where bycatch is most probable, or decreasing fishing effort. Conservation risk can also be reduced in some circumstance by improving the science underlying the stock assessment to improve the accuracy (decrease bias and uncertainty) of reference points, introducing reference points for fisheries where there are presently none, improving the estimation of stock abundance and status, and improving the estimation of risk of different management options. However, while these changes are an integral part of the fishery assessment management system, such changes to the scientific process are outside the scope of DFO's Fishery Monitoring Policy. Following the establishment of proposed measures to decrease conservation risk, both the conservation risk and estimation quality should be re-assessed using the RST and QAT respectively (Fig. 2) to confirm dependability.

6.2.2 Operational limits: an approach to controlling quality in a compliance application

In situations were the dependability of a parameter estimation process is deemed insufficient in a compliance application, the following approach can be used to correct the situation.

Following the usual statistical notation, let $0 < \alpha < 1$ be a pre-selected value of the probability of an incorrect conclusion detrimental to the conservation goal (the significance level in the statistical vocabulary) and let $z_\alpha = \varphi^{-1}(\alpha)$ (examples: $z_{0.50} = 0$ and $z_{0.05} = -1.64$).

Considering the statistical test presented in the previous section, the rejection limit required to obtain a probability α of an incorrect conclusion detrimental to the conservation goal is calculated as:

$$\hat{L}_{oper} = \hat{L} - b_{\hat{L}} + b_{\hat{\theta}} + z_{\alpha}\sqrt{\sigma_{\hat{L}}{}^2 + \sigma_{\hat{\theta}}{}^2}$$

where $\hat{L}_{oper}$, which we call an operational limit, is the limit required to bound the probability of an incorrect conclusion detrimental to the conservation goal to α. If $\hat{L}_{oper}$ is the limit used for tactical decision making, we obtain the following probability of avoiding a decision detrimental to conservation:

- The predetermined significance level of the test, α, is the probability of concluding, incorrectly, that the limit was not reached when it was reached exactly. If we set $b_{\hat{L}} = 0$, $b_{\hat{\theta}} = 0$ and $\alpha = 0.50$ (i.e. $z_{\alpha} = 0$), the conclusion is based on the simple comparison between $\hat{\theta}$ and $\hat{L}$, the usual approach to monitoring for compliance.
- Choosing a small value for α (e.g. $\alpha = 0.05$ and $z_{\alpha} = -1.64$), the operational limit $\hat{L}_{oper}$ corresponds to a precautionary or risk-averse approach and the probability of reaching an incorrect conclusion detrimental to the conservation goal is $1 - \alpha$ (e.g. $1 - \alpha = 0.95$).

The last point illustrates the difference between the quality of the parameter estimation process and the dependability of the decision process. Improving quality of the parameter estimation process (by modifying the monitoring program) can only impact $b_{\hat{\theta}}$ and $\sigma_{\hat{\theta}}$. Even with a perfect parameter estimation process (i.e. $b_{\hat{\theta}} = 0$ and $\sigma_{\hat{\theta}} = 0$), uncertainty will remain from the estimation of the true limit, L. In fact, this uncertainty may often be underestimated, as it likely does not account for all uncertainties involved in the stock assessment. The parameter α can be seen as a dependability tuning parameter; it allows the computation of a limit $\hat{L}_{oper}$ under which the probability of avoiding a decision detrimental to conservation is $1 - \alpha$.

Figure 2 illustrates this concept. The traditional decision rule is to conclude that the limit has been breached when the observed value of the parameter is greater than the declared limit or, equivalently, when the difference (observed parameter value – declared limit) is greater than zero. Each panel in Figure 2 illustrates the difference (true parameter value – correct limit) for different bias scenarios. The operational limits on the difference are shown for values of $1 - \alpha$ = 0.50, 0.75, 0.90 and 0.95.

- In panel A of Figure 2, the parameter and the limit are estimated without bias but with a small level of uncertainty (SE = 5%); the operational limit must be lowered slightly to account for the risk due to this uncertainty.
- In panel B of Figure 2, the parameter estimate SE is 25%; the operational limits must be lowered much more to account for the greater uncertainty.
- In panel C of Figure 2, the parameter estimate is negatively biased (i.e. the parameter is likely to be underestimated); the operational limits are shifted left.
- In panel D of Figure 2, the estimate of the limit is positively biased slightly (the declared limit is likely to be set somewhat too high) and its SE is 25%; the operational limits are shifted and lowered to account for the uncertainty in the declared limit.
- In panel E of Figure 2, bias and uncertainty are present in both the declared limit and the parameter estimate; the operational limits must be further shifted and lowered.

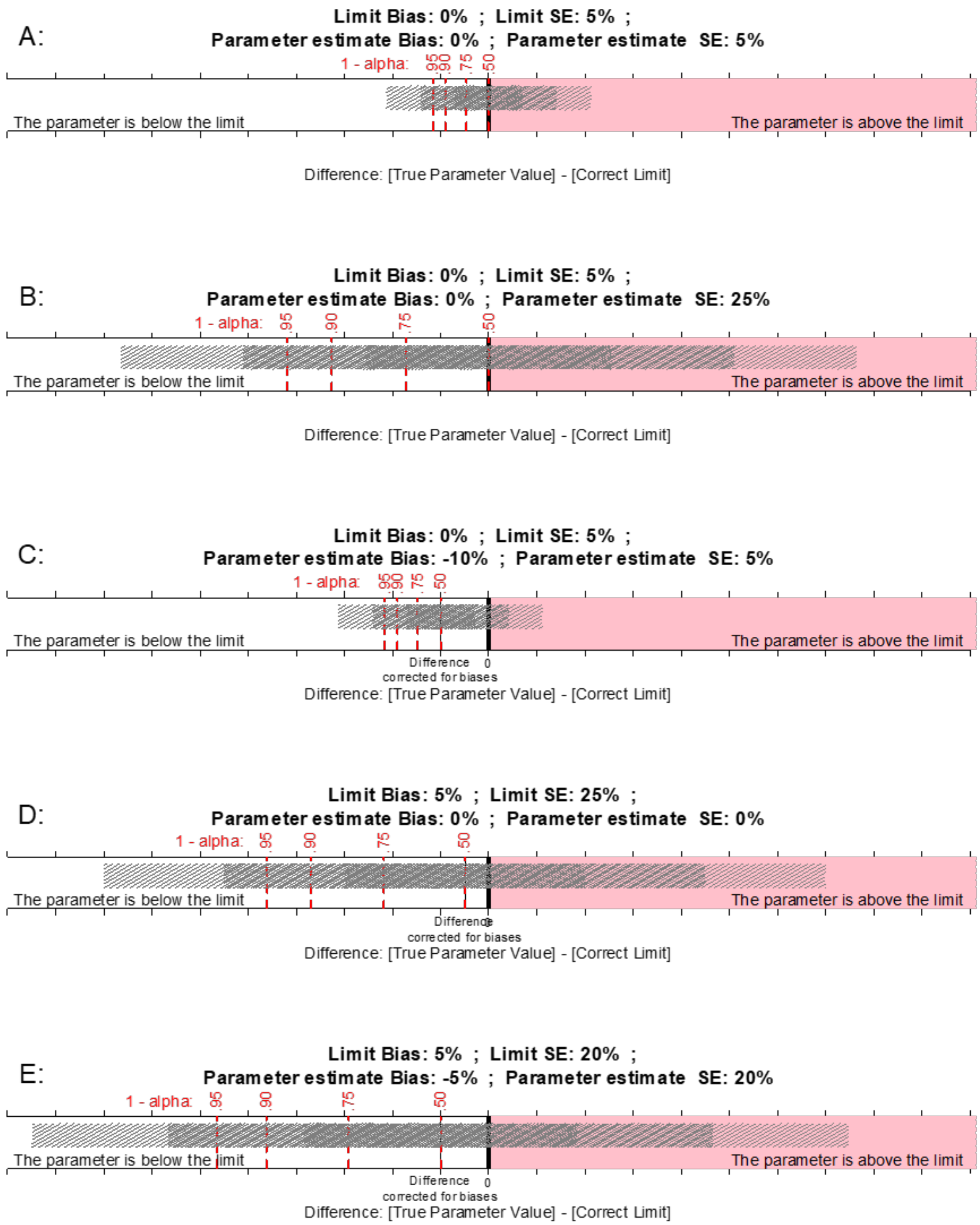

Figure 2. Illustration of the concept of an operational limit. The traditional decision rule is to conclude that the limit has been breached when the observed value of the parameter is greater than the declared limit or, equivalently, when the difference (observed parameter value – declared limit) is greater than zero. Each panel illustrates the difference (true parameter value – correct limit) with the decision threshold 0 indicated by the thick black line. The operational limits on the difference are shown by a red dashed line for $1 - \alpha$ *= 0.50, 0.75, 0.90 and 0.95. The grey band illustrates the uncertainty in the observed difference.*

6.3 OPTIONS FOR NEW MONITORING PROGRAMS

The implementation of the fishery monitoring policy is likely to lead to a requirement in some fisheries to establish monitoring programs where none existed previously. While there are tools to gauge conservation risk in these and other cases (section 2), there will likely be little or no direct information to estimate quality and therefore dependability. There are three complementary approaches that can be used to design and assess a new monitoring program with respect to quality and dependability. Considerations related to complexity in the fisheries, such as the remoteness of certain ports and the capacity of vessels to accommodate monitoring, and other key considerations for the development of monitoring programs (see Zollett et al. 2011) are also very important but beyond the scope of this document.

First, the assessment of conservation risk will determine the broad requirements for bias and variability, which in turn help to identify the monitoring tools that might be appropriate. For example, at high risk, monitoring programs will have to support estimates that are unbiased and of high precision (Appendix I). The intolerance to bias will likely eliminate many or all monitoring tools that involve reporting by resource users, as well as third party monitoring where there is a potential for bias such as at-sea observer programs with partial coverage (e.g. Benoît and Allard 2009; Faunce and Barbeaux 2011). The requirement on precision will motivate a monitoring program with high sampling rates, potentially a census. Beauchamp et al. (2019) and Mangi et al. (2015) discuss elements that can inform on the potential bias and variability associated with different monitoring tools.

Second, it may be possible to borrow information from monitoring programs on similar fisheries and to assume, as an initial step, that the profile for bias and variability across statistical and operational factors would be the same should an identical program be implemented. In addition it may be possible to use spatial and temporal variability in catches in scientific surveys in the area to inform the statistical variability assumed in the initial evaluation of potential quality and to help plan sampling stratification for catch monitoring (Figus and Criddle 2019).

The third option is to conduct a pilot monitoring program (NMFS 2004). This would allow for an estimation of the statistical variability in the estimated parameter and may provide information that can be used to optimize sampling, for example as relates to stratification or procedures to select samples. For pilot at-sea observer programs in the US, NMFS (2004) recommend pilot programs involving 0.5-2% coverage or 100 trips per stratum, whichever is smallest.

Regardless of the approach that is adopted, it is clear that new monitoring programs should be reassessed within a few years of implementation to ensure that they meet quality and dependability objectives.

7. DISCUSSION

The assessment of the adequacy of fishery monitoring clearly needs to account for many aspects of the fishery assessment and management systems, and in itself constitutes an integral part of that system. In very simplified terms:

- The scientific assessment of resource or population status and definition of sustainable removals is dependent on the quality (bias and variability) of estimates derived from fishery monitoring. Failure to recognize biases in particular (e.g. catch under-reporting) can undermine the dependability of the assessment with respect to establishing sustainable harvest rates.
- The assessments provide estimation of risks posed to populations and species by fishing, including risks of specific management actions (e.g. total allowable catch).

- The effectiveness of fishery management depends on the quality of the scientific advice and the ability to correctly implement controls in the fisheries that affect the resource.
- Correctly implementing controls depends in part on having a dependable fishery monitoring and catch estimation process and accounting for errors in the estimation process when establishing limits.
- Ensuring the monitoring is fit for purpose and that operational limits are established correctly requires the best possible evaluation of the statistical and operational factors that affect quality of estimates and compliance-to-limit decision based on the monitoring program, the assessment of risk to conservation, and the definition of risk tolerance (the likelihood of non-sustainable catch).

It is clear from the preceding that it is not possible to separate the scientific assessment process, from that used to set, review, and implement fishery monitoring objectives and plans. It therefore makes sense that the assessment of catch monitoring be harmonized with population assessments (stock assessments, recovery potential assessments, allowable harm and potential biological removal assessments). Regional, zonal and transboundary population assessment processes provide a peer review venue during which catch monitoring can be reviewed. These processes are attended by DFO Science, DFO Fisheries Management and certain stakeholders that can contribute information to the process. Furthermore, most assessment processes are followed by (management) advisory committee meetings that also involve DFO Fisheries Management, DFO Science and a broader group of stakeholders. Together, the science and management advisory meetings provide a venue for assessing the quality of monitoring programs, ensuring that the population assessments best account for the quality, assessing conservation risk, and establishing dependability with respect to risk. The management advisory meetings are a venue for establishing fishery monitoring plans that match quality and risk, with respect to monitoring costs. For commercial fisheries it therefore makes sense to phase in the review of monitoring programs under the new policy as part of the existing multi-year stock assessment cycle. Periodic review of monitoring programs could also follow this cycle but would likely not be required at every assessment to avoid an undue burden. Substantial actual, anticipated or contemplated changes in the fishery, the management scheme, the monitoring, or the stock status that could impact the dependability of monitoring would constitute grounds for triggering a re-assessment.

Reviewing dependability for monitoring programs according to assessment cycles, including recovery potential assessments for species of conservation concern, will address monitoring for fisheries affecting many targeted species and some bycatch species. However, the process is likely to be incomplete for many species that are incidentally captured in numerous fisheries and for which fishing could be an important source of harm. Ensuring sustainability may therefore require some targeted catch monitoring program reviews for certain important or vulnerable bycatch species.

In this document and in Allard and Benoît (2019), the evaluation of dependability is undertaken as a distinct process that requires input from population assessments and information from existing monitoring programs and related studies, and for which the outputs affect monitoring and perhaps fishery management plans. An alternative to this segmented approach is to evaluate dependability in the context of the population dynamics, and the assessment and fishery management systems, using management strategy evaluation (MSE). MSE comprises the simulation of stock dynamics under simulated observation, scientific assessment, and fishery management systems (Smith et al. 1999; Rademeyer et al. 2007; Punt et al. 2014). Typically MSE involves the evaluation of different management procedures that can meet a prioritized set of fishery management objectives, while accounting for imperfections in the data used to assess and manage the population. The data include simulated survey and fishery monitoring observations. Punt (1999) used MSE to evaluate the consequence to management and stock viability of various levels of observer coverage in the Australian blue grenadier fishery. MSE could be used more broadly to inform the design of fishery monitoring programs and to assess their dependability. The consequences of statistical and operational fishery monitoring program characteristics that affect estimation process quality can readily be simulated using the inputs to the QAT. Dependability is not established using a statistical assessment as in the QAT, but rather based on the consequences it has on the simulated medium and long-term sustainability of the population. However, undertaking a MSE can be a long and complex process and will not be feasible for a large number of fisheries or populations under DFO's responsibility, at least not in the foreseeable future. However, for some priority cases, MSE may be a useful and viable means for ensuring dependability of fishery monitoring. This is particularly true for fisheries or populations for which an MSE is being undertaken to address other objectives and where there may be little incremental cost of adding an evaluation of dependability.

Completing an evaluation of quality using the QAT requires specifying anticipated values or ranges of values for the variability and bias elicited by 15 operational characteristics. The process involves some subjectivity and can be somewhat time consuming. There is a clear need for additional methods and approaches to streamline the process and to ensure national consistency in application, given the large number of Canadian fisheries and populations of interest, for which an assessment of dependability will be required under the fishery monitoring policy. It should be possible to identify a value or range of values for the variability and bias contributed by certain operational characteristics that are common across fisheries. These include adjustment errors, data handling errors, and measurement errors for different classes of measurement tools. Similarly, for some monitoring tools deployed in standard or similar ways, the variability and bias contributed by operational characteristics should be similar or the same. This includes mandatory dockside monitoring and logbooks. For these classes of tools, it may be possible to determine dependability overall when there is no statistical variability or bias, or at least the contribution of most or all operational characteristics to quality.

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10. APPENDICES

10.1 APPENDIX I: SUMMARY OF THE DRAFT RISK SCREENING TOOL

10.1.1 Background and definitions

The existing draft Risk Screening Tool (RST) is a qualitative method to screen a fishery against a set of generic conservation and compliance related fishery risk factors. It is integral to the review of fishery monitoring programs as part of the application of the Fishery Monitoring Policy. The two other inputs to the review process are the assessment of data quality, and consideration of the complexity of the management regime for the fishery. These three parts of the monitoring program review provide a gap analysis which, in turn, informs the setting of monitoring objectives and a monitoring plan.

The review of monitoring programs considers risks to resource conservation, and risks to compliance with laws and regulations. Only the considerations related to risks to conservation in the RST are presented in this research document.

The policy requires a consistent set of biological components to be considered for all fishery units, providing a structured approach to evaluate monitoring programs. The three components are target species, bycatch species, and community and habitat.

Target species are those that a fisher is licensed or otherwise allowed to direct for, i.e. the target species of the fishery. In a multispecies fishery, this includes any species that the fisher is licensed or allowed to direct for on a given fishing trip regardless of whether the fisher does so or not. Risk factors examined for target species are stock status and the impact of the fishery on incidental discards.

Bycatch species are any retained species or specimens that the fisher was not licensed, or otherwise allowed, to direct for but is required or permitted to retain, as well as all non-retained catch, including catch released from gear and entanglements, whether alive, injured or dead, regardless if they are target species or non-target species. Species assessed as at risk under the *Species at Risk Act* (SARA) that are captured through fishing activities would generally fall into the bycatch category. Risk factors to be examined are risks from the fishery to retained bycatch species and non-retained bycatch species.

The ‘community and habitat’ component is meant to capture changes (direct or indirect) to the ecosystem including to other key species and habitat impacts. This includes sensitive areas such as sensitive benthic habitat as defined by the *Policy on* M*anaging the Impacts of Fishing on Sensitive Benthic Areas* (DFO 2009a) and especially Significant Benthic Areas as described in the *Ecological Risk Assessment Framework* (ERAF) *for Coldwater Corals and Sponge Dominated Communities* (DFO 2019a), as well as other Vulnerable Marine Ecosystems as defined by the Food and Agricultural Organization of the United Nations (FAO) and the Northwest Atlantic Fisheries Organization (NAFO). Risk factors examined for community and habitat are the risk of the fishery to other key species, direct impacts on habitat structure or composition, and indirect impacts on habitat structure or composition.

The broad conservation objectives for each biological component are:

- Target Species: Ensure that the fishery is managed in a manner that supports the sustainable harvesting of aquatic species and minimizes the risk of fisheries causing serious or irreversible harm to target species (as defined by the spirit and intent of *A Fishery Decision*-M*aking Framework Incorporating the Precautionary Approach* (DFO 2009b) and the *Policy on* M*anaging Bycatch* (DFO 2019b)).

- Bycatch Species: Ensure that the fishery is managed in a manner that supports the sustainable harvesting of aquatic species and that minimizes the risk of fisheries causing serious or irreversible harm to bycatch species (as defined by the *Policy on* M*anaging Bycatch* (DFO 2019b)). For those bycatch species listed under SARA, an additional objective is to prevent the species from being extirpated or becoming extinct, to provide for the recovery of wildlife species that are extirpated, endangered or threatened as a result of human activity, to manage species of special concern to prevent them from becoming endangered or threatened (as defined by the *Species at Risk Act*), and to respect legal prohibitions against harming listed species or their habitat except where authorized under incidental harm permits.
- Community and Habitat: Mitigate the impacts of the fishery on sensitive areas or avoid impacts of fishing that are likely to cause serious or irreversible harm to sensitive marine habitat, community and species (as defined by the *Policy on* M*anaging Impacts of Fishing on Sensitive Benthic Areas* (DFO 2009a)).

10.1.2 Risk screening procedure

Generic procedure

Risk is a function of both consequence and likelihood and is evaluated in terms of immediate and short-term impacts (2 to 3 year horizon). In the RST, consequence refers to the degree of impact on a risk factor as a result of fishing activity, and is based on an ordinal four-category classification (Appendix Table 1.1).

Appendix Table 1.1 Ordinal categories for the consequence, defined as the degree of impact on the risk factor, resulting from a fishing activity.

Ordinal	Consequence	Notes and Examples
1	These actions in the fishery have a negligible impact on the risk factors of focus.	These actions have negligible interactions with this risk factor in the fishery. As a result, this occurrence poses minimal concern in the fishery.
2	These actions in the fishery have a minor impact on the risk factors of focus.	These actions have minor interactions with this risk factor in the fishery. As a result, this occurrence poses some concern in the fishery.
3	These actions in the fishery have a moderate impact on the risk factors of focus.	These actions have moderate interactions with this risk factor in the fishery. As a result, this occurrence poses notable concern in the fishery.
4	These actions in the fishery have a significant impact on the risk factors of focus.	These actions have significant interactions with this risk factor in the fishery. As a result, this occurrence poses a significant concern in the fishery.

In the RST, likelihood is the probability of each consequence occurring, also based on an ordinal four-category classification (Appendix Table 1.2).

When assigning a likelihood score to each consequence category, an assessor should ensure that the underlying or implied probabilities sum to 100% across the consequence categories.

Appendix Table 1.2 Ordinal categories for the likelihood of each consequence occurring.

Ordinal	Likelihood	Descriptor
1	Rarely	The consequence has never been heard of in these circumstances, but it is not impossible within the time frame. Probability less than 5%.
2	Unlikely	The consequence is not expected to occur in the timeframe but it has been known to occur elsewhere under special circumstances. Probability of 5% to less than 20%.
3	Possible	Evidence to suggest this consequence level is possible and may occur in some circumstances within the timeframe. Probability of 20% to less than 50%.
4	Likely	A particular consequence level is expected to occur in the timeframe. Probability of 50% to 100%.

A risk score for each conservation risk factor is calculated by selecting the largest value from among those produced by product of each consequence level score and its associated likelihood score. This risk score is then attributed to a risk level according to the following scheme:

- risk level = Low for risk scores of 1, 2, 3, or 4;
- risk level = Moderate for risk scores of 6, 8, or 9;
- risk level = High for risk scores of 12 or 16.

Risk scores for each risk factor are not combined to get an overall risk score for the fishery. By determining separate risk scores for each conservation factor, fishery monitoring measures can be designed specifically to address the higher risk issues.

10.1.3 Consequence descriptors for conservation factors related to catch

The draft RST includes tables that provide descriptors associated with each consequence level for factors related to catch, community and habitat, and compliance. Given the context of the present report, we present below the descriptors for conservation factors related to catch. One of the objectives of this report is to ensure the completeness and validity of these descriptors.

Appendix Table 1.3. Impact on the stock status of targeted species.

Nominal	Consequence	Notes and Examples
1	Fishery causes negligible impacts to population size, recruitment, range, dynamics or disruptions to behaviour (including trophic relationships) of target species.	The management framework (e.g. decision rules) is such that, where precautionary approach reference points are available, the target fishing mortality (F_{target}) is well within the fishing mortality reference point (F_{lim}) and the fishing mortality rate (F) is well above the Limit Reference Point (LRP) of the stock. The stock status is healthy. In the absence of a fishing mortality reference point there is a reasonable expectation that removals will be small with respect to the size of the stock and its productivity. Evidence in support includes: (1) small catches relative to fishable biomass estimates from surveys; (2) the fishery occurring in a small and marginal portion of the stock distribution; and/or (3) an increasing trend in post-recruitment abundance.
2	Fishery causes minor impacts to population size, recruitment, range, dynamics (including trophic relationships) or disruptions to behaviour of target species. Species capacity to recover from a depleted state is negligibly impacted.	The management framework (e.g. decision rules) is such that, where precautionary approach reference points are available, the target fishing mortality (F_{target}) is well below either the fishing mortality reference point (F_{lim}), when the stock is in the Healthy Zone, or the fraction of F_{lim} believed to be sustainable, when the stock is in the cautious zone. Furthermore, there is a low risk that F_{target} may be exceeded by a relatively large amount due to quota overruns, unreported fishing or illegal fishing. In the absence of a fishing mortality reference point there is a reasonable expectation that removals will be small with respect to the size of the stock and its productivity. Evidence in support includes: (1) small catches relative to fishable biomass estimates from surveys; (2) the fishery occurring in a small and marginal portion of the stock distribution; and/or (3) an increasing trend in post-recruitment abundance. For species assessed as “at risk” by COSEWIC recovery potential assessment of species at risk indicates fishing pressure as an unlikely obstruction to recovery.
3	Fishery causes moderate impacts to population size, recruitment, range, disruptions to behaviour or dynamics such that stock level depletions/extinctions and/or range contractions of target species are not anticipated. A species capacity to increase from a depleted state may be adversely impacted, particularly if monitoring is inadequate.	The management framework is such that, where precautionary approach reference points or suitable proxies are available, the target fishing mortality (F_{target}) or removals are near or below the removals reference point when the stock is in the Healthy Zone or the fraction of the removal reference believed to be sustainable when the stock is in the cautious zone. Furthermore, there is a moderate risk that F_{target} or removals may be exceeded by a relatively large amount due to quota overruns, unreported fishing or illegal fishing. In the absence of a fishing mortality or removal reference point there is a reasonable expectation that removals will be moderate with respect to the size of the standing stock and its productivity. Evidence in support includes: (1) catches relative to fishable biomass estimates from surveys that suggest a fishing mortality (F) that is close to or below 0.4 (elasmobranchs) or 0.8 (teleosts) times the species natural mortality (M); (2) the fishery occurring not just in areas representing a marginal portion of the stock distribution; and/or (3) a stable trend in post-recruitment abundance. For species assessed as “at risk” by COSEWIC, the recovery potential assessment of species at risk indicates fishing pressure as a possible obstruction to recovery.

Nominal	Consequence	Notes and Examples
4	Fishery causes significant impacts to population size, recruitment, range, disruptions to behaviour and/or dynamics such that stock level depletions/extinctions and/or severe range contractions of target species occur. A species capacity to increase from a depleted state is adversely impacted.	The management framework is such that, where precautionary approach reference points or suitable proxies are available, there is a high likelihood that the realized fishing mortality could exceed levels deemed sustainable in some years, by design and/or due to a moderate to high risk that F_{target} or removals may be exceeded by a relatively large amount due to quota overruns, unreported fishing or illegal fishing. In the absence of a fishing mortality or removal reference point there is a reasonable expectation that removals can be large with respect to the size of the stock and its productivity. Evidence in support includes: (1) catches relative to fishable biomass estimates from surveys that suggest a fishing mortality (F) that is close to or above 0.4 (elasmobranchs) or 0.8 (teleosts) times the species natural mortality (M); (2) the fishery occurring throughout the stock area or concentrated in areas of stock concentration; and/or (3) a decreasing trend in post-recruitment abundance. For species assessed as “at risk” by COSEWIC, recovery potential assessment of species at risk indicates fishing pressure as a likely contributor to continued population decline.

Appendix Table 1.4. Impact on the stock status of discards of the target species.

Nominal	Consequence	Notes and Examples
1	Fishery causes negligible impacts on released/discarded incidental catch of target species because no target species are discarded.	The management framework (e.g. decision rules) is such that, where precautionary approach reference points are available, the target fishing mortality (F_{target}) is well within the fishing mortality reference point (F_{lim}) and the fishing mortality rate (F) is well above the Limit Reference Point (LRP) of the stock. he stock status is healthy.
2	Fishery causes minor impacts on released/discarded incidental catch of target species with low probability of death to released/discarded incidental catch of target species.	The management framework (e.g. decision rules) is such that, where precautionary approach reference points are available, the target fishing mortality (F_{target}) is well below either the fishing mortality reference point (F_{lim}), when the stock is in the Healthy Zone, or the fraction of F_{lim} believed to be sustainable, when the stock is in the cautious zone. Furthermore, there is a low risk that F_{target} may be exceeded by a relatively large amount due to quota overruns, unreported fishing or illegal fishing. In the absence of a fishing mortality reference point there is a reasonable expectation that removals will be small with respect to the size of the stock and its productivity. Evidence in support includes: (1) small catches relative to fishable biomass estimates from surveys; (2) the fishery occurring in a small and marginal portion of the stock distribution; and/or (3) an increasing trend in post-recruitment abundance. In the case of bycatch species (including species at risk), additional evidence is an inferred M of the bycatch species which is higher than the M of the target species, unless the bycatch species is suspected of having higher catchability to the gear. Species (including species at risk) that are the target of one or more other fisheries, removals are small relative to removals in the targeted fisheries and are therefore very unlikely to in and of themselves produce an unsustainable fishing mortality. For species assessed as “at risk” by COSEWIC, recovery potential assessment of species at risk indicates fishing pressure as an unlikely obstruction to recovery.

Nominal	Consequence	Notes and Examples
3	Fishery causes moderate impacts on released/discarded incidental catch of target species with moderate probability of death to released/discarded incidental catch of target species.	The management framework is such that, where precautionary approach reference points are available, the target fishing mortality (F_{target}) is near or below the fishing mortality reference point (F_{lim}) when the stock is in the Healthy Zone or the fraction of F_{lim} believed to be sustainable when the stock is in the cautious zone. Furthermore, there is a moderate risk that F_{target} may be exceeded by a relatively large amount due to quota overruns, unreported fishing or illegal fishing. In the absence of a fishing mortality reference point there is a reasonable expectation that removals will be moderate with respect to the size of the standing stock and its productivity. Evidence in support includes: (1) catches relative to fishable biomass estimates from surveys that suggest an F that is close to or below 0.4 (elasmobranchs) or 0.8 (teleosts) times the species natural mortality (M); (2) the fishery occurring not just in areas representing a marginal portion of the stock distribution; and/or (3) a stable trend in post-recruitment abundance. In the case of bycatch species (including species at risk), additional evidence is an inferred natural mortality of the bycatch species that is similar to the M of the target, unless the bycatch species is suspected of having higher catchability to the gear. Species (including species at risk) that are the target of one or more other fisheries, removals are moderate relative to removals in the targeted fisheries and may, in and of themselves, produce an unsustainable fishing mortality on occasion. For species assessed as “at risk” by COSEWIC, the recovery potential assessment of species at risk indicates fishing pressure as a possible obstruction to recovery.
4	Fishery causes significant impacts on released/discarded incidental catch of target species with high probability of death to released/discarded incidental catch of target species.	The management framework is such that there is a high likelihood that the realized fishing mortality could exceed levels deemed sustainable in some years, by design and/or due to a moderate to high risk that F_{target} may be exceeded by a relatively large amount due to quota overruns, unreported fishing or illegal fishing. In the absence of a fishing mortality reference point there is a reasonable expectation that removals can be large with respect to the size of the stock and its productivity. Evidence in support includes: (1) catches relative to fishable biomass estimates from surveys that suggest an F that is close to or above 0.4 (elasmobranchs) or 0.8 (teleosts) times the species natural mortality (M); (2) the fishery occurring throughout the stock area or concentrated in areas of stock concentration; and/or (3) a decreasing trend in post-recruitment abundance. In the case of bycatch species (including species at risk), additional evidence is an inferred natural mortality of the bycatch species that is lower than the natural mortality of the target. Species (including species at risk) that are the target of one or more other fisheries, the fishery will be considered high consequence if (1) the target fishery is considered high consequence and the bycatch levels are more than just negligible, or (2) the target fishery is considered moderate consequence but the bycatch levels may, in and of themselves, produce an unsustainable fishing mortality. For species assessed as “at risk” by COSEWIC, recovery potential assessment of species at risk indicates fishing pressure as a likely contributor to continued population decline.

Appendix Table 1.5. Impact on the stock status of retained or discarded bycatch.

Nominal	Consequence	Notes and Examples
1	Fishery causes negligible impacts to population size, recruitment range or dynamics (including trophic relationships) of retained bycatch species beyond natural variability.	The management framework (e.g. decision rules) is such that, where precautionary approach reference points are available, the target fishing mortality (F_{target}) is well within the fishing mortality reference point (F_{lim}) and the fishing mortality rate (F) is well above the Limit Reference Point (LRP) of the stock. The stock status is healthy.
2	Fishery causes minor impacts to population size, recruitment, range or dynamics (including trophic relationships) of retained bycatch species beyond natural variability. Species capacity to increase from a depleted state is negligibly impacted.	The management framework (e.g. decision rules) is such that, where precautionary approach reference points are available, the target fishing mortality (F_{target}) is well below either the fishing mortality reference point (F_{lim}), when the stock is in the Healthy Zone, or the fraction of F_{lim} believed to be sustainable, when the stock is in the cautious zone. Furthermore, there is a low risk that F_{target} may be exceeded by a relatively large amount due to quota overruns, unreported fishing or illegal fishing. In the absence of a fishing mortality reference point there is a reasonable expectation that removals will be small with respect to the size of the stock and its productivity. Evidence in support includes: (1) small catches relative to fishable biomass estimates from surveys; (2) the fishery occurring in a small and marginal portion of the stock distribution; and/or (3) an increasing trend in post-recruitment abundance. In the case of bycatch species (including species at risk), additional evidence is an inferred of the bycatch species which is higher than the natural mortality of the target species, unless the bycatch species is suspected of having higher catchability to the gear. Species (including species at risk) that are the target of one or more other fisheries, removals are small relative to removals in the targeted fisheries and are therefore very unlikely to in and of themselves produce an unsustainable fishing mortality. For species assessed as “at risk” by COSEWIC, recovery potential assessment of species at risk indicates fishing pressure as an unlikely obstruction to recovery.

Nominal	Consequence	Notes and Examples
3	Fishery causes moderate impacts to population size, recruitment, range or dynamics (including trophic relationships) of retained bycatch species, but stock level depletions/extinctions and/or range contractions are not anticipated. Species capacity to increase from a depleted state may be adversely impacted, particularly if monitoring is inadequate.	The management framework is such that, where precautionary approach reference points are available, the target fishing mortality (F_{target}) is near or below the fishing mortality reference point (F_{lim}) when the stock is in the Healthy Zone or the fraction of F_{lim} believed to be sustainable when the stock is in the cautious zone. Furthermore, there is a moderate risk that F_{target} may be exceeded by a relatively large amount due to quota overruns, unreported fishing or illegal fishing. In the absence of a fishing mortality reference point there is a reasonable expectation that removals will be moderate with respect to the size of the standing stock and its productivity. Evidence in support includes: (1) catches relative to fishable biomass estimates from surveys that suggest an F that is close to or below 0.4 (elasmobranchs) or 0.8 (teleosts) times the species natural mortality (M); (2) the fishery occurring not just in areas representing a marginal portion of the stock distribution; and/or (3) a stable trend in post-recruitment abundance. In the case of bycatch species (including species at risk), additional evidence is an inferred natural mortality of the bycatch species that is similar to the natural mortality of the target, unless the bycatch species is suspected of having higher catchability to the gear. Species (including species at risk) that are the target of one or more other fisheries, removals are moderate relative to removals in the targeted fisheries and may, in and of themselves, produce an unsustainable fishing mortality on occasion. For species assessed as “at risk” by COSEWIC, the recovery potential assessment of species at risk indicates fishing pressure as a possible obstruction to recovery.
4	Fishery causes significant impacts to population size, recruitment, range and/or dynamics (including trophic relationships) of retained bycatch species such that stock level depletions / extinctions and / or severe range contractions occur. Species capacity to increase from a depleted state is adversely impacted.	The management framework is such that there is a high likelihood that the realized fishing mortality could exceed levels deemed sustainable in some years, by design and / or due to a moderate to high risk that F_{target} may be exceeded by a relatively large amount due to quota overruns, unreported fishing or illegal fishing. In the absence of a fishing mortality reference point there is a reasonable expectation that removals can be large with respect to the size of the stock and its productivity. Evidence in support includes: (1) catches relative to fishable biomass estimates from surveys that suggest an F that is close to or above 0.4 (elasmobranchs) or 0.8 (teleosts) times the species natural mortality (M); (2) the fishery occurring throughout the stock area or concentrated in areas of stock concentration; and / or (3) a decreasing trend in post-recruitment abundance. In the case of bycatch species (including species at risk), additional evidence is an inferred natural mortality of the bycatch species that is lower than the natural mortality of the target. Species (including species at risk) that are the target of one or more other fisheries, the fishery will be considered high consequence if (1) the target fishery is considered high consequence and the bycatch levels are more than just negligible, or (2) the target fishery is considered moderate consequence but the bycatch levels may, in and of themselves, produce an unsustainable fishing mortality. For species assessed as “at risk” by COSEWIC, recovery potential assessment of species at risk indicates fishing pressure as a likely contributor to continued population decline.

10.1.4 Description of required quality as a function of risk level

The draft RST includes the following table that links the requirements of estimation process quality with the risk levels for risk factors related to catch (target species and bycatch). We have provided revisions to parts of the text that are likely incorrect with respect to the intention in the draft RST.

Appendix Table 1.6. Link of the requirements of the estimation process quality with the risk levels to target, bycatch, species at risk, and community for the risk factors related to catch (target species and bycatch).

Components	Low Risk	Moderate Risk	High Risk
Risk to	target / bycatch / species at risk / community	target / bycatch / species at risk / community	target / bycatch / species at risk / community
Requirements for quality	No specific requirements for quality (precision and accuracy) of catch data. While there is a low risk of not meeting the objective, the fishery dependent reporting must provide adequate* information to estimate catch for the purposes of meeting the objective for the component.	The monitoring program should be such as to produce an estimate of catch and non-retained catch for which the precision is sufficient to have a reasonable* likelihood of being able to determine whether the objective for the component is being met.	The monitoring program adopted should be such as to produce an estimate of catch and non-target catch for which the precision is sufficient to have a high* likelihood of being able to determine whether the objective for the component is being met.
Information needs	No specific requirements for quality (precision and accuracy) of catch data.	The sampling design is such that accuracy is as high as possible and bias is limited.	The monitoring program should include as close to a full census of catch and non-retained catch as possible, with sampling design demonstrated to have high accuracy and known to be theoretically unbiased.
Revised text	None proposed	Given that the quality requirement addresses precision, the information needs should deal only with bias to avoid some redundancy. Revision to information needs: The sampling design is such that bias is limited.	Given that the quality requirement addresses precision, the information needs should deal only with bias to avoid some redundancy. Revision to information needs: The monitoring program should include as close to a full census of catch and non-retained catch as possible, with sampling design demonstrated be theoretically unbiased.

* The words ‘adequate’, ‘reasonable’ and ‘high’ in the draft RST table need to be defined and qualified with respect to the determination of dependability. This task constitutes part of the objectives of this research document.

10.1.5 References cited in Appendix I

DFO. 2009a. Policy on Managing the Impacts of Fishing on Sensitive Benthic Areas (Date modified: 2009-03-23).

DFO. 2009b. A Fishery Decision-Making Framework Incorporating the Precautionary Approach. (Date modified: 2009-03-23).

DFO. 2019a. Ecological Risk Assessment Framework (ERAF) for Coldwater Corals and Sponge Dominated Communities (Date modified: 2019-09-27).

DFO. 2019b. Policy on Managing Bycatch (Date modified: 2019-09-27).

10.2 APPENDIX II: REVISED CONSERVATION RISK CONSEQUENCE DESCRIPTORS IN THE RST

Appendix Table 2.1. Proposed revised consequence descriptors for the impact on the population from all fisheries and for all categories of removals (retained and discarded removals for both targeted catch and bycatch).

Nominal	Consequence	Descriptors
1	Fisheries cause negligible impacts to population size, recruitment range or dynamics (including trophic relationships), generally within the variation due to natural variability.	**Precautionary approach in place** • The management framework (e.g. decision rules) is such that the target fishing mortality (F_{target}) is well below the fishing mortality reference point (F_{lim}) when the stock is in the Healthy Zone, or the fraction of F_{lim} believed to be sustainable when the stock is in the Cautious Zone. • Furthermore, there is no expectation that F_{target} may be exceeded by a relatively large amount due to quota overruns or unreported catches that would cause fishing mortality to approach unsustainable levels. • The stock is most likely to be in the Healthy Zone, though it may be in the Cautious Zone due to natural variability.
		No precautionary approach in place i) A proxy for F_{lim} can be defined The inferred fishing mortality rate is well below the proxy for F_{lim} defined as follows: • proxy-F_{lim}=1.5 x proxy-F_{MSY} where, • proxy-F_{MSY} = 0.87 x natural mortality (teleost fish), or • proxy-F_{MSY} = 0.41 x natural mortality (elasmobranchs) ii) A proxy for F_{lim} cannot be defined In the absence of a proxy for the fishing mortality reference point there is a reasonable expectation that removals will be negligible with respect to the size of the stock and its productivity. Evidence in support includes more than one of the following: • negligible catches relative to fishable biomass estimates from surveys; • convincing evidence that the catchability of the population to the fishery is very low; • convincing evidence that the fishery selects only for highly abundant juvenile stages associated with high natural mortality; • the fishery occurs in a marginal portion of the population distribution and outside any biologically sensitive time periods for the population; • the population does not display an aggregative behavior that could accidentally result in overfishing in any given year; and • there is an increasing trend in post-recruitment abundance, provided the stock is not severely depleted and the index of abundance is reliable and is tracking abundance well.
		Additional considerations • In the case of a population caught only as bycatch additional evidence is an inferred natural mortality rate of the bycatch species that is much higher than the natural mortality of the target species (thus indicating higher productivity under natural conditions), unless the bycatch species is suspected of having higher catchability to the gear. • In the case of a principally discarded species, post-release survival is expected to be very high. • For species assessed as “at risk” by COSEWIC, recovery potential assessment of species at risk does not indicate fishing pressure as obstruction to recovery.

Appendix Table 2.1 (continued). Proposed revised consequence descriptors for the impact on the population from all fisheries and for all categories of removals (retained and discarded removals for both targeted catch and bycatch).

Nominal	Consequence	Descriptors
2	Fisheries cause minor impacts to population size, recruitment, range or dynamics (including trophic relationships) beyond variation due to natural variability. The population's capacity to increase from a depleted state is not impacted.	**Precautionary approach in place** • As above, F_{target} is well below F_{lim} for stocks in the Healthy Zone, or a fraction of F_{lim} for stocks in the Cautious Zone. • Furthermore, it is unlikely that F_{target} may be exceeded by a relatively large amount due to quota overruns or unreported catches that would cause fishing mortality to approach unsustainable levels.
		No precautionary approach in place *i) A proxy for F_{lim} can be defined* • The inferred fishing mortality rate is below the proxy for F_{lim} *ii) A proxy for F_{lim} cannot be defined* In the absence of a proxy for the fishing mortality reference point there is a reasonable expectation that removals will be small with respect to the size of the stock and its productivity. Evidence in support includes more than one of the following: • small catches relative to fishable biomass estimates from surveys; • convincing evidence that the catchability of the population to the fishery is low; • convincing evidence that the fishery selects mainly for highly abundant juvenile stages associated with high natural mortality; • the fishery occurs in a small portion of the population distribution and outside any biologically sensitive time periods for the population; • the population generally does not display an aggregative behavior that could accidentally result in overfishing in any given year; and • there is an increasing trend in post-recruitment abundance, provided the stock is not severely depleted, the index of abundance is reliable, and is tracking abundance well.
		Additional considerations • In the case of a population caught only as bycatch additional evidence is an inferred natural mortality rate of the bycatch species that is higher than the natural mortality of the target species (thus indicating higher productivity under natural conditions), unless the bycatch species is suspected of having higher catchability to the gear. • In the case of a principally discarded species, post-release survival is expected to be high. • For species assessed as "at risk" by COSEWIC, recovery potential assessment of species at risk indicates fishing pressure as an unlikely obstruction to recovery.

Appendix Table 2.1 (continued). Proposed revised consequence descriptors for the impact on the population from all fisheries and for all categories of removals (retained and discarded removals for both targeted catch and bycatch).

Nominal	Consequence	Descriptors
3	Fisheries cause moderate impacts to population size, recruitment, range or dynamics (including trophic relationships) beyond variation due to natural variability. The population's capacity to increase from a depleted state may be adversely impacted.	**Precautionary approach in place** • As above, F_{target} is near or below F_{lim} for stocks in the Healthy Zone, or a fraction of F_{lim} for stocks in the Cautious Zone. • Furthermore, F_{target} may be exceeded by a relatively large amount in some years due to quota overruns or unreported catches that would cause fishing mortality to exceed sustainable levels.
		No precautionary approach in place i) A proxy for F_{lim} can be defined • The inferred fishing mortality rate is close to, yet below, the proxy for F_{lim} ii) A proxy for F_{lim} cannot be defined In the absence of a proxy for the fishing mortality reference point there is a reasonable expectation that removals will be moderate with respect to the size of the stock and its productivity. Evidence in support includes more than one of the following: • moderate catches relative to fishable biomass estimates from surveys; • evidence that the catchability of the population to the fishery is moderate; • the fishery selects mainly for life-history stages whose loss may hinder productivity (e.g. mature individuals); • the fishery occurs in a moderate portion of the population distribution and/or during a time that may overlap with a biologically sensitive period for the population; • the population displays an aggregating behaviour that could result accidental overfishing in some years; and • there is a stable trend in post-recruitment abundance, provided the index of abundance is reliable, and is tracking abundance well.
		Additional considerations • In the case of a population caught only as bycatch additional evidence is an inferred natural mortality rate of the bycatch species that is similar to the natural mortality of the target species. • In the case of a principally discarded species, post-release survival is expected to be moderate. • For species assessed as "at risk" by COSEWIC, the recovery potential assessment of species at risk indicates fishing pressure as a possible obstruction to recovery.

Appendix Table 2.1 (continued). Proposed revised consequence descriptors for the impact on the population from all fisheries and for all categories of removals (retained and discarded removals for both targeted catch and bycatch).

Nominal	Consequence	Descriptors
4	Fisheries cause significant impacts to population size, recruitment, range and/or dynamics (including trophic relationships) leading to eventual population depletions and / or range contractions, and possibly enhanced risk of local extirpation. Species capacity to increase from a depleted state is adversely impacted.	**Precautionary approach in place** • The management framework is such that there is a high likelihood that the realized fishing mortality could exceed levels deemed sustainable in some years, by design (e.g. $F_{target} \approx F_{lim}$) or F_{target} is likely to be exceeded by a relatively large amount in some years due to quota overruns or unreported catches that would cause fishing mortality to exceed sustainable levels.
		No precautionary approach in place i) A proxy for F_{lim} can be defined • The inferred fishing mortality rate is at or above , the proxy for F_{lim} ii) A proxy for F_{lim} cannot be defined In the absence of a proxy for the fishing mortality reference point there is a reasonable expectation that removals will be large with respect to the size of the stock and its productivity. Evidence in support includes more than one of the following: • large catches relative to fishable biomass estimates from surveys; • evidence that the catchability of the population to the fishery is moderate to high; • the fishery selects for life-history stages whose loss may hinder productivity (e.g. mature individuals); • the fishery occurs in a large portion of the population distribution and/or during a time that may overlap considerably with a biologically sensitive period for the population; • the population displays an aggregating behaviour that is expected to result in accidental overfishing in a some years; and • there is a declining trend in post-recruitment abundance, provided the index of abundance is reliable, and is tracking abundance well.
		Additional considerations • In the case of a population caught only as bycatch additional evidence is an inferred natural mortality rate of the bycatch species that is lower than the natural mortality of the target species. • In the case of a principally discarded species, post-release survival is expected to be low. • For species assessed as “at risk” by COSEWIC, recovery potential assessment of species at risk indicates fishing pressure as a likely contributor to continued population decline.