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Promising Indicators of Fisheries Productivity for the Fisheries Protection Program Assessment Framework

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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

Amendments to the *Fisheries Act* in 2012 effectively changed the focus of promoting fisheries sustainability in Canada from managing the habitat that sustains fish populations (Fish Habitat Management Program - FHMP) to managing the ongoing productivity of fish stocks related to Commercial, Recreational or Aboriginal (CRA) fisheries (Fisheries Protection Provisions - FPP). The new and central role of fisheries productivity is clearly stated in Section 6 of the *Fisheries Act* such that the Minister must consider "the contribution of the relevant fish to the ongoing productivity of commercial, recreational or Aboriginal fisheries" when evaluating the capacity of a proposed development or activity to cause "the death of fish or any permanent alteration to, or destruction of, fish habitat". Since the amendment of the Act, the science and policy divisions of Fisheries and Oceans Canada (DFO) have been providing technical advice on the implications of the new focus on the FPP.

The topic of this review is to discuss and evaluate appropriate indicator metrics that can link the changes in the components to productivity to a qualitative or quantitative change in CRA fisheries productivity. For a consistency of terminology within this paper, we will define "indicators" as metrics which have a direct link to CRA fisheries productivity, thus an indicator can also be a component of productivity metric. Advice from the recent drafts of the proposed FPP framework (Bradford et al. 2013, Koops et al. unpublished manuscript[1]) has suggested some desirable qualities in potential indicators.

This literature review has gained insight into the use of different metrics by other fields of fisheries research, management and biomonitoring and also in their theoretical links to "ongoing productivity of CRA fisheries". It sounds simplistic to say that any indicator can be linked to fisheries productivity given a proper set of assumptions, but this is generally true and a great strength of this field of research. The many linkages back to the intrinsic rate of population growth, carrying capacity or steepness coefficient offers flexibility to proponents to demonstrate quantitatively, if needed, how proposed alterations to habitat is expected to affect the CRA fisheries. In the great majority of cases, we would expect that proponents would measure only the indicator and reference the specific qualitative linkages proposed in this review and others. A common theme that has emerged throughout the literature review is that there is likely no "one-size-fits-all" indicator, or even set of indicators. This is not surprising as considerable effort was expended under the FHMP to find simple and ubiquitous metrics, yet it also proved challenging. A suite of indicators is likely most effective, but will also only be required for a particular size of project. Further, through the search for specific indicators we have discovered that some are already well parameterized for Canadian fish, offering reference points and relationships (often allometric) with other indicators.

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[1] Koops, M.A., Randall, R.G., Clarke, K.D., Enders, E.C., Smokorowski, K.E., Doka, S.E., Watkinson, D.A., and Bradford, M.J. A review of scientific evidence supporting generic productivity-state response curves. DFO Can. Sci. Advis. Sec. Working Paper (March 2013).

Indicateurs prometteurs de la productivité des pêches pour le cadre d'évaluation du Programme de protection des pêches

RÉSUMÉ

Les modifications à la *Loi sur les pêches* en 2012 ont effectivement changé l'objectif de la promotion de la durabilité des pêches au Canada : de la gestion de l'habitat qui soutient les populations de poissons (Programme de gestion de l'habitat du poisson) que l'on faisait jusque-là, il s'agit maintenant de gérer la productivité continue des stocks de poissons visés par les pêches commerciales, récréatives ou autochtones (CRA) (dispositions en matière de protection des pêches). Le nouveau rôle central de la productivité des pêches est clairement énoncé à l'article 6 de la *Loi sur les pêches*, tel que le ministre doit tenir compte de « la contribution des poissons visés à la productivité des pêches commerciales, récréatives ou autochtones » lorsqu'il évalue la capacité d'un projet de développement ou d'une activité proposé de causer « la mort de tout poisson ou la modification permanente ou la destruction de son habitat ». Depuis la modification de la loi, les divisions des Sciences et des politiques de Pêches et Océans Canada (MPO) ont fourni des conseils techniques sur les répercussions du nouvel objectif des dispositions en matière de protection des pêches.

Le présent examen vise à aborder et à évaluer les paramètres d'indicateur appropriés qui peuvent établir le lien entre les changements dans les composantes de la productivité et les changements qualitatifs ou quantitatifs dans la productivité des pêches CRA. Aux fins d'uniformité de la terminologie du présent document, les « indicateurs » sont des paramètres ayant un lien direct avec la productivité des pêches CRA; un indicateur peut donc être une composante des mesures de la productivité. Un avis découlant des récentes ébauches du cadre proposé des dispositions en matière de protection des pêches (Bradford *et al.* 2013; Koops *et al.* manuscrit non publié[1]) souligne certaines qualités que devraient avoir les indicateurs.

Cette analyse documentaire a donné un aperçu de l'utilisation de différents paramètres dans d'autres domaines de la recherche sur les pêcheries, de la gestion et de la biosurveillance halieutiques, ainsi que de leurs liens théoriques avec la « productivité continue des pêches CRA ». Il peut paraître simpliste de dire que n'importe quel indicateur peut être lié à la productivité des pêches s'il est établi en fonction d'un ensemble d'hypothèses adéquat, mais ceci est généralement le cas et c'est là une grande force de ce domaine de recherche. Les nombreux liens avec le taux intrinsèque de croissance de la population, la capacité de charge ou le coefficient directeur permet aux promoteurs de démontrer quantitativement, au besoin, quel devrait être l'impact des modifications proposées à l'habitat sur les pêches CRA. Nous nous attendons à ce que, dans la grande majorité des cas, les promoteurs ne mesurent que l'indicateur et fassent référence aux liens qualitatifs précis proposés dans cet examen et ailleurs. Un thème commun qui est ressorti de l'analyse documentaire est qu'il n'existe vraisemblablement pas un indicateur « unique et universel », ou même ensemble d'indicateurs. Cela n'est pas surprenant, car un effort considérable a été déployé dans le cadre du Programme de gestion de l'habitat du poisson pour trouver des mesures simples et omniprésentes, ce qui s'est tout de même révélé difficile. Une suite d'indicateurs serait sans doute la solution la plus efficace, mais elle ne sera aussi nécessaire que pour des projets d'une

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certaine envergure. De plus, durant la recherche d'indicateurs précis, nous avons découvert que certains indicateurs étaient déjà bien paramétrés pour les poissons du Canada, fournissant des points de référence et des liens (souvent allométriques) avec d'autres indicateurs.

INTRODUCTION AND INSIGHTS

Amendments to the *Fisheries Act* in 2012 effectively changed the focus of promoting fisheries sustainability in Canada from managing the habitat that sustains fish populations (Fish Habitat Management Program - FHMP) to managing the ongoing productivity of fish stocks related to Commercial, Recreational or Aboriginal (CRA) fisheries (Fisheries Protection Provisions - FPP). The new and central role of fisheries productivity is clearly stated in Section 6 of the *Fisheries Act* such that the Minister will weigh "*the contribution of the relevant fish to the ongoing productivity of commercial, recreational or Aboriginal fisheries"* when evaluating the capacity of a proposed development or activity to cause "*the death of fish or any permanent alteration to, or destruction of, fish habitat"*. Since the amendment of the Act, the science and policy divisions of Fisheries and Oceans Canada (DFO) have been providing technical advice on the implications of the new focus on the FFP including:

- The identification of species and habitats that support CRA fisheries (Kenchington et al. 2012);
- The detailed interpretation of the phrase "ongoing productivity of CRA fisheries" (Randall et al. 2012);
- the detailed interpretation of the phrase "contribution of relevant fish to the ongoing productivity of CRA fisheries" (Koops et al. 2013); and,
- A review of productivity-state response curves, which effectively link environmental change to changes in fish traits and fisheries qualities (Koops et al. unpublished manuscript[1]);

This series of technical advice by DFO is culminating into a promising framework to assess fisheries productivity under the FPP (Bradford et al. 2013). However one of the challenges of implementing the FPP is determining how impacts to the ongoing productivity of CRA fisheries will be measured. In the previous FHMP, the productive capacity of fish habitat was the conceptual endpoint of assessing the potential and cumulative impacts of developments on fisheries sustainability (Goodchild 2004). However, productive capacity is not a parameter that can be easily measured so a wide range of environmental and ecological metrics were used to characterize the magnitude and direction of change (Minns et al. 2011). Similarly, under the new FPP framework measuring the impact of a development at the scale of a CRA fishery may be extremely difficult unless the project itself is large enough to create large ecosystem transformations (e.g. infilling of a watershed or lake). However, the impacts of smaller projects are not trivial and certainly the cumulative impact of disparate human activities on CRA fisheries is known to be great (Minns 2009) and responsible for large declines in freshwater biodiversity (Chu et al. 2003).

The FPP framework outlines a conceptual methodology to directly relate eight common impacts on aquatic habitat to local responses from fish and fish populations to non-trivial effects at the scale of a fishery (see Bradford et al. 2013). Twenty-one Pathways of Effects (PoE) provide key linkages between activities on or near water and the "PoE Endpoints" which are the probable responses of aquatic environments (e.g. a change in sediment concentrations leads to increased suspension of sediment in water column). The direction and magnitude of the impacts are derived from the environmental mechanisms associated with the PoE Endpoint, and therefore can characterize the state of the environment following a potential activity. Within the FFP framework, relationships have been described between the state of the environments and an aspect of a fish or a fish population that relates to production (i.e. state-production curves, see Koops et al. unpublished manuscript[1], and component of productivity, see Bradford et al.

2013). There are multiple sets of factors that influence the production of fish tissue or fish populations and so ecological mechanisms and vital rates are nested within each component of productivity, termed sub-components (e.g. Food Supply and Foraging Efficiency are sub-components of Growth). However components of productivity are conceptual and thus are themselves defined by quantitative, semi-quantitative or qualitative metrics (much like the representation of productive capacity in the previous FHMP). The state-production relationships are therefore based on the components of productivity's metric and the PoE endpoint as the dependent and independent axes, respectively. These relationships are generally non-linear and can involve both upper and lower thresholds.

A final step in this process requires a translation from local scales of fish tissue or fish population production to the generally regional scales of CRA fisheries production. Randall et al. (2012) defines fisheries productivity as "the sustained yield of all component populations and species and habitats which support and contribute to a fishery". Like the ability to apply straightforward metrics for assessing the mechanism from the PoE endpoints, it is important at this stage that either:

1. The components of productivity metrics are easy to measure and can relate directly to fisheries production, or
2. That another type of indicator can represent one or a set of metrics well enough to be used in assessment and monitoring programs, as well as also provide a direct relationship to fisheries productivity.

The topic of this review is to discuss and evaluate appropriate indicator metrics that can link the changes in the components to productivity to a qualitative or quantitative change in CRA fisheries productivity. For a consistency of terminology we will define "indicators" as metrics which have a direct link to CRA fisheries productivity, thus an indicator can also be a component of productivity metric. Advice from the recent drafts of the proposed FPP framework (Bradford et al. 2013, Koops et al. unpublished manuscript[1]) has suggested some desirable qualities in potential indicators:

- "the choice of metric would depend on the risk to fisheries productivity, data availability, the spatial scale and nature of the project";
- Policy guidelines should "ensure a consistency in approach regardless of location or spatial scale";
- The majority of project impacts will occur below the spatial scale of fisheries, therefore "metrics for assessments are more likely to be surrogates for effects on productivity that can be potentially scaled up to effects on fisheries yield";
- "Surrogate metrics of productivity should inform users of potential impacts to fishery productivity, regardless of project type";
- Potential metrics should be derived from general models of fish life history (Shuter et al. 1998, Hayes et al. 2009);
- Dealing with density dependence requires that "the presence of compensatory processes will not affect the choice of metrics for the assessment" yet requires the recognition that it could alter "the severity of the impact a project may have on fisheries productivity" (citing Shuter 1990);
- Qualitative or heuristic indicators are likely appropriate when quantitative estimates are unavailable or if the expected size of the impact is relatively small. When using qualitative approaches, preferred methods would include "causal linkage analysis" or a "weight of

evidence approach" (Koops et al. 2013, unpublished manuscript[1]). Heuristic and qualitative methods could provide directions and severities of fisheries productivity change, respectively;

- Quantitative indicators would be used to predict the scale and magnitude of the projects impacts as well as the relationship between the component of productivity and CRA fisheries. Importantly, quantitative indicators can often identify thresholds; and,
- Indicators, metrics and components of productivity will likely require some estimate of space on an aerial basis (i.e. m^2 or ha).

The need for a review and evaluation of potential indicators to represent components of productivity and their metrics prompted Bradford and colleagues (2013) to highlight some specific recommendations for the FPP program:

- "The best indicators are those that provide sufficient information for effective decision making, but no more" (from Failing and Gregory 2003);
- At least 3 considerations are critical when evaluating if a measure of productivity will be an effective indicator to CRA fisheries productivity, "1) the magnitude of change in the component of productivity, 2) the sensitivity of CRA fisheries productivity to a change in the component, and 3) the scale at which the assessment will be conducted";
- It is important to determine if different indicators have an additive or synergistic relationship, or if alternatively, one indicator can overwhelm the effects of the others; and
- The choice of indicators should also address the current states and biodiversity of the CRA fisheries and fish that support it (Koops et al. 2013, Randall et al. 2012).

This advice sets the stage for this following review. Much of the effort to date in developing the framework for the FPP program has arisen from the extensive knowledge base DFO accumulated during the FHMP. While there are excellent reviews of direct linkages between habitat and productivity (Minns et al. 2011), there is a danger of missing the key relationships between production at the scale of the individual or population and the scale of the fishery if the focus starts at the habitat level. The objective of the FPP program is to address the "ongoing productivity of CRA fisheries", and so it is important to make sure that processes that affect fisheries, rather than fish populations, are well addressed. So instead this review focusses much more on scaling down from fisheries studies to fish populations rather than scaling up from studies on the consequences of environmental impact which often ends at populations. Given this focus the review begins with a summary of insights from the major fields of fisheries research and management including:

1. The Food and Agriculture Organization (FAO) of the United Nations' and the International Council for the Exploration of the Sea's (ICES) 20th century fisheries reference points,
2. The use of stock assessments to manage commercial fisheries,
3. The evolution of fisheries related ecological traits , and
4. The recent development of a 21st century ecosystem-based fisheries management framework.

With these collective insights the next chapter of the review examines the key parameters needed to understand the "ongoing productivity" of any fishery. The review works backwards to gain insight from the most common methods and models that are used to estimate the key fisheries parameters. These are the "links" from a wide range of metrics to fisheries dynamics. Then, the chapter explores which common indicators suggested by Bradford et al. (2013) at the scale of fish or fish populations are best related to fisheries dynamics through the reviewed "links". Finally the chapter discusses how other factors can modify the effects on indicators on productivity including density dependence and biodiversity. The final chapter of the review provides recommendations on the most appropriate indicators as well as addresses how to use them.

While indicators are of great use for monitoring change, there is also a need to characterize potential reference points which can inform management decisions. For example, in classic fisheries models, the Maximum Sustainable Yield (MSY) puts an upper limit on the harvested biomass. Further, other simpler indicators such as 20% of the virgin biomass or a maximum fishing mortality can inform a manager of when the stock may be fished unsustainably (Methratta and Link 2006). Therefore the objective of this review will be to also include suggestions on how to develop indicator reference points under the FPP.

20TH CENTURY FAO AND ICES FISHERIES REFERENCE POINTS

Fisheries reference points are as old as some of the first fisheries models because they provided criteria to set quotas and monitor stock responses to harvests. One of earliest, yet still common, reference point is the MSY which is the annual and repeated harvest rate that yields the most return without negatively impacting stock productivity. The MSY is based on the simple concept that any removal from a stock would need to be replenished by some sort of biological process (e.g. logistic population growth) before the next harvest. Over its lifetime the use of the MSY has received criticism because it attempts to place a static value on an inherently dynamic process yet its conceptual simplicity, strong theoretical foundations and a direct application to management has allowed it to endure.

Over the latter half of the 20th century the United Nations held numerous conventions on developing regulations and procedures for harvesting from the ocean. Recognizing that ocean fisheries were a common property harvested by many different regulatory entities, the need for clear rules and reference points were paramount. Even before some of the more dramatic stock collapses from the 20th century the 1958 United Nations Ocean Convention in Geneva proposed that maximum yields should be based on "economic, social and biological values… rather than being limited to maximizing net profits [through MSY]". While it was not widely attempted at the time, the idea that reference points could be based on multiple and sometimes unrelated parameters was recognized relatively early in the history of fisheries management. Thirty years later, the ICES Advisory Committee on Fishery Management reported that "biological reference points are intended to provide guidance concerning management, and that no biological reference point can serve as a universal target". Thus a range of Target Reference Points for managers to aim for and Limit Reference Points for managers to avoid or use were adopted, as well as the recognition that a "one-size fits all" model was unrealistic.

The development of Fisheries Reference Points in the 20th century can provide some insight into evaluating indicators under the FPP. The following insights were taken from FAO documents designed to teach and promote standardized fisheries management across the globe (e.g. Caddy and Mahon 1995, Cadima 2003):

- The main reference points were based on yields and theoretically justified with Schaefer's surplus production model or a Beverton-Holt style stock recruitment model. The only exception to these reference points was size at age metrics which were based on simple rules and assumptions (e.g. limiting catch to individuals greater than the average size at maturity would ensure that 50% of the stock could still reproduce). Data needs for most reference points included catch biomass, fish sizes at their ages of maturity, and mortality rates (both natural and fishing related).

- Mortality rates are extremely hard to measure in natural environments, whereas catch rates are usually well estimated (even though misreporting and discards occur in commercial catch statistics).
- Reference points may be better judged by their ability to provide conservative advice under conditions of uncertainty, than by the robustness of their theoretical underpinnings (Note: similar arguments were made in support of the biological basis of the habitat based Morpho-Edaphic Index; Ryder 1982).
- Reference points for highly migratory stocks are no different than those for resident stocks. The challenge is in the application of the reference point rather than its derivation.
- Managers using fishing mortality based reference points will encounter problems when harvests contain multiple species. The life-history (e.g. growth rate or fecundity) of each species will need to be addressed or the overall reference point will lead to the overexploitation of some species, and underexploitation of others.
- Harvesting both predators and prey will require a very precautionary approach. For example an arbitrary reference point for capelin off eastern Canada would be no more than 10% of the spawning biomass in recognition that the natural mortality rate from predators should be quite high.
- A significant reduction of biomass from keystone species (e.g. a top predator or a species that fulfills a critical role for keeping the ecosystem functioning) could bring about extremely fast and irreversible change.
- There are five main sources of uncertainty in the calculation of reference points: Measurement, Process, Model, Estimation and Implementation errors. Note that sources of uncertainty include not only statistical error in detecting stock status and environmental trends or errors in population analysis, but also wrong decisions and an inefficient management framework.

While the target reference points that were commonly employed were simple, it is interesting to note that the need for a more holistic approach were prescient of the more modern ecosystem approach. These insights are in a general agreement with a few key directives from the emerging FPP framework:

1. A range of simple indicators will likely be required to address both the changes in individual fish populations (i.e. stock effects) and the changes across communities of fish species (i.e. ecosystem effects),
2. Confounding factors such as density dependence or stock migration may best be applied to indicators rather than make up individual indicators, and
3. Characterizing the power of the indicator to accurately and precisely (in a qualitative and statistical sense) represent CRA fisheries productivity is important.

STOCK ASSESSMENT PARAMETERS

Stock assessments are critical to understanding the current state of a fishery and estimate the effect of future harvests on the ongoing productivity of the fish populations within it. Like the FAO and ICES reference points, stock assessment models are typically based on a plausible demographic model of the fish's ecology (Sparre and Venema 1998, Kuparinen et al. 2012b). There are many different theoretical models that form the foundation of stock assessments, but they all generally work off the biomass that was caught under the effort the commercial boats expended (Schnute 1985). The main difference between the range of models available to managers is how each of them addresses fish recruitment (i.e. the successful hatch of fish, or also the number that survive to a marketable size), fish growth and natural mortalities (Magnusson and Hilborn 2007). Thus managers can build these models with a limited data set including commercial catch biomass, distributions of fish lengths, weight and age, and a small mixture of other scientific survey data (Kuparinen et al. 2012b). Magnusson and Hilborn (2007) point out that these different data types complement each other such that the catch informs on the scale of the fishery, the age-composition of the catch informs on the relative cohort size, and the abundance-index data informs on the relative changes in abundance over time. Because a key function of a stock assessment is to inform quotas, the intrinsic rate of population growth and/or the stock-recruitment relationship that underlies the model is of critical importance (Myers 2001), and will likely be of importance for the FPP program as well.

The best estimates of these population characteristics may come only from recovery periods following overfishing (Cotter et al. 2004, Magnusson and Hilborn 2007). From the perspective of the FPP, the best estimates would therefore arrive too late, as permanent environmental impacts are often not as easy to reverse as fishing effort. While the stock-recruitment relationship is the most important to stock assessments, it is also the most stochastic. Simulations (e.g. Magnusson and Hilborn 2007) and meta-analyses (e.g. Myers 2001) tested the strength of some of the key parameters for stock assessments to influence the outcomes of the models with the following general conclusions:

- High fecundities do not add much variation to stock-recruitment curves (contrary to popular belief);
- Effects of spawner abundance on recruitment is variable (and increasingly so at low adult abundances) suggesting strong density-dependent mortality on juveniles;
- Intercohort density dependent effects are generally weak; and,
- Density dependent effects related to predation are stronger in freshwater than marine systems (Note: Cotter and colleagues in 2004 found that mortality estimates based on predation are generally uncertain).

Recent insights from the evaluation of stock assessment methods generally caution that the models used are often more elaborate than can be justified by the quality of the data (Cotter et al. 2004). However, alternatives exist and it stands to reason that even simple empirical metrics should have direct linkages to the expected population dynamics whether they are modelled or not (Schnute and Richards 1998). For example, size-based stock assessments have been identified as are an alternate way forward and have been shown to be effective at modelling the growth trajectories of individual fish (Pope et al. 2006, Froese et al. 2008, Andersen and Beyer 2013). Notwithstanding simpler measures, Cotter and colleagues (2004) provide a set of ideals to follow when choosing stock assessment models that balance the reluctance to adopt over-parameterized models with the need for defensible methods (Table 1).

Table 1. Ideal elements of a stock assessment model

Ideals	**Detail**
Make Simple, Parsimonious Models	Be explicable to the fishing industry and other interested, non-mathematical parties
	Favour parsimony as a smaller set of parameters are easier to fit and justify, and are less likely to mistake sampling variation for an important biological process
	Be calculable in less than 1 hour
Favour Independent Data	Weight each data set in the assessment in relation to how much independent information it contributes, and to how much information it shares with other data sets being used.
	Use no sampling data or estimates more than once
	Make due allowance for dependence among data when estimating standard errors and statistical significance.
Minimize subjective judgement	Subjective decisions reduce the scientific repeatability of an assessment, undermine theory, and make the assessment harder to explain.

FISHERIES LIFE HISTORY TRAITS

Fisheries reference points and stock assessments may recognize that fish traits change overtime but because they rarely include time series longer than 3 or 4 decades they rarely incorporate how the life-history of fish (i.e. the behaviour and physiology that determine how they develop and reproduce throughout their life) can be evolving under the pressure of the commercial harvest (Stenseth and Dunlop 2009). The weight of a catch can be made up of many small individuals or a few large individuals, so it’s not always obvious that either a) a mature fish is getting smaller over time, or b) a particular size of fish is more vulnerable than another. Further, studies of life history evolution typically require addressing variation among the growth rates of individuals within a population, which is a scale that stock assessments rarely examine.

Age-specific probabilities of survival and rates of growth and fecundity are all influenced by a fish’s surrounding environmental and ecological factors, as well as by each other (Koops et al. 2004, Kuparinen et al. 2012a). Studies of evolutionary ecology demonstrate many examples of trade-offs between life history strategies in different ecological conditions (Winemiller and Rose 1992, Lester et al. 2004, Winemiller 2005, Charnov et al. 2013). Under life history evolution, if an external factor increases mortality rates or harvests on only large individuals, species may gradually shift from the set of long-lived to short-lived traits to compensate for the loss of reproductive potential. There are two concerns from the perspective of using evolving life history traits (e.g. age of maturity) as indicators in the FPP framework:

1. The trait is only as important as the fish's growth strategy allows it to be (e.g. dwarf and trophy sized lake trout both contribute enough offspring to keep their numbers sustained in the same populations, the dwarves mature early and have few eggs but over a long life, and the trophies mature late and have lots of eggs but over a shorter time – here a declining age at maturity is more of a concern for the trophies because their success hinges on being big enough to make lots of eggs when they mature), and
2. Changes in life history traits may be a result of ongoing harvesting rates in active CRA fisheries yet be mistaken for the influence of a project (e.g. to continue the former example, if anglers only keep the trophies and release the dwarves it will appear to the FPP that the project is harming the fishery because monitoring programs only find fish with early ages at maturity).

Therefore the key to understanding the effect of life history trait evolution on fisheries productivity is therefore to uncover the fundamental link between individual life histories (commonly including body growth rates, age at maturity and mortality rates; Dulvy et al. 2004) and per capita rates of population growth (Kuparinen and Hutchings 2012). The field of fisheries life history evolution research is still quite young, however there are few insights from it that are valuable in the context of choosing indicators:

- Most lines of evidence for life history evolution have to do with fish size, and often involve fishing induced changes to life history traits including age specific weight or length, the growth rate or the size of maturity.
- Separating the environmental effects and the fishery effects on a fish's life history requires that both abiotic and biotic effects on growth rates be considered (Swain et al. 2007). The current FPP framework that leads environmental change to an expected change in a component of productivity metric would likely satisfy this insight. However, it re-enforces the need for a set of indicators, rather than one or two that might be both influenced strongly by life history evolution.
- Density dependence is an important mechanism in life history evolution (Kuparinen and Hutchings 2012) and so should be considered when assessing a potential impact of environmental change. This is also already considered under the emerging FPP framework and addressed in more detail later in this review.
- Large bodied and slow growing species with later age at maturity (e.g. Lake Sturgeon) will be much more sensitive to environmental impact and take longer for populations to recover. These types of species might make good long term indicator species in total biomass, abundance, density and average size.
- Kuparinen and her colleagues (2012) caution that "detailed life-history analyses may well be worthwhile to conduct for key species with particular economic, societal, and conservational relevance, such as Pacific (*Oncorhynchus spp.*) and Atlantic salmon (*Salmo salar*) and Atlantic cod (*Gadus morhua*), but for many species this may be neither practical nor feasible." For the FPP it is probably better to recognize life history evolution, measure indicators that could address it if there was a need, but not focus on it. This is similar to the suggestion of how the effect of density dependence should be addressed (Bradford et al. 2013).

ECOSYSTEM APPROACH INDICATORS

In recent decades it has become increasingly recognized that the failure of single-species fisheries management to prevent overfishing could be addressed by taking a more holistic, ecosystem-based approached. Although multispecies MSY estimates and predator-prey relationships have been incorporated into fisheries models before, the field of ecosystem-based management (EBFM) is relatively young and the search for appropriate metrics and reference points is ongoing (Fulton et al. 2004, 2005, Patrick et al. 2010). For this reason alone, insights from this growing field would be worth reviewing, however even more so, the principles and processes that governs ecosystem-based management at the scale of global fisheries (Jennings 2005) are almost identical to those stated in the FPP framework (Bradford et al. 2013). Thus the metrics considered by EBFM should also be considered by the FPP framework with the important caveat that while they both ultimately estimate fisheries productivity, the types of impacts and pressure they exert on fisheries are fundamentally different. Link and colleagues (2012) proposed a triad of drivers that affect fisheries productivity comprising of Trophodynamics, Biophysical Environment and Exploitation (Figure 1). Quite clearly the EBFM will consider indicators that link exploitation to fisheries productivity best (Rice and Rochet 2005), whereas the FPP framework may benefit most from considering links involving the two other drivers. It is worth noting from our previous analysis of life history evolution that exploitation is likely to affect large individuals from upper trophic levels, whereas environmental impacts have the potential to affect all sizes and trophic levels more evenly and subtly. However, as long as this caveat is kept in mind, there are many recent studies attempting to evaluate appropriate metrics of fisheries productivity (Fulton et al 2004b).

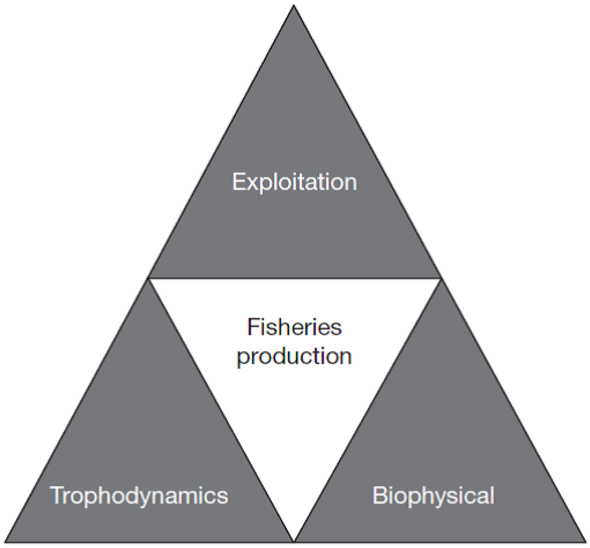


Figure 1. A triad of drivers that can affect fisheries production (from Figure 1 in Link and colleagues 2012).

In a fairly short time, a great deal of effort has been expended to both:

1. Determine a suitable framework for EBFM (Rice and Rochet 2005), and
2. Evaluate appropriate metrics to set reference points and monitor multispecies fish stocks and their environment (Fulton et al. 2005, Methratta and Link 2006, Shin et al. 2010).

Interestingly, the general patterns that emerge from the field’s search for a suitable framework re-enforce the conclusions this review has drawn so far from basic fisheries reference points and stock assessment models. Generally, the effects of exploitation need to be well understood up to the scale of a set of metrics that are directly related to fisheries productivity. Ideally this set of metrics would be relatively easy to measure, yield statistically robust estimates and cover a range of information. Rochet and Trenkel (2003) recommended three categories that a set of indicators need to address:

1. Population level indicators for each species within the fisheries (including non-target fish),
2. Assemblage level indicators that represent the combination of populations that live within a region but don't necessarily interact, and
3. Community level indicators that reflect the dynamics of networks of populations and species.

Fulton and colleagues (2005) completed simulations testing previous reviews they had conducted of over 100 potential indicators (Fulton et al. 2004). They generally found that no single indicator was adequate to characterize fisheries dynamics and rather a set of indicators would need to address four main biological groups:

1. Non-target species with fast turnover rates (e.g. plankton),
2. Fish species targeted by fisheries,
3. Habitat defining groups (e.g. marine coastal benthic species), and
4. Environmentally sensitive groups (e.g. upper trophic level species with slow population dynamics).

This same study tested a subset of promising indicators for robustness, defined as "consistency of performance across alternative ecosystem types, levels of perturbation intensity, and sampling uncertainty". The indicators best described by robustness and linking directly to fisheries productivity were related to:

1. Total biomass estimates across multiple groups,
2. Size at maturity of top predators,
3. Efficiency of consumption or average trophic levels,
4. Catch estimates of demersal fish (e.g. fish that feed on the lake or ocean floor), and
5. Total production and respiration (Fulton et al. 2005).

The analysis also revealed that most indicators were highly correlated with each other. Methratta and Link (2006) proposed eight indicators as a general set for evaluating marine fish community status which generally included biomass or length estimates for different functional or trophic groups of species.

A further insight from Fulton and colleagues' (2005) analysis was that indicators worked better when data was aggregated by zones then across the entire system. This could suggest that there are limits to scaling up a set of metrics across the size of projects (e.g. an interprovincial pipeline crossing multiple watersheds could represent an upper limit). Certainly any project that was expected to cross through difference ecozone (i.e. tundra to taiga to boreal forest) would require some sort of correction for the effect of latitudinal differences (Venturelli et al. 2010). Further, differences in indicator performance are noted between open (e.g. marine) and closed (e.g. freshwater) systems. Cross-ecosystem comparisons can be achieved by defining metrics as ratios between the observed indicator value and the theoretical indicator value in a pristine state (Shin et al. 2010). While predicting the characteristics of a stock in an unfished state is extremely difficult, under the FPP framework the existing habitat prior to the development could provide the pristine (or pre-impact) indicator values. This approach while routed in the EBFM literature could lead to indicators which mimic the type of indices that are more typically used in freshwater fish habitat studies (e.g. Index of Biotic Integrity, Habitat Suitable Indices) as well as the Before-After-Control-Impact (BACI) environmental effects monitoring designs (Underwood 2009).

The EBFM approach demonstrates that a set of indicators is more useful than a “one-size-fits-all” model. This is certainly comparable to findings from reviews and workshops on DFO’s previous FHMP which attempted to find habitat models that could be used to estimate productive capacity across a range of projects (Smokorowski and Derbowka 2008). This is because the relationship between a particular external pressure and the response from a fishery is rarely one to one (see Figure 2). Thus each EBFM study reviewed here recommended that multiple indicators always be considered with the objective to cover as many different aspects of the aquatic ecosystem as possible, including the effects of other anthropogenic factors (Rochet and Trenkel 2003, Fulton et al. 2005, Rice and Rochet 2005).

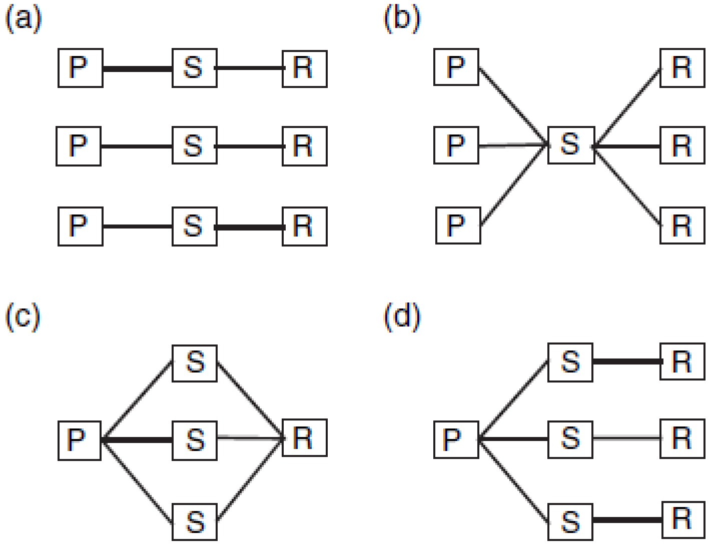


Figure 2. Possible relationships between pressure (P), state (S) and response (R). Relationships (b-d) illustrate that indicators for P, S and R are rarely expected to map one on one as in (a).

Finally, an emphasis on the need for simple metrics is re-enforced throughout the EBFM literature. Link (2012) suggests that relatively simple fish production models will generally give similar results to the much more data and computationally intensive age-structured models (citing Fogarty et al. 2012). The concept that one of the roles of indicators is to provide “reasonable advice” over hard-fast rules is also restated often. For this same reason metrics that involve biomass or fish sizes are preferred and expected to yield simple indicators that perform consistently regardless of the intensity of exploitation or the underlying ecosystem (Rochet and Trenkel 2003, Fulton et al. 2005). However, if simple metrics will be used as indicators, their link to fisheries productivity must be clearly defined. Thus the conceptual steps from data collection to fisheries productivity must be transparent and explicit (Shin et al. 2010). And last, ideally indicators are associated with reference points that are explicitly defined (see Table 2 as an example from Link 2005) yet so far setting appropriate warning thresholds or reference points in EBMF has proven difficult (Fulton et al. 2005), and would likely prove to be just as hard under the FPP framework.

Table 2. Ecosystem indicators translated into warning thresholds and limit reference points for EBFM (B: biomass; subscripts ∑, TL3, benth, plank: all surveyed species in the system, all species at trophic level 3, all benthivores, all planktivores, respectively; PP: primary production; $B_{\sum Cons}$*: the sum of biomass consumed by all species in the ecosystem;* S_{min}*: minimum number of species;* L/S_{max}*: maximum mean number of interactions per species;* C_{max}*: maximum numbers of cycles observed:* $N_{scav\text{-}med}$*: median abundance of scavengers;* $V_{jelly\text{-}med}$*: median biovolume of gelatinous zooplankton;* A_{max}*: maximum area of living, hard coral; N/A: not applicable.*

Indicator	**Description**	**Warming threshold**	**Limit reference point**
$\bar{l}$	Mean length, all species	30%	50%
β	Slope size spectrum, all species	N/A	10%
$B_{flatfish}$	B of all flatfish species	$B_{flatfish} > 50\% B_{\sum}$	$B_{flatfish} > 75\% B_{\sum}$
$B_{pelagic}$	B of all pelagic species	$B_{pelagic} > 75\% B_{\sum}$ or $B_{pelagic} < 20\% B_{\sum}$	$B_{pelagic} > 85\% B_{\sum}$ or $B_{pelagic} < 10\% B_{\sum}$
B_{TL4+}	B of all species at trophic level 4 and above	$B_{TL4+} > 25\% B_{TL3}$	$B_{TL4+} > 50\% B_{TL3}$
B_{pisc}	B of all piscivores	N/A	$B_{pisc} > B_{benth} + B_{plank}$
$L_{\sum}$	Landings of target species	$L_{\sum} > 5\%$ PP	$L_{\sum} > 10\%$ PP
$\overline{L/S}$	Mean number of interactions per species	10% below $\overline{L/S}_{max}$	N/A
B_{remov}	Fishery removals of all species (Landings, bycatch, discards, etc.)	N/A	$B_{remov} > B_{ECons}$
S	Species richness (number of species)	$S < S_{min}$, for 3 years	$S < S_{min}$, for 5 years
C	Number of cycles	30% below C_{max}	N/A
N_{scav}	Abundance of scavengers	100% above $N_{scav\text{-}med}$	200% above $N_{scav\text{-}med}$
V_{jelly}	Volume of gelatinous zooplankton	100% above $V_{jelly\text{-}med}$	200% above $V_{jelly\text{-}med}$
A_{coral}	Area of live, hard coral	30% below A_{max}	50% below A_{max}

SYNTHESIS OF INSIGHTS

In this chapter the review has focussed on the use of indicators at the scale of a CRA fishery. As already recognized within the FPP framework, it will be very rare for a project submitted under the *Fisheries Act* to be of significant magnitude that its impact are measured at that scale (e.g. a whole lake destruction from a mine). However, the cumulative effects of small projects (e.g. dock construction) could have that magnitude of impact across a landscape (Minns 2009, Radomski et al. 2010). Further, the insights from these fields of research and management while tailored to larger spatial scales convey some general advice that is applicable to the FPP framework. There are many other sources for gaining insights into potential indicators for the FPP framework from other management programs that are currently searching for similar metrics (e.g. Integrated Coastal Management, Circumpolar Biodiversity Monitoring Program), or from the general field of indicator research that publishes regularly in field specific journals (e.g. *Ecological Indicators*). However, the insights gained so far in this review appear to be common across these many different programs and can be synthesized as five main points:

1. Indicators should be based on sound ecological theory, conservative under uncertain conditions, simple to understand and explain to non-fisheries specialists, and estimated with data that is easy to collect well and analyze.

2. A set of indicators will be needed to relate components of productivity to the “ongoing productivity of CRA fisheries”. Each of them must complement the others but not seek to measure the same ecological function (i.e. no strong correlations). Tailoring indicators to different scales of ecosystem organization is one way of addressing this insight (i.e. community, population and individual level indicators). Another is to select indicators that

are specific to different trophic or socioeconomic levels (i.e. CRA fisheries species, environmentally sensitive organisms or forage-fish level indicators). Finally, an ideal set of indicators would allow for change in one indicator to be better understood within the context of an associated change in another indicator (e.g. decrease in fish size under increasing population density).

3. Sources of error must be well understood, even if not estimated. Sources of error are found in field measurements, the understanding of the ecological process, the effectiveness of the statistical analysis, the final estimation of the change in indicator value, and the implementation of the decision framework. Generally it appears that field estimates of mortality rates are usually quite uncertain, while estimates of biomass or morphological characteristics are better. A strong understanding of the theory behind the use of the indicators should help reduce many of these errors.
4. Confounding or modifying factors such as density dependence, life history evolution, fisheries exploitation and spatial phenomena (e.g. fish migration or the extent of a fish stock) must be considered at the stage of interpreting the change in the indicator, and not necessarily used as an objective for an indicator. Conditions where these four considerations apply should be understood as best they can and spelled out explicitly.
5. Most fisheries reference points are ultimately related to the rate of growth of the fish stock which can be measured as a population growth rate or a stock-recruitment relationship. The data needs for these points typically include the biomass of the catch, the fish sizes at different ages (including when they mature), and both the natural and fishing mortality rates. The challenge with multi-species fisheries is to understand how the different life-history traits among the stocks (e.g. fecundity or individual growth rates) will affect the rate that populations recover from harvesting.

FISHERIES PRODUCTIVITY INDICATORS

FISHERIES MODEL ENDPOINTS

The best and most direct estimates of fisheries productivity come out of a stock which has been greatly overexploited and then monitored during its recovery. Even though there are several more recent stock collapses, the most appropriate example remains the cessation of fishing in the North Sea during World War II. One of the best alternatives is to estimate fishery productivity based on a scientific understanding of how the stock size most likely changes over time. This type of understanding is synthesized as a population mathematical model for which the scientific community has a good understanding of the effect of the model parameters on the model output (see Figure 3). Therefore under the FPP framework, if the indicator can give some reliable insight into how the parameters of a fisheries population models might be changing over time, then it is much easier to predict how the productivity of the fishery is changing. Further, in most cases there would be little need for any mathematical modelling by the proponents themselves because the influence of the particular parameter on productivity would be well known. The challenge at this stage of indicator development is therefore to ensure that there is a strong theoretical link between the indicator and the population model.

As mentioned in the previous section, the most common fisheries models are based upon the logistic population growth curve (Figure 3), and the stock-recruitment model (Figure 4). These models are created with simple assumptions and are used in different forms throughout ecology and management including non-fish related fields. Therefore, if one of these two models can be properly parameterized by one or more indicator metrics (or at least relationships derived from project specific metrics), then the link between the component of productivity and the ongoing productivity of CRA fisheries can be said to have been achieved (even if not explicitly attempted for every project). Both models are also chosen because they have the ability to incorporate density dependence, and because their behaviour is well known, and so could accommodate qualitative indicators as well.

Briefly, logistic growth means that the population growth rate will gradually decrease as the population size reaches its carrying capacity (K). The intrinsic growth rate (r) is the maximum rate that the population will grow at, which will occur when the population is small. Both those parameters make attractive endpoints because strong scientific evidence exists that can:

1. Explain their behaviour,
2. Provide estimates for their typical values,
3. Provide alternative ways to express them,
4. Provide links to other fisheries parameters, and
5. Characterize how they might behave under degrading habitat conditions.

Further, they have a direct link to accepted fisheries reference points such as the maximum sustainable yield. Last, the logistic model provides a compelling endpoint to address three main confounding ecological factors that may not find appropriate indicators:

1. Density dependence: K provides a basic level of density dependence to the model which is that the population growth rate gradually decreases as the population size gets larger. However, if density dependence was suspected to act in other ways, it is not difficult to modify the basic shape of growth curve which provides a greater range of density dependent patterns of population regulation.
2. Life history evolution: r is expected to be influenced by life history evolution yet this field of research is still young. Over time if more explicit relationships are derived between trait evolution and r, these could be incorporated into the model of population growth.
3. Fishing and Trophic Effects: the logistic model allows proponents and managers to add the impact of a fishery harvest, or a predator-prey interaction. While it may not be needed on a regular basis, the ability of the model to integrate the likely strong effect of a commercial harvest into the endpoint of a habitat model is a very powerful benefit. Further, the behaviour of the logistic model within predator-prey or competition (e.g. invasive species) situations is fairly well known and could help provide at least qualitative insight into these confounding factors.

Note that the qualities of the logistic model allows both fisheries managers to easily check assumptions and verify parameters, but also provides flexibility to proponents to tailor the assumptions of the indicators to their project.

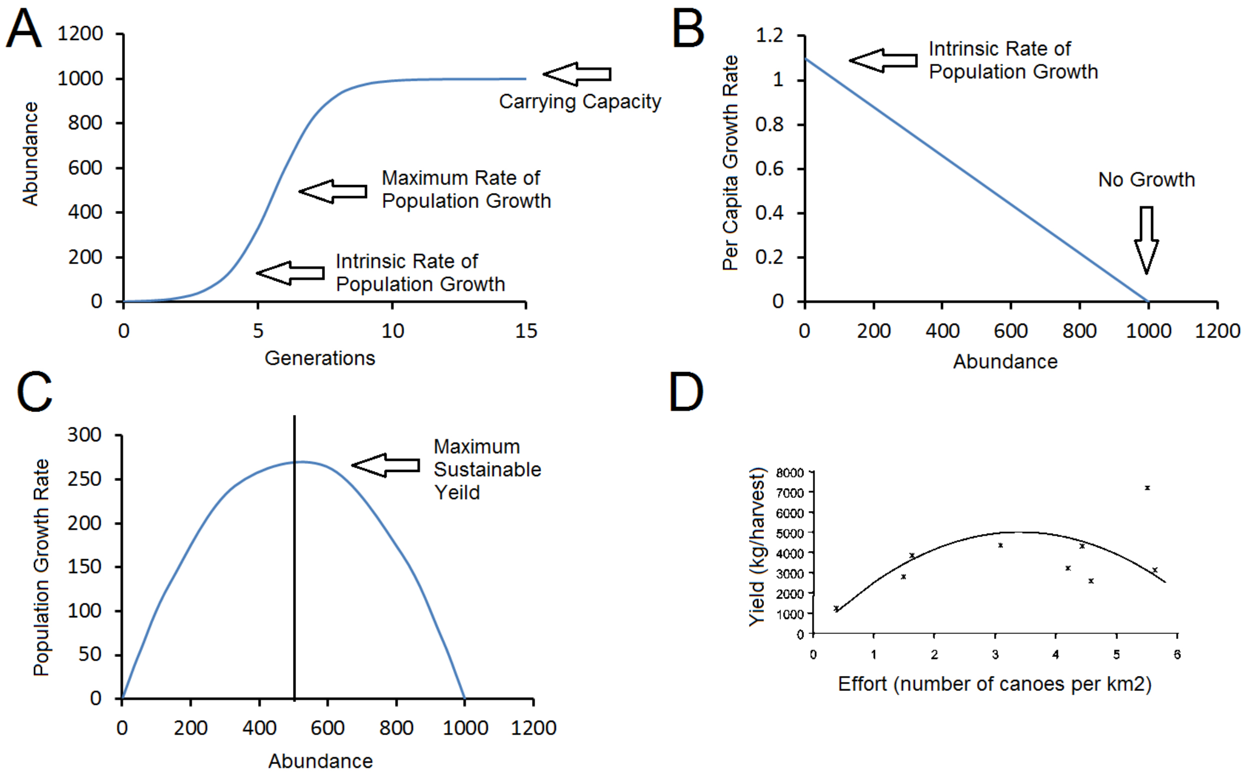


Figure 3. Examples of the logistic growth curve A) logistic growth over time (note how population growth starts small but increases exponentially, but then starts to decrease towards an asymptote), B) the growth rate of the population per individual (per capita growth rate) versus the abundance (note how the greatest growth is at low population sizes), C) population growth rate versus the abundance (note that even through the per capita growth rate is decreasing, the highest population growth rate is at half the carrying capacity), and D) the logistic growth curve as a fishery model, note that high efforts start to result in lower yields because they have exceeded the maximum sustainable yield (from Sparre and Venema 1998).

A stock-recruitment model can also be related to mortality and density dependent estimates. It measures the number of new fish (recruits) that arise from a particular number of spawning fish. While the parameters that drive the model (a and b) are not as intuitive as the logistic growth model, they are both related to density independent (a) and density dependent (b) growth. The many different forms of the stock-recruitment relationship (e.g. Beverton-Holt, Ricker; see Figure 4) provide managers and proponents with some additional flexibility regarding the type of density dependence they expect in their populations. In some ways, stock-recruitment models fall under the logistic growth model's umbrella as K and r can be estimated from it. The slope of the curve at very small population sizes (Figure 4B) is generally assumed to be the maximum growth rate (like r) and the curve reaches an asymptote due to density dependence (and cross the 1:1 replacement line at a carrying capacity threshold; Figure 4A). However the most useful parameter from a fisheries perspective is the steepness coefficient (h). The steepness coefficient is used to estimate maximum sustainable yield reference points as well as to conduct stock projections. Thus h could be particularly helpful for offsetting plans, or for predicting the long term consequences of a project. It can be parameterized by the maximum number of recruits per spawning biomass (this is the slope of the SR curve as spawning biomass approaches 0), the sex-ratio, and the expected surviving biomass per recruit.

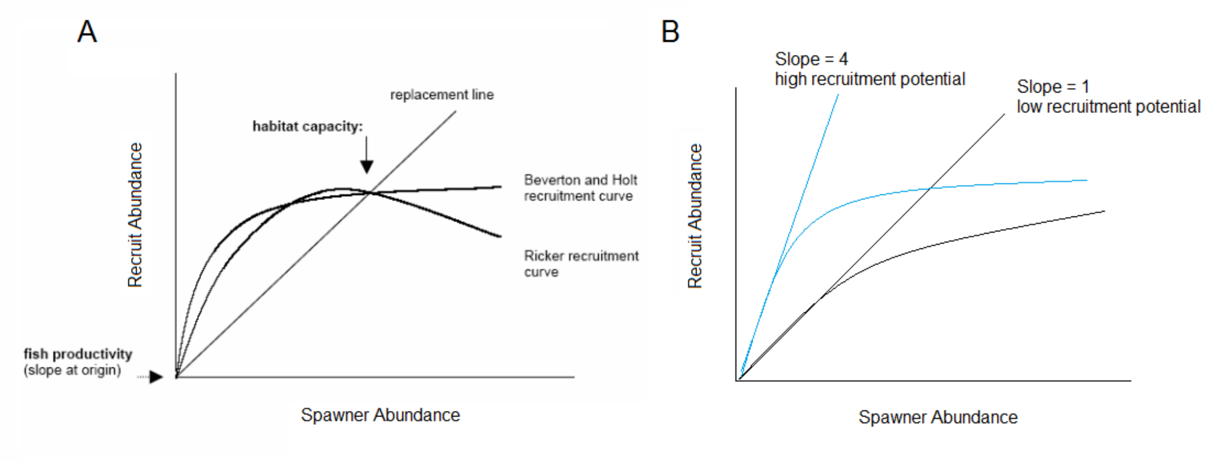


Figure 4. A generalized stock-recruitment relationship showing the Beverton-Holt and Ricker relationship (from Randall 2003), B. Two spawner-recruit relationships demonstrating the growth rate at small population sizes by extending the slope (adapted from Ratz and Lloret 2003).

This review suggests that the K, r and h parameters provide appropriate fisheries endpoints for indicators to aim for (this view is shared by other authors including Meyer 2001). This means that candidate indicators should have a defined theoretical link to one of these values (and not that the indicators are one of these three values). The next section will explore the common theoretical links.

LINKING ENDPOINTS WITH INDICATORS

There are some common models that can link individual or population scale metrics with the fisheries endpoints from the previous section. This section briefly reviews them and suggests candidate indicators (**in bold**).

Production:Biomass and other Allometric Relationship

Production (P) at the level of a fish population can be well defined as the average population biomass (B) multiplied by the rate that the population grows over some time period (usually a year). This relationship is useful to biologists because it means that the ratio between production and biomass (P:B) is proportional to aspect of growth, be it a specific growth rate, or simply the average size of an individual in the population. Figure 5 illustrates this relationship for lake and river fish. The key link with the fisheries endpoints is that the maximum P:B ratio for a fish species is equivalent to the maximum rate of population increase, r. Thus field studies that can identify changes in a population's biomass, or the average size of an individual from that population, can link back to population productivity characteristics through the P:B ratio. Because the P:B ratio changes with fish size it can be estimated using known allometric relationships (e.g. standard physiological and metabolic relationships between characteristics of animal body plans, such as, length versus weight). Therefore a suitable indicator could be the **weight at maturity** of a target fish species. This linkage is useful for data poor situations and could be potentially informative on the effects of life-history evolution yet it is a fairly coarse approximation and can run into uncertainty when considering overall fisheries yield as a few large fish versus many small fish. Further, Randall and Minns (2000) provide **length**, **weight** and **age at maturities** for 79 freshwater species including estimates of the P:B ratios for those species. These offer suitable reference parameters for desktop estimates of the effects of an environmentally induced change in body size on population productivity.

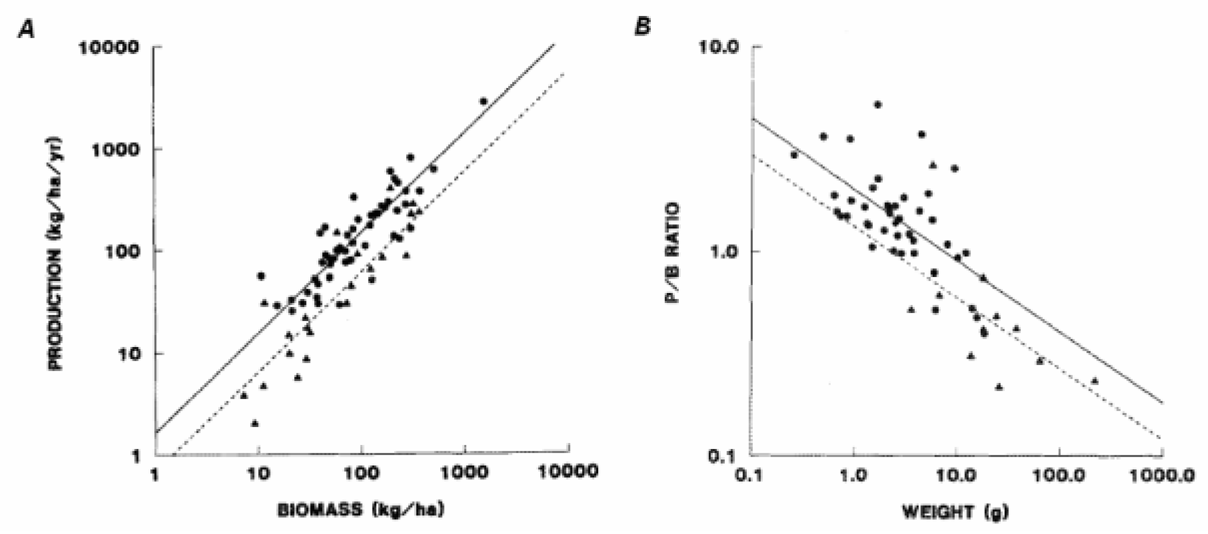


Figure 5. The relationship between A) biomass and production within a particular habitat, as well as B) the average weight of an organism and the production / biomass ratio for lakes (triangles) and rivers (circles) (from Randall et al 1995).

Allometric relationships form the basis of many growth models in ecology. Importantly, the validity of using allometry to predict the effects of changes in individual scale indicators will have on population or fishery productivity is contingent on assuming that certain aspect of a fish's life doesn't change except with size (Economo et al 2005). Further, it depends also on known physiological or metabolic trade-offs that are specific to higher taxonomic groups (i.e. fish, birds and mammals). These trade-offs are expected to be:

1. **age-at-maturity** with **mortality rate**,
2. **reproductive growth** with **mortality rate**, and
3. **mass at birth/hatch** with **mass at maturity** (Charnov 2002 in Economo et al 2005).

These trade-offs demonstrate less that change in one metric will cause an equivalent response in the other because these relationships are built over long time periods, but instead indicate that the effects of some of these metrics have less of a direct link to population productivity using allometric models. In Economo and colleagues (2005) general allometric model, population productivity was estimated by the **mortality rate**, **maximum body weight** and two **metabolic coefficients** that measure the speed that animals grow towards their maximum size. This model gave the same qualitative relationship in the P:B to weight relationship in Figure 5B.

Growth Models and von Bertalanffy

As mentioned in the previous link, the length of time it takes for an animal to reach its maximum size (in relation to its total life span) can be an important determinant of population productivity. In fish populations there is a great range such that many forage fish can reach maximum body sizes within a year or two (Drake et al 2008) whereas longer lived species such as Lake Sturgeon take a proportionately much longer time. Another key link therefore between individual metrics and population growth is found in growth models, and particularly in the von Bertalanffy growth model (VBG; Figure 6). The shape of the VBG model can be estimated for individual growth by looking at the amount they grew each year (although its most often back-calculated through growth rings) or for populations using the range of fish sizes for each of the age classes (e.g. **length** and **age frequency distributions**). The great use of determining the curve of the relationship is that its explicit parameters have biological meaning. Many studies have sought to re-interpret the parameters of the original VBG equation (e.g. Lester et al 2004) because at the time of its discovery (1930s), von Bertalanffy described it as a pattern rather than a process. Still, the curve can estimate:

1. The average **maximum length of individuals**,
2. The length at age 0, and
3. A coefficient of growth (k).

Further these paramaters can be reinterpreted using a **gonad to body weight** ratio, **age-at-maturity** and **body growth rate** (all metrics that can be measured in wild populations). The coefficient of growth (k) is defined after an individual has matured by how much growth the animal invests in reproduction verses body size (Lester et al 2004). This parameter has a great deal of significance in fisheries because it is often used to estimate the rate of **natural mortality (M)** (Charnov 2008). Further, the M/k can be related all the way back to the maximum sustainable yield (through the optimal cohort biomass defined by Beverton 1992 and Holt 1958 in Frisk and colleagues 2001) which as mentioned above is a reasonable fisheries endpoint for ongoing productivity. Therefore, metrics that allow for statistical estimates of VBG curve or parameterization of the VBG model, would be suitable indicators. Last, because it has been used so ubiquitously in ecology there are modifications of the VBG model that allow habitat changes to directly influence the shape of the curve (e.g. temperature effects on bullhead growth in Kielbassa et al 2010).

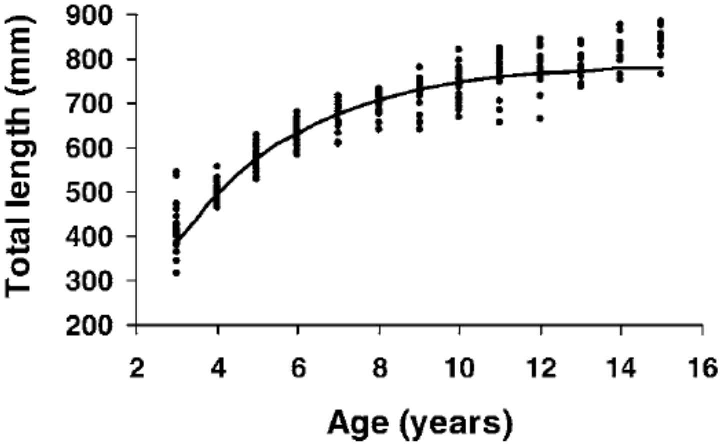


Figure 6. A von Bertalanffy growth curve of a Lake Trout population from Lake Huron. (from He and Bence 2007).

Size Spectrum Models

The allometric relationship of population Production:Biomass to individual body size (Figure 5B) can be extended into including trophic levels. Classifying predators and prey into “levels” may be more related to what was proposed first in the history of ecological theory than ecological reality. More recent models of predator diets, trophic linkages and stable isotope studies show a tangled network of energy flowing between and across individuals, than any hierarchical levels. For example, a Lake Trout egg is eaten by the prey of the Lake Trout adults. While trophic divisions are still useful for many reasons, in terms of understanding community productivity, seeing organisms as simply different sized bodies might be more useful. Size spectrum models are based on this idea and the following observations:

1. There is a negative relationship between body size and individual metabolism;
2. The variability in the prior relationship is supplied by the effect of trophic interactions.

Understanding how environmental change will affect both these observations will allow managers to separate short terms changes in **species abundance** due to predator-prey interactions, and long term changes in **system productivity**. In Figure 7, there are regression lines for the range of body sizes between phytoplankton to fish in different environments (Boudreau and Dickie 1992). Note how the slopes are all similar but the lines are not overlapping (i.e. some environments have higher biomasses for the body sizes of the species). This demonstrates the effect of nutrient availability on **primary production.** Because all the lines are parallel, this type of relationship suggests that knowledge of any **biomass per the body size** of one of the size classes along this continuum should allow a manager to predict the overall productivity of the ecosystem. While this is a compelling hypothesis the loss of energy when predators convert prey biomass into growth (i.e. the mean efficiency of the system) can affect this relationship, so other considerations need to be taken. But generally, the insight that **mid-trophic level biomass** can inform upon the state of top predator biomass (and system production) is in some agreement with predator-prey theory such that reductions in CRA fisheries productivity would release predation on forage fish which would then increase in abundance.

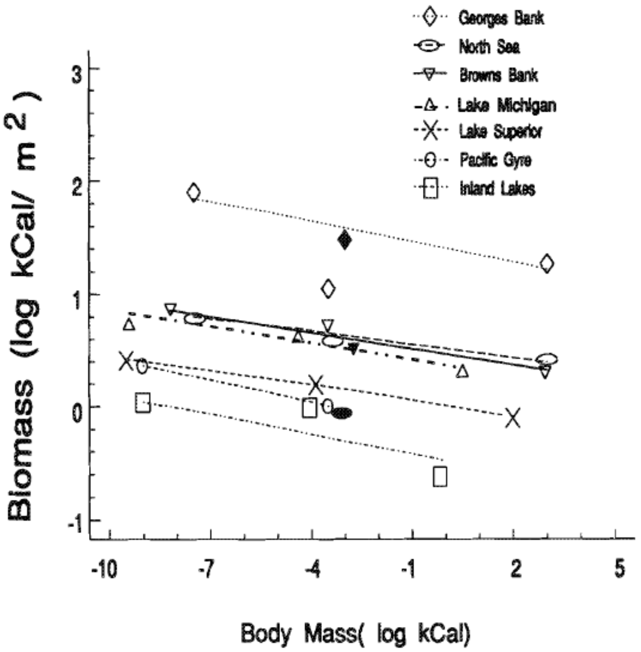


Figure 7. Regression lines for the range of body sizes between phytoplankton to fish in different environments (Boudreau and Dickie 1992).

The Beverton-Holt stock-recruitment model can be coupled with the von Bertalanffy growth curve to create a framework for conducting stock assessments. As mentioned earlier for this model to work a manager must have fairly good estimates of fish **growth**, **mortality**, **maturation** and **recruitment**, as well as good aging data from the stock. However certain parameters can be estimated through the VBG (e.g. M/k from the previous section) which leaves only two main parameters to measure the **average maximum length in the population** (L_∞), and the **density independent survival** coefficient of the Beverton-holt stock recruitment model (α). Anderson and Beyer (2013) further simplified this relationship with size-spectrum theory down to the **average maximum size in the population** (W_∞).

The size spectrum literature is full of allometric and metabolic relationships between size and different physiological traits and behaviours of organism including optimizing life history strategies (Maury and Poggiale 2013), **consumption intake** (Pawar et al 2012), **predator:prey biomass ratios** (Jennings and Mackinson 2003) and density vs encounter rates (Thygesen et al 2005). Indicators addressed here include **metabolic rates**, **abundance and biomass for specific trophic levels, and average body sizes**. A commonly derived allometric relationship is the **number of eggs per body size** (Figure 8). While these relationships provide good references, it should be pointed out that they are still coarse approximations so comparisons of observed fecundities at a site (e.g. downstream from pulp and paper mill) to the derived relationships may not be quantitatively informative. However, the relationships have qualitative value.

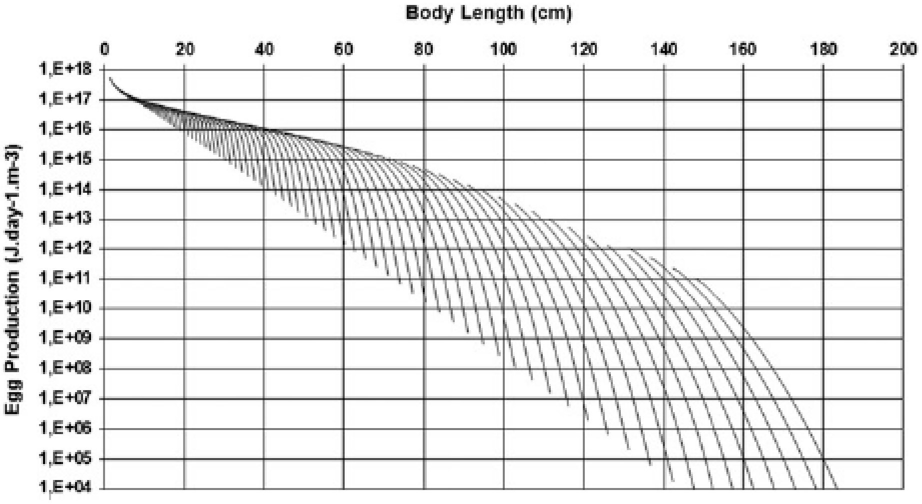


Figure 8. Relationship of body size to the number of eggs produced per individuals (from Maury and Poggiale 2013).

Stage Structured / Demographic Models

While the previous links have all been based on allometric relationships and biomass, demographic models primarily use **mortality rates** and **fecundity** to estimate the **abundance of individuals** (Figure 9). The types of models are typically stage structured which means that the probability of surviving from one life stage (or age) to the next determines the growth or decline of the population. The effect on population growth can be estimated numerically (i.e. by running the model and finding the equilibrium where the population size stops changing – in this case the biomass of the annual deaths would equal the population productivity), or directly following the derivations of McAllister and colleagues (2001). There is a wide range of applications for stage-structured models including:

1. Estimate the functional dependence of population growth rates (lambda) on stage specific **mortality rates**, **individual growth rates** and **fecundity.** Using this method Velez-Espino and Koops (2009) found that Black Redhorse in the Great Lakes were most sensitive to **young of year mortality rates** related to habitat loss.
2. The Area-Per-Individual model incorporates allometric density relationships with **body size** to estimate the minimum area an egg, young-of-year or adult individual requires.

These areas can be linked with their mortality rates much like in Velez-Espino and Koops (2009) model. Using this model Minns (2003) estimated that **adult foraging habitat** was much more valuable to maintaining population sizes than spawning or young-of-year habitat in Lake Trout and Sculpin.

3. Gutjahr-Gobell and colleagues (2006) used an age-structured model to relate their laboratory results of the impact of endocrine-disrupting chemicals on decreased **fecundity** in the estuarine Cunner to their population growth rate. They found that a 38.7% decrease in egg production resulted in an estimated 10% decrease in population growth rate which would lead to a 65% decrease in abundance over 10 years of exposure.
4. The Euler-Lotka stage-structured model was derived to estimate population growth rate in Leopard Sharks using **female maturity**, **maximum reproductive age** and **average fecundity**. Density dependence was included by incorporating the stock-recruitment relationship derived from fisheries models. The results were verified by field studies and demonstrated that sharks were more susceptible to overexploitation than previously thought.

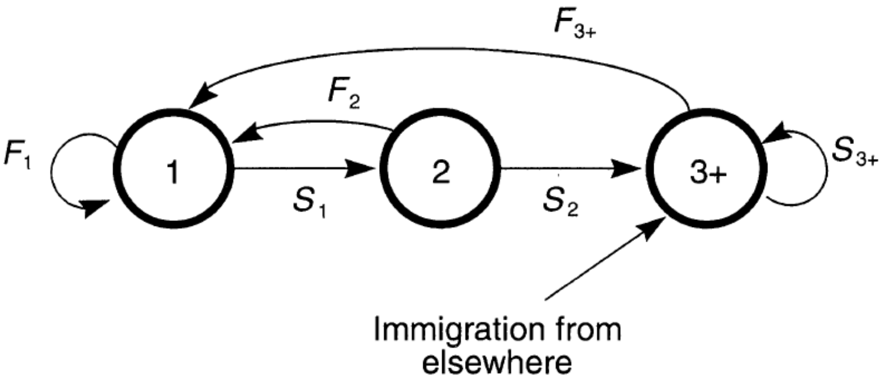


Figure 9. A general stage-structured population model which demonstrates 3 stages of life and the survival (S) and fecundities (F) associated with each stage (from Hitchcock and Gratto-Trevor 1997). Note that in this model only stage 3 adults migrate, but all life stages reproduce.

Individual Based Bioenergetic Models

Individual based bioenergetics models was left as the last example of a link between an indicator and the fisheries endpoints because they are so flexible that given enough processing power and known metabolic, physiological or behavioural relationships virtually any metric could be linked to population growth rate. The value in this is not that all indicators can be considered useful, but that the relative sensitivity of population growth to variation in one metric over others can be tested. For example, some of the previous models suggested that size-based relationships between metabolism, growth and mortality explain most of the variation across communities of organisms. While any indicator could be incorporated into the model if a relationship existed (e.g. fin thickness), their effect on ultimate population growth rates under the reasonable range of parameters observed in the wild may be practically unnoticeable.

The underlying structure of a bioenergetics model is simple: the net energy available for tissue or reproductive growth is equal to the energy gained through consumption minus the energy lost through maintaining the functioning of the body and acquiring food (Figure 10). Bioenergetic models can be as complex or involve as many individuals as the designer wants up to the processing limits of the computer. They generally can be memory intensive because they need to keep track of every interaction of every individual as they proceed through their lives. However, depending on the question there are ways to simplify the models, and commercially available software (Fish Bioenergetics 3.0; Hartman and Kitchell 2008).

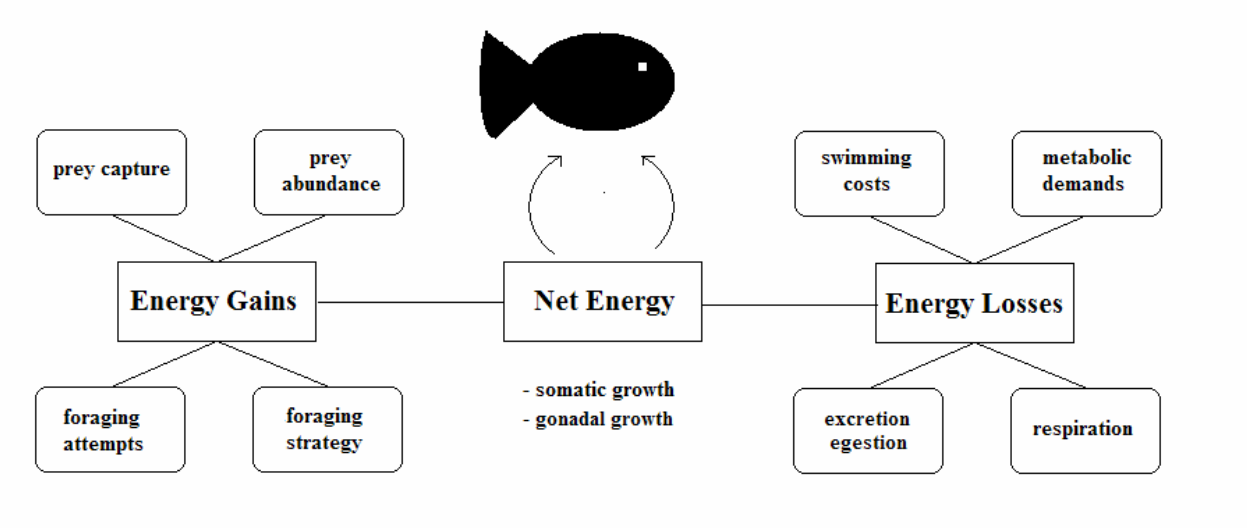


Figure 10. A general schematic of bioenergetics modelling (from de Kerckhove et al 2008).

Translating the individual based nature of most bioenergetics models to populations is relatively simple. Since the growth, reproductive success and death of every individual is track, it is relatively easy to obtain direct estimates of the generation of biomass over time. This allows the modeller to apply certain restrictions on vital rates (e.g. apply the DFO Pathway of Effects directly to a component of productivity) and observe the change in population productivity. Further, reference points can be easily determined this way by finding the thresholds where populations start to decline. There are many examples of using bioenergetics models to generate population reference points in fish (VanWinkle et al. 1993, Steele 2012) however a constant challenge is justify the applicability of the model parameters. Because bioenergetics models are individual specific, the choice of parameters to describe the individual also restricts the application of the model to other species (or even populations) without re-assessing the validity of the parameters and metabolic relationships. For this reason, bioenergetics models are appropriate for estimating the effect of environmental change in specific conditions, that can be field verified afterwards (VanWinkle et al 1993).

Limitations notwithstanding, currently published bioenergetics models can be very insightful towards selecting indicators and probable are most useful when comparing the relative effects of different metrics (Steele 2012). Pethybridge and colleagues (2012) demonstrates in anchovy a 15% decrease in r due to either a 5% drop in **temperature**, a 18% drop in **food abundance** or a 30% increase in **egg mortality rates** which demonstrates that variation in temperatures could mask declines in the other two indicators. Rinke and Petzold (2003) demonstrated how to tease apart these two main effects in zooplankton showing that **temperature** mainly influenced **early life growth**, and **food abundance** influenced **maximum size** and **fecundity** (Rinke and Petzold 2003). In a community setting Giacomini and colleagues (2013) demonstrated that **body growth rates** were critical for defining community stability, followed by **egg size** and **maximum body size**. However, bioenergetics models should still be used as a guide rather than a solution to choosing indicators. Although the relative influence of certain parameters can seem obvious in the modelling scenarios, their behaviour under natural conditions might be very different if key parameters were left out of the model (VanWinkle et al 1993; Pethybridge et al 2012).

HIERARCHICAL PRODUCTIVITY INDICATORS

Potential productivity indicators have been proposed within the FPP framework as components of productivity (see Appendix Table 3 in Bradford et al 2013), and this review has uncovered more from established fisheries indicators and the linkage models to fisheries endpoints. Each of the proposed or uncovered indicators will be reviewed here under the component of productivity they belong to (e.g. size based indicators are included under Growth). Please note that excellent reviews of indicators can be found in the EBFM literature including Rochet and Trenkel (2003), Fulton and colleagues (2004b), Fulton and colleagues (2005) and Shin and colleagues (2005), as well as within general environmental, ecological and fisheries based research including Patrick and colleagues (2010), Jorgensen and colleagues (2013) and Perez-Dominguez et al (2012). This review incorporated the advice from existing reviews while updating the current knowledge base with more recent publications and un-reviewed case studies. Indicators are discussed regarding:

1. The ease that the metric can be measured
2. Their benefits and limitations
3. Their redundancy against other promising indicators
4. Their potential value as qualitative indicators, and
5. Potential reference points and warning thresholds.

Component of Productivity: Growth

Abundance or Biomass

Abundance and Biomass are the most common indicators used for fisheries management, stock assessment, conservation and ecosystem based monitoring (i.e. 80% of indicators in Fulton and colleagues 2004b). The assumption that populations will grow (or shrink) in response to improving (or degrading) habitat condition is intuitive and provides a direct qualitative link to fisheries productivity. Both abundance and biomass can be used to estimate population growth rates (r) either through Production:Biomass Relationships (Randall et al 1995) or through time series analyses of population recoveries following disturbances (Franco et al. 2010, Lambert 2011). The metric is relatively easy to measure and is included in almost any type of fisheries survey. The many examples of abundance and biomass falling under degrading habitat conditions or overexploitation (e.g. Ripley et al 2005) led the International Union for the Conservation of Nature Red List Criteria to define conservation listing based on 50%, 70% or 90% decreases in abundance or biomass over 10 years or 3 generations (Maxwell and Jennings 2005). These reference points could be adopted for qualitative estimates of important thresholds to population productivity. Further, Fulton and colleagues (2004b) recommended Juvenile Abundance (as did Gilliers et al 2006), Stock Size of Species at Risk and Spawning Biomass as particularly robust metrics for which historical values within some watersheds could provide suitable reference points. However, the influence of density dependence, environmental fluctuations and indirect trophic effects can often obscure or lead to unpredictable responses to habitat change or fisheries harvests (Abrams 2002, Fulton et al 2004b, Matsuda and Abrams 2005, Maxwell and Jennings 2005). This literature review found the following main concerns with this potential indicator:

1. It may not track changes in habitat degradation at all or equally across species:

  - 50% decrease in cover resulted in no change in abundance in stream fish – Ellis and Bell 2013
  - industrial development in Alberta negatively affected bull trout, whitefish and rainbow trout abundances, but not arctic grayling across >100 sites – Scrimgeour et al. 2008
  - no detection of changes from metal contamination even though a strong response was observed in community based indices (e.g. the Index of Biotic Integrity) – Bervoets et al. 2005
  - lake acidification led to immediate declines in abundance in forage fish but took longer to influence both abundance and biomass in rainbow trout and white sucker indicating that they responded more to lost prey than the habitat change– Mills et al 1987

2. The ability to detect trends in the indicator under reasonable monitoring program may be severely constrained by its typically high natural variability. Five studies found that even large population declines (25% decrease) would generally take over ten years of annual sampling to be detected (Wilson et al. 1999, Hatch 2003, Maxwell and Jennings 2005, Budy et al 2007, Al-Chockhachy et al. 2009). This is a common concern with abundance, biomass and diversity based estimates (Peterman 1990, Fulton et al 2004b, Jones and Petreman 2012).

Both these concerns suggest that abundance and biomass are suitable indicators as long as they include a range of species, appropriately selected species or are conducted over long time periods (Fulton et al 2004, Jones and Petreman 2012). Further, the use of reference sites would greatly aid the selection of reference thresholds, as well as differentiate between natural and project induced change (Jones et al 2008).

Body Size (Length or Weight)

Body size metrics are also common indicators for fisheries management (i.e. 30% of indicators in Fulton and colleagues 2004b) and are used to:

1. Calculate total biomass and fish condition,
2. Parameterize the von Bertalanffy growth curve,
3. Estimate important life history traits such as size at maturity (Woodward et al 2005), and
4. Determine the vulnerability of certain species to overexploitation (Patrick et al 2010, Thorson et al 2012).

Further, because fisheries harvests are often size selective managers are usually interested in knowing if the size distribution of the stock is changing over time. Changes in size distributions within populations are typically in response to harvesting of large individuals (Shin et al 2005) or gradual reductions in growth rates (McPhie and Campana 2009) and so can occur over the course of several years (Mills et al 1987). However, short term changes in size have been found in some fish populations following disturbances including juveniles shrinking in length by over 10% in the winter (Huusko et al 2011). This last point is surprising in that it is the length that shrunk rather than the weight. Because length is determined by harder structures it typically doesn't vary as much as weight (i.e. variation was 80% lower in length than weight in Faller et al 2003). Fulton and colleagues (2004b) generally found weight or length based estimates to be moderately robust and along with other studies (Iserman 2007, Drake et al 2008, Matsumura et al 2011) recommended that thresholds include age specific sizes (e.g. size at maturity and juvenile sizes were both considered robust indicators across a wide range of studies). Like biomass and abundance estimates, length and weight based indicators can be influenced by natural variability and density dependence leading to unpredictable behaviour, and so typical monitoring time periods (i.e. < 5 years) may be too short to detect significant changes in the metrics (Allen and Pine 2000, Iserman 2007). Shin and colleagues (2005) recommends that any size-based indicator be used as part of a suite of indicators, as well as offers recommendations for complementary sets.

Body Condition

Body condition (often measured as the ratio of weight to length) is another common fisheries metric, however it is becoming less used because of concerns that it can give misleading results (Fulton et al 2004b). On theoretical grounds it is much more valuable to choose one of the two metrics within the ratio (e.g. length or weight) as the indicator, and to base this choice on which one is more likely to be affected by environmental change and less so by natural or seasonal variability (Hering et al 2006, Jia and Chen 2013). Otherwise there is a danger that the ratio will demonstrate a great change in metric value only because of a small absolute but large relative change between length and weight, which will not be indicative of a great change in habitat quality. This perhaps explains why half of the 11 field studies examined in this review demonstrate poor or inconsistent results when evaluating condition factor and why condition factor is often a very successful indicator in experimental or laboratory studies where one of the two measurements (usually length) is controlled (Wenger et al 2012). However, the great benefit of condition factor is that it is easy to measure and interpret while lending itself well to comparisons between populations. It is worthwhile to note that condition factor appears to be a more relevant index when comparing populations of one species among a few sites that were surveyed at the same time of year and within the year. This suggests that as an indicator it is more useful for assessments that include reference sites, rather than as a time series at one location. Further, standard condition factor values can be found in the literature or online databases for many Canadian fish species (e.g. FishBase). Declining condition factors generally indicate a state of nutritional deprivation or higher frequencies of disease (Mills and Chalanchuk 1986, Budy et al 2007, Uusi-Heikkila et al 2010). However, note that if an impact generally decreased growth rate for developing fish, this could influence weight and length in equal proportions and so not result in any change in condition factor. Temperature appears to be highly correlated with condition factor and should be included when using this metric (Jones et al 2003, Jones and Tonn 2004, Lambert 2011). Finally, notwithstanding the limitations of using body condition as a metric to track environmental change, the interpretation of body condition remains a worthwhile pursuit such that heavier fish (for their length) have a greater reserve of energy to contribute to other life history traits (i.e. reproduction) or to fight periods of disease or starvation (Koops et al. 2004).

Trophic Level (Stomach Contents)

A change in Trophic Level (and Abundance-based indicators that group species by Trophic Levels) is considered a highly robust indicator (Fulton et al 2004b, Patrick et al 2010, Perez-Dominguez et al 2012) however it generally only applies to impacts that are expected to either impede predator access to, or completely eliminate the presence of, a particular source of prey, and is therefore more likely to be applicable under large, ecosystem level developments. For this reason, as well as the generally high effort required to properly characterize a species' trophic level, this indicator is not commonly used for the assessment of habitat change but is found much more often in characterizing the risk of fisheries to overexploitation (Patrick et al 2010, Perez-Dominguez et al 2012). The metric is most useful if it can be compared to a pre-development value, however, any decrease should be considered negative and any substantial changes in trophic position (i.e. > 1) should signal the need for further investigation (Fulton et al 2004b). A great benefit of using Trophic Level as an indicator is that the foraging habits of Canadian fish are generally well known (Scott and Crossman 1977, Coker et al 2001) and thus at least a qualitative reference point is typically available. However, while any change in Trophic Level has intuitive ecological ramifications, the link between foraging level and the sustainability of population productivity is less clear and likely species specific. For some species (e.g. Lake Trout) changes in prey species confer great changes in body size, but not necessarily in population abundance (Matuszek et al 1990). Further, recent work using stable isotopes and genetic barcoding suggest that Trophic Levels may not always be static functions and are instead based on seasonal availability of prey within the year and predator-prey cycles over multiple years.

Growth Rate

Growth Rate (and particularly age specific growth rates) appears to be a robust indicator of environmental impacts and fishery induced change (Fulton et al 2004b). A wide range of environmental impacts (e.g. metal contamination, temperature change, increased sedimentation) can acutely target growth rates leading to strong correlations between the indicator and the state of the environment (Gilliers et al 2006, Liao et al 2006, Wenger et al 2012). However, it is important to determine beforehand how acute the relationship between the environmental impact and growth is expected to be. For example, growth rates decreased for Lake Trout but increased for White Sucker under experimental lake acidification (Mills et al 1987) and so the effects may not be broadly felt across species. Further, like condition factor, growth rates will be strongly influenced by temperature which should be included as a covariate (Searcy et al 2007) and limits comparisons across large latitudinal differences (Conover et al. 2005). A great benefit of measuring growth rates is that the variability of the estimate is often lower than many other size-based indicators (e.g. a coefficient of variation of 20% in Budy et al 2007 versus 50% for the Gonado-Somatic Index and 60% for weight in Faller et al 2003) because:

1. They are typically estimated and compared across longer time periods (> 5 yrs) which removes seasonal effects and time lags (Drake et al 2008), and
2. They are associated with high selection differentials thus the signal to noise ratio should be fairly high (Matsumura et al 2011 although see Conover et al 2005).

Age-specific growth rates contribute an additional level of certainty because juvenile growth rates are expected to be more sensitive to environmental quality than adults whose growth is often hidden as an investment in reproduction (Lester et al 2005). More information on juvenile growth (and mortality) is often helpful in disentangling potential confounding factors. For example, under environmental impacts that increase juvenile mortality, the growth rates of the remaining fish could very well increase and thus provide a misleading indicator of habitat quality. This is why size-dependent mortality in juveniles have been demonstrate to be an important indicator in marine systems (Conover et al 2005). Individual growth rates are intuitively linked back to population productivity and therefore to the intrinsic rate of population growth (Denney et al 2002). However, in this respect understanding the trades-offs between juveniles and adults is also important. To continue our previous example, if reduced habitat quality also led to cannibalism of juveniles by adults, improved adult growth rates would be observed but at the expense of decreased population growth (Botsford 2010).

Component of Productivity: ***Survival***

Mortality Rates

Mortality is a very common indicator in fisheries management because it forms the basis of most harvest models in the forms of Total (Z), Natural (M) and Fishing (F) mortalities. In Fulton and her colleagues (2004b) assessment of EBFM indicators, mortality was a top indicator (with reference points) in the form of Adult Survivorship (no decline), the ratio of Fishing to Total Mortality (> 0.5 and > 0.8 for groundfish), and the Total Mortality of Species at Risk (species specific thresholds). While these examples examine the mortality imposed by the fishery, anthropogenic mortality (A) can be included as an additional factor such that

$Z = M + A + F$

However, sources of mortality are not expected to be completely independent from each other, and so interactions (denoted as X:Y) among the three of them are expected such that

$Z = M + A + F + M{:}A + A{:}F + F{:}M + M{:}A{:}F$

This illustrates how the addition of another source of mortality (A) can greatly complicate intuitive relationships between environmental change and Z. Peron (2013) offers tools to measure the five categories of pairwise interactions between A and M, which include:

1. Over-compensation,
2. Complete compensation,
3. Partial compensation,
4. Complete additivity, and
5. Overadditivity.

However, it isn’t clear whether the same methods would be possible with the addition of F (and thus 3 sources of mortality). Only “complete additivity” represents an independent relationship between sources of mortality, however it is likely not common in situations that involve density dependent population regulation. However, notwithstanding these limitations, an increase in Z should still offer a strong indication that at least one of these three sources of mortality has also increased, and could provide a warning for the need for additional studies. An intuitive way to reduce the confounding effects of F would be to measure rates of mortality for age classes before they are recruited to the CRA fishery. A few studies have found that egg and juvenile mortality rates are more robustly correlated with environmental change than rates in adult age classes (lake herring - Pangel et al 2004, smallmouth bass - Dunlop et al 2007, salmon - Pulkinen and Mantyniemi 2013).

A benefit of measuring mortality is that it provides one of the key parameters for creating Leslie Matrix-Models and so has a direct link to the intrinsic rate of population growth (although see Gedamke et al 2007 for common misconceptions about estimating *r*). A long debated topic in fisheries management has been whether M can be estimated while a stock is being fished (see Lee et al 2012 for a recent debate), however, this may not be as much as a problem under the FPP. If appropriate reference sites were chosen M would be assumed to be equivalent across sites, and so only A would vary and contribute to the observed differences in Z. Common methods for determining mortality involves mark-recapture methods or surveys of population densities over time. While mark-recaptures methods can often provide the best estimates for disentangling multiple sources of mortality, even small levels of post-release handling mortality could cause large biases in estimated rates (Arnason and Mills 1987). Last, the effect of density dependence on mortality has been explored under a wide range of scenarios including under highly variable ecological dynamics (Minto et al. 2008). Similar to size-related indicators, ambient temperature contribute a strong influence to mortality rates and should be measured alongside with any mortality-related indicator (e.g. overwintering mortality in Murphy et al. 2012 and larval survival in Landsman et al. 2011).

Component of Productivity: Individual Performance

Stress response (Blood Chemistry)

A stress to an organism is defined as a measurable biological response of an individual to an external stimulus or stressor (Selye 1976). The response of individuals to a wide range of contaminants and other physiological or metabolic stresses has been generally well characterized in laboratory settings, however it remains a great challenge to translate individual responses to population level responses (Hellawell 1988, Adams et al. 1996, Adams 2002, Nadzialek et al. 2011, Segner 2011). For example, the presence of PCBs can have large effects on fish growth in the laboratory but is frequently not identified as a contributing factor to population declines in field studies (Adams et al. 1996). Traditionally, attempts to scale individual responses to populations involved applying mean response sample statistics to many individuals but this descriptive approach didn't meet with much success. Density dependence, compensatory growth, fecundity changes and natural variability can all hide the effects of a single stressor (see Figure 11, Adams 2002). Even in large population collapses where pollution is suspected to be a contributing factor it can be very difficult to conclusively identify the cause (Barnthouse et al 1990) and consequently there are few studies that attempt to explicitly measure stressor effects systematically or translate changes at lower levels of biological organizations to populations (Barnthouse 1993, Power 2002). Determining relationships between environmental stressors and organism level responses in the wild is difficult enough because:

1. There are often long time-lags between the exposure and the response,
2. There are often multiple complex mechanisms that lead to a response, and
3. Other ecological factors may obscure or overwhelm responses (Adams et al 1996).

Moving research in aquatic toxicology towards more explanatory approaches could help identify toxicological pathways in greater detail and therefore allow researchers to determine whether other stressors in the wild would enhance or obscure the responses (Segner 2011). Further, a transition from single stressor to multiple stressor studies would be of great value for isolating important indicators (Segner 2011).

An alternative approach is to ignore individual responses and instead measure population level responses to stressors directly using three main techniques: "compare-and-contrast", "graded-exposure-response" and "sequential sampling" (Powers 2002). The "compare-and-contrast" method samples population level characteristics (e.g. fecundity or condition) from areas with and without the presence of the stressor. This approach is intuitive yet is limited by the same ecological processes that obscure scaling up individual responses (van den Heuvel et al 1999). Further, these types of studies typically involve one moment in time, and so the static nature of the data does not fit a toxicological response which typically occurs over a long time period. The "graded-exposure-response" method samples populations over a gradient of stressor concentrations (Powers 2002) and the "sequential sampling" method over a longer time period preceding and following a single stressor exposure event (Mills and Chalanchuk 1987, Mills et al 2000). The limitation of any of these methods is that they remain primarily descriptive of one particular situation and thus do not provide any meaningful predictive power for other environments or species.

It is thus very difficult to choose an appropriate indicator that is specific to blood chemistry and not covered by another class (ie. fecundity or growth). Even general blood indicators of contamination (e.g. 7-ethoxyresofurin O-deethylase) can demonstrate strong relationships with growth (e.g. $r^2 = 0.55$ with age-3 growth in Adams et al 1996) which do not translate to field studies. Last, condition indices typically associated with pollutants (e.g. gonado-somatic, spleno-somatic, or histo-somatic indices) have shown some success under very strong environmental stressors but typically exhibit high coefficients of variation over time and high variation among species.

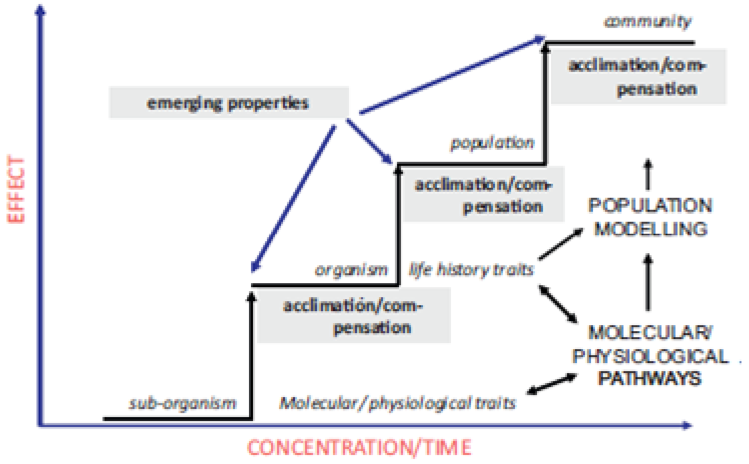


Figure 11. Determining the effect of a concentration of a stressor from organism to population responses (Adams 2011).

Disease (Histopathology)

Disease is considered a robust indicator of ecosystem dysfunction (Rapport 1999, Vethaak et al. 2009, Blazer et al 2010). The presence of skin lesions or abnormal histopathological characteristics (see Figure 12) is often associated with degraded water quality and environmental contaminants (Vethaak 1991). However, it is often very difficult to identify the specific pathogen, environmental stressor or toxicant that is causing the disease responses because external abnormalities are usually a cumulative effect of opportunistic pathogens, environmental conditions and a combination of chemicals. Further, population level responses to disease are highly variable, and so like general physiological stressors it is difficult to predict how the identification of a disease will affect population productivity. For example, viral pathogens infecting centrarchid populations in the United States caused mortalities in only a few of the infected populations while the others exhibited no decreases in productivity (Goldberg 2002). Mortality rates from bacterial infections appear to be better understood and in some cases relationships between environmental conditions can be demonstrated. For example, increased nutrient concentrations due to eutrophication as well as higher water temperatures can turn a range of benign bacterial communities into virulent strains (Wedekind et al 2010), and similarly, harmful algal blooms containing cyanobacteria can produce enough ammonia to cause large fish kills of Catostomids from *Flavobacterium columnare*, even though the bacteria is commonly found in the system (Blazer et al 2010). Therefore, disease appears to be an excellent warning indicator that a subset of other indicators should be investigated. For example, if disease rates began to increase following a development, this might signal the need for initiating surveys on mortality or fecundity rates. However, ultimately these other indicators would provide more conclusive links back to population productivity. Further, disease indicators might be more useful as a factor which is applied to other indicators (e.g. along with density dependence and fishing effects). Last, biochemical markers identified in immunotoxicology studies can be associated with specific changes to fish habitat yet it requires a level of effort often beyond the scope of typical fisheries assessments (Segner et al 2012). However, a great benefit of including disease in a suite of indicators is that abnormal external characteristics are generally easily identified (see Fig 12) and the public can intuitively accept disease as an impact worth mitigating.

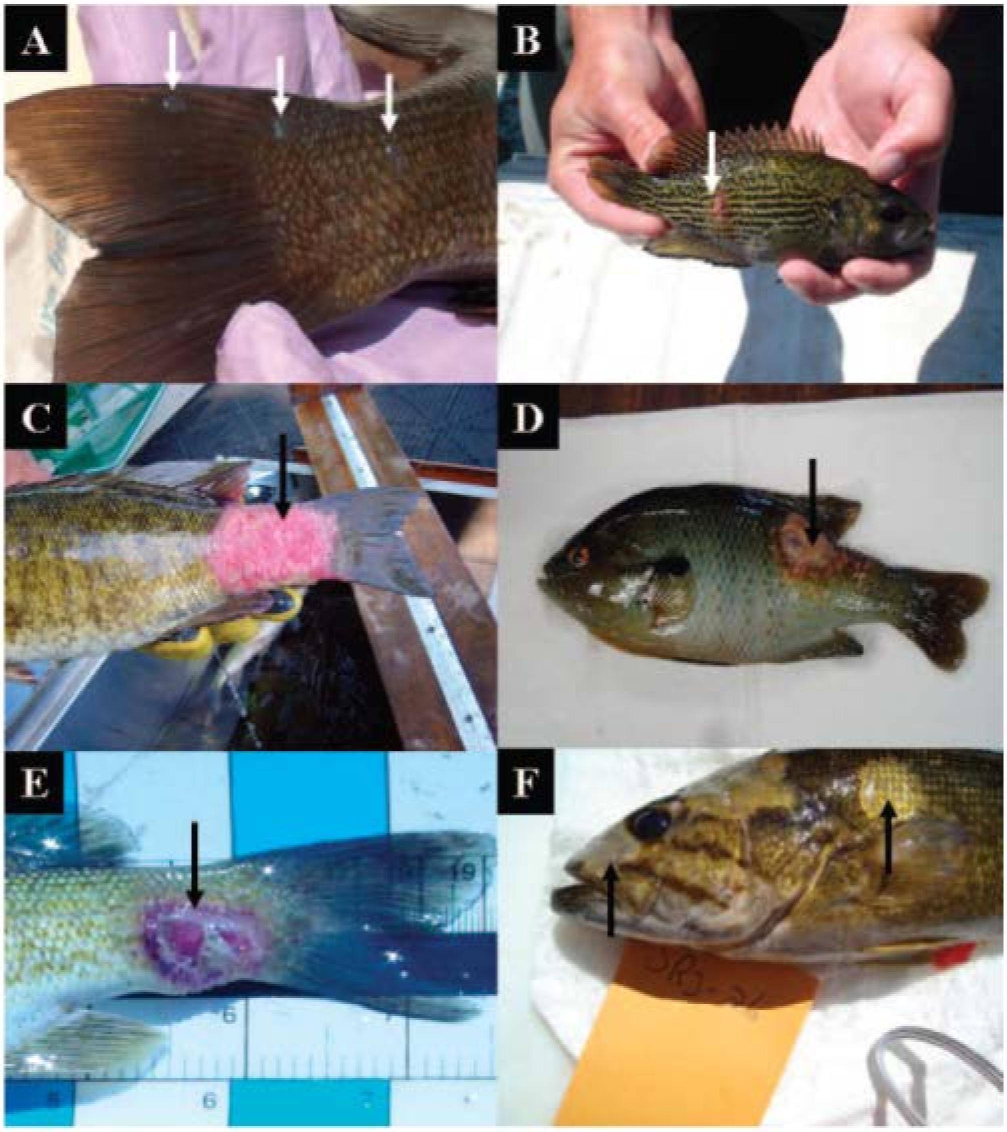


Figure 12. Examples of disease and external lesions on centrarchids in the United States afflicted with bacterial and fungal infections. (from Blazer et al 2010).

Component of Productivity: Migration

Movement

Although the movement of organisms provides a fundamental assumption in nearly all ecological models (*note* if not explicitly assumed, it is almost always implicit), until very recently (i.e. < 15 years) it was difficult to measure and rarely applied beyond simple representations of random or deterministic movement. However, with the advent of smaller tracking technologies (e.g. telemetry, GPS) and higher computer processing power for image based surveys (e.g. videography, hydroacoustics), the ability to characterize fish movement as a quantifiable trait has now become widely available, and as such, movement models linking migration or metapopulation dynamics to environmental change will likely become more common. However, a key question is whether movement is directly linked to population productivity, or is a factor which influences other indicators and is thus closer to the indirect influence of density dependence and disease. Chaumot and colleagues address this question well in two publications (2003a,b), and find three conclusions that we may extend to reason why movement might be better characterized as an indirect effect:

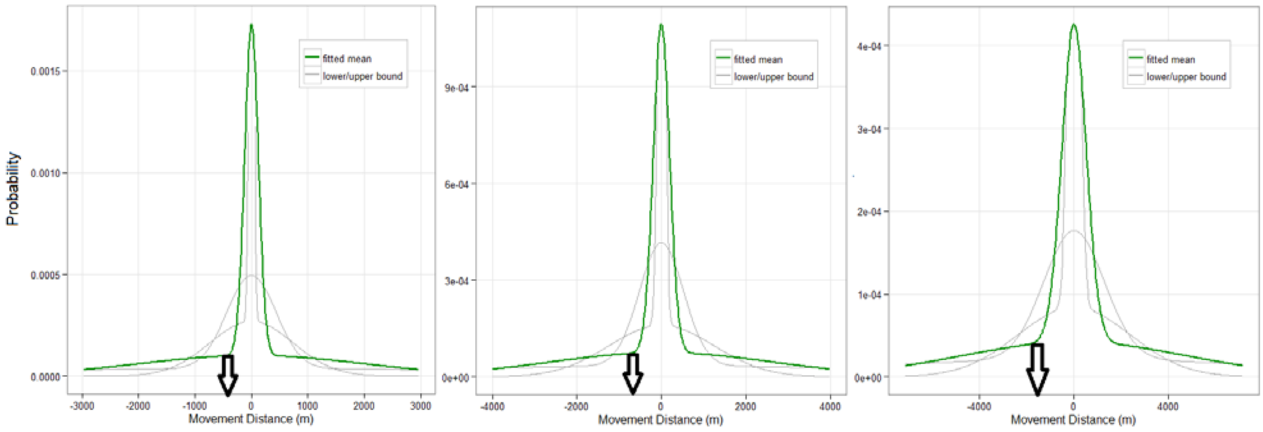
1. Metapopulation dynamics can link productive patches with other ones with high mortality rates and thus reduce productivity across the entire system, however in this scenario movement should not be expected to change in the lower quality habitat,
2. Fish may avoid areas with contaminants, and therefore emigration rates will increase, but if the metapopulation is well connected and suitable habitat was not limited, overall population productivities will not decline, and
3. It may be very difficult to properly characterize the spatial extent of a metapopulation for highly migratory species, and thus difficult to interpret the ecological significance of altered movement patterns.

These three examples ultimately suggest that to better understand movement patterns, another indicator will be required to identify degraded or improved habitat conditions. Advice regarding fisheries reference points from the FAO/ICES supports this conclusion such that spatial distributions are not expected to change the value of reference points, but instead the spatial unit it is applied to. Therefore the challenge is in the application of reference point rather than its derivation.

In the event that migration was considered a suitable indicator (e.g. for fish passage studies) there are appropriate methods to quantify the indicator. The simplest is to measure rates of fish passage at a particular location before and after a barrier is installed or removed. An assumption that was common under the Fish Habitat Management Program was that improvements to fish passage opened up new spawning, rearing, foraging or overwintering habitat upstream and so contributed positively to population growth. This assumption is relatively safe considering the fragmentation of watersheds has been shown to have substantial impacts to the health, productivity and distributions of fish populations in Canada (McLaughlin et al 2006, Bourne et al 2011), and that headwater fish habitat is usually much more sensitive to habitat disturbances and therefore in limited quantity. However, a direct link from habitat quality and individual movement is more challenging to demonstrate. Further not all fish species, or even individuals within a population, react to habitat degradation by altering their movement patterns (Hughes 1999, Reid et al 2002). Shy and bold continuums may divide populations into those that are willing to seek new habitat and those that remain under worsening conditions. Radinger and Wolter’s (2013) riverine fish movement model FISHMOVE assigns a probability of movement to resident and migratory individuals in a population based on the:

1. Size of the watercourse,
2. Size of the fish and
3. Ratio of width to area of the caudal fin, and therefore in one model addresses the individual, species and habitat specific nature of movement (see Figure 13).

This type of model could thus provide appropriate reference points to better predict changes in movement under changing habitat conditions. However, it also suggests that because movement is strongly influenced by body size, it may be best to use size-based indicators and apply the influence of altered movement patterns afterwards.



Aspect Ratio : 0.50 Aspect Ratio : 1.00 Aspect Ratio : 1.50

Figure 13. The aspect ratio of the caudal fin is changed left to right from 0.5 to 1.0 to 1.5. Note that the shape of the distribution changes little but instead x-axis which shows the distance moved does (indicated by the arrows). All three of these figures are developed for a relatively large fish over a 3 month period in a 2nd order stream in which half the population are movers.

Component of Productivity: Reproduction

Fecundity

Like mortality, fecundity is a fundamental parameter in Leslie Matrix-Models and Life-Tables which are used to estimate the intrinsic growth rate of populations (Velez-Espino and Koops 2009) and is recognized as a key fisheries management parameter (Patrick et al. 2010). Fecundity is typical measured as absolute (i.e. the number of eggs per female) or relative (i.e. the number of eggs per kilogram per female). Standard estimates of fecundity can be found for many fish species in Canada (Teletchea et al 2008, Mims et al 2010) as well as allometric relationships with body size (Minns et al 1996) which all can provide suitable reference points. Patrick and colleagues (2010) report that absolute fecundities of less than 100 eggs per female generally denote vulnerable populations, however, few Canadian freshwater fish contain such low levels of egg production. While changes in fecundity have intuitive linkages to population productivity, in wild populations the effects of age-specific mortality and typically strong density-dependent regulation in the first years of life will obscure relatively small changes in egg number or quality (for example, McPhie and Campana 2009 doubled mortality rates for young-of-year skates in a demographic analysis to gain more reasonable demographic models which would obscure even large changes in fecundity). Further, many fish females contribute vastly more eggs than will be fertilized or even hatch, and so a large buffer between population decline and lost egg production may exist across many species. In extreme examples, the bulk of population growth might be contributed by a few very fecund individuals, rather than from an average fecundity across the whole adult population. A great example includes the Sea Lamprey for which only a handful of successful upstream migrants could repopulation an empty stream. Therefore, fecundity may be a useful indicator only once egg numbers decline to extremely low values. Last, fecundity is expected to be strongly correlated to body size and maternal condition (although scarce empirical support for the role of maternal condition was found in Koops et al. 2004), and so within a suite of indices that include size-based metrics, the need to remove redundant measures (Jia and Chen 2013) could lead to the exclusion of fecundity as an indicator.

Gonadal Somatic Index (GSI)

The GSI was briefly addressed in the Stress Indicator section as a potential indicator of altered individual performance due to contaminants or metabolic and physiological stress. However, it is just as often used to estimate the effects of habitat change on a population's reproductive investment. The GSI is calculated as the ratio of the weight of the fish's reproductive organs to its total body weight, is best measured near the spawning season and is conceptually linked with the shift from somatic to reproductive growth in a biphasic or Von Bertalanffy growth model (Lester et al. 2005). The weight of fish reproductive organs is sometimes considered a better estimate of reproductive ability than direct egg counts as egg quality can vary substantially and have a large effect on early life survivorship (Sadovy 2001). Also, while the GSI is calculated for each individual it is most useful as a population level statistics, and therefore should have a strong link to population productivity. The main limitation of the GSI is that it appears to suffer from high coefficients of variation (Faller et al 2003) which could obscure even drastic changes in population health over long periods of time (Petreman 1990). Further, it isn't clear how changes in GSI should be characterized across sexes. For example, under the effluence of a pulp and paper mill, 21 of 24 female populations demonstrated declines in GSI while male populations were evenly split between declines and no observation change (Kovacs et al 1997). Similarly, species differences in sensitivity were found between walleye and whitefish under the same effluent which was reasoned to be due to differences in reproductive strategy (Johnson et al 2012). This final point illustrates a general lack of systematic linkages between worsening habitat conditions and reproductive investment in fish. For example, under 12 weeks of food deprivation no changes in GSI were observed in Atlantic Cod by Dutil and colleagues (2006). However, Gunderson (1997) found that GSI was highly correlated with adult instantaneous mortality which provides an attractive link back to intrinsic population growth rates.

Spawner Density

Density of Spawners or Total Spawner Biomass is recognized separately from general abundance metrics as a robust indicator under the EBFM framework (Fulton et al 2004b). However, as mentioned above, the density of spawners may not be directly correlated with population growth unless the number of adults decline to very low levels. If so, other size-based indicators such as "Biomass" or "Abundance" should also exhibit a noticeable change. However, the benefit of sampling over known spawning shoals (or other types of spawning habitat) is that it allows for more efficient population estimates through mark –recapture within aggregates of individuals. However, this benefit is only accrued for species with strong site fidelity or within closed systems where all suitable shoals have been identified. On the other hand, spawner-recruitment curves are often based on the negative influence of too many spawners due to density-dependent regulation and increased egg and juvenile mortality rates. However, here too juvenile mortality rates will be a more critical indicator for population growth then the high density of adults. This is particular true if spawning success is based purely on the availability of suitable habitat (e.g. the number of spaces available on the shoal) such that once the adult densities exceed the number of available redds on the shoal, population recruitment will simply level off rather than decline.

Egg, Larval and Juvenile Mortality

The mortality rates of eggs and juveniles have been identified as robust predictors of population growth across many indicator reviews and demographic studies (Fulton et al 2004b, Dunlop et al. 2003, McPhie and Campana 2009, Gedamke et al 2007, Perez-Dominguez et al 2012, Velez-Espino and Koops 2009). As mentioned above, fish fecundity rates are generally high and age-specific natural mortality rates often decline as fish age and escape predation or metabolic vulnerabilities, and so early life survivorship can represent the greatest bottleneck to population growth. However, measuring early-life stage mortality can be challenging for many species and require intensive surveying through cohorts and years. Natural mortality rates for adults can be estimated from von Bertalanffy curves, however juvenile mortality rates are expected to be one to three orders of magnitude higher (see Velez-Espino and Koops 2009 for black redhorse mortality rates between age 0 and 11).

DATA NEEDS FOR COMMON INDICATORS

Field or meta-data requirements for all the metrics identified in Bradford and colleagues (2013) and Fulton and colleagues (2004b) were compared and contrasted to determine if there were any similarities between indicator development in the EBFM field and the emerging FPP framework. Perhaps not surprisingly the top data needs for metrics identified by Fulton and colleagues (2004b) included primarily biomass and abundance, with a secondary set of diet/consumption and mortality while for Bradford and colleagues the data needs were more varied and equivalent including length/weight frequency data, abundance, mortality and fecundity. Interestingly, disease and movement data were both considered of a low priority for both fields. Fulton and colleagues (2004b) indicated what type of supporting information would increase the robustness of the estimate of a metric. Reference samples, a suite of species responses, and spatial resolution were the top considerations.

INFLUENCE OF DENSITY DEPENDENCE

Density dependence is considered a sub-component of productivity under the FPP framework (Bradford et al 2013) and is expected to exert an influence on many of the indicators discussed in the previous sections. Density dependence alone is a vague term which only refers to the relationship between a metric and the abundance of individuals within a defined parameter (e.g. volume of habitat, age-class or trophic level). While density dependent mechanisms can be estimated using statistical and simulation models (e.g. Minto et al. 2008), their accurate determination would require a level of quantitative analysis that likely go beyond the reasonable bounds of complexity for most projects addressed by the *Fisheries Act*. Fortunately, density dependence as defined in regards to the productivity endpoints is better understood and easier to define. Therefore the challenge is mainly left to use qualitative knowledge of density dependent processes as a modification to the expected response of an aspect of fish life history to some sort of external environmental pressure. Thus, under the framework proposed above, the addition of density dependent effects to an indicator would occur at the level of the interpretation of the qualitative (or quantitative) change of the indicator (rather than the measurement). If there were reasons to believe that density dependent effects were likely to occur, the state-response curve, indicator reference point or the management decisions would require some sort of qualification. This requires that:

1. Situations typically involving density dependence be identified, and
2. At least qualitative response to density dependence be understood for each main indicator.

The former of these two requirements is easier to address. There is guidance from theoretical and empirical studies on when density dependence is most likely to be occurring. Further, some simplifying assumptions could be made such only a few key situations required the consideration of density dependent effects. For example, density dependence is expected to be a stronger factor for long lived life histories, juvenile stages of development, unfished populations, and predation in freshwater systems. The weak density dependent effects on fast growing species or populations with high exploitation rates simplifies the treatment of the indicators greatly by allowing managers to generally ignore density dependence for a majority of

CRA fisheries. Winemiller (2005) argues that the triangular representation of life history strategies in North American freshwater fish from Winemiller and Rose (1992) can be used to distinguish between species for which the intrinsic population growth rate will not be strongly influenced by density dependence (i.e. the carrying capacity). Winemiller (2005) generally characterized the influence of density dependence as a negative relationship with a fish's fecundity (also found by Goodwin et al 2006). Other theoretical work has disentangled the logistic growth equation to separate r and K on the assumption that in many cases r will be a constant and independent of density dependent factors that don't explicitly modify growth rates (De Vlader 2006). Further, depensatory mechanisms from low population sizes (e.g. increasing catch rates) are likely more of a concern regarding exploitation, rather than habitat degradation if assuming that Alee affects in nature are still rare (Liermann and Hilborn 2001). Predicting the qualitative effect of density dependence may be more difficult for the magnitude of change but shouldn't be as hard for the direction of change. This approach may not address some of the more unexpected responses from density dependent processes (e.g. unstable dynamics, odd equilibrium points), but the derived benefit from losing this degree of accuracy will follow the recommendations in previous sections for "reasonable and conservative advice" over "complicated and uncertain advice".

A small but notable exception to generally discounting the effects of density dependence on CRA fisheries is the effect of an interaction between exploitation and habitat loss. Because regional harvest rates from recreational and subsistence fisheries are not well known but are expected to be generally high (see Post et al 2002, Hunt and Lester 2005, Post 2013) if a new development negatively impacts fish populations and local anglers respond quickly to declining catch rates by fishing elsewhere, the release from the harvest may contribute to misleading indicator values such as increased biomass and maximum size (Abrams 2002). Therefore, the state of the local fishery may also make a suitable secondary indicator.

COMMUNITY METRICS AND BIODIVERSITY

Community and biodiversity metrics are a main focus of the EBFM field (Jennings 2005) as well as make up a large proportion of models submitted to the journal *Ecological Indicators* (Jorgensen et al 2013) and so there is a great deal of guidance on incorporating appropriate metrics for monitoring community change. However, many community metrics are geared towards measuring general ecosystem health (e.g. the Index of Biotic Integrity) and not the ongoing productivity of a fishery. Determining how biodiversity interacts with productivity in a general ecological context is a hotly debated topic (see debate surrounding Mittelbach et al 2001) for which evidence supporting positive correlations or unimodal relationships are most common. Unimodal relationships are thought to occur because at high biodiversity interspecific interactions are expected to be high which reduce overall productivity (Mittelbach et al 2001). How these general ecological relationships translate to fisheries productivity is not clear. For example, translating these relationships in terms of forestry yields could be misleading as monocultures generally exhibit the highest levels of productivity. However, multispecies yields in fisheries are thought to be lower than the sum of the individual MSYs because of interspecific interactions (Pauly 1980) which would theoretically agree with a unimodal relationship. Therefore in high productivity / high biodiversity environments it's possible that reductions in biodiversity through habitat degradation to could lead to brief periods of higher fisheries productivity in terms of overall yields. Yet if habitat loss continued the productivity would begin to decline as well. Like with density dependence it may be possible to determine which species are most likely to be sensitive to changes in biodiversity and so only apply community metrics to management decisions involving indicators in specific situations (Le Quesne and Jennings 2012). However, using biodiversity simply as a modifying influence on the indicators presented in this review seems to overcomplicate the FFP framework and ignore the main benefit of multispecies metrics. First, it overcomplicates the FPP framework because the main mode of action for the effect of biodiversity on productivity is density dependence which will therefore affect the same organisms (e.g. long lived, low fecundity) in the same type of situations (e.g. low exploitation) that are already covered in the density dependence section. Second, community metrics were originally developed due to the recognition that different species reacted to environmental change at different sensitivities, and thus could act as sentinel species for environmental degradation. Karr's (1981) Index of Biotic Integrity is one of the most commonly used community metrics (734 citations in academic journals from 1981 to 2011 as cited in Ruaro and Gubiani 2013), is considered a robust indicator for EBFM (Fulton et al. 2004b), and has been used to monitor a wide range of environmental change from habitat infilling (Long and Walker 2005), riparian clearing (Gergel et al. 2002) to aquatic contaminants (Mayon et al 2006). The strength of the IBI as a metric is similar to that of disease indicators in that though it will not necessarily exhibit direct links to population productivity, its sensitivity to environmental change can provide an early warning that other indicators might soon require further investigations (Mayon et al. 2006).

DISCUSSION AND RECOMMENDATIONS

FRAMEWORK FOR LINKING INDICATORS TO PRODUCTIVITY

This literature review has gained insight into the use of different metrics by other fields of fisheries research, management and biomonitoring and also in their theoretical links to "ongoing productivity of CRA fisheries". It sounds simplistic to say that any indicator can be linked to fisheries productivity given a proper set of assumptions, but this is generally true and a great strength of this field of research. The many linkages back to the intrinsic rate of population growth, carrying capacity or steepness coefficient offers flexibility to proponents to demonstrate quantitatively, if needed, how proposed alterations to habitat is expected to affect the CRA fisheries. In the great majority of cases, we would expect that proponents would measure only the indicator and reference the specific qualitative linkages proposed in this review and others. A common theme that has emerged throughout the literature review is that there is likely no "one-size-fits-all" indicator, or even set of indicators. This is not surprising as considerable effort was expended under the FHMP to find simple and ubiquitous metrics, yet it also proved challenging. A suite of indicators is likely most effective, but will also only be required for a particular size of project. Further, through the search for specific indicators we have discovered that some are already well parameterized for Canadian fish, offering reference points and relationships (often allometric) with other indicators.

This final section of the review will offer a set of general insights on the use of indicators under the FPP framework and discuss how to implement them in field or desktop studies.

1. The FPP must be explicit about the fisheries productivity endpoints, even if they are only conceptual in nature and often beyond the capabilities of the data available for any given project. The intrinsic growth rate (r) and the population carrying capacity (K) were the two most ubiquitous and flexible endpoints found in this review (*note* the steepness coefficient is much more common in stock assessment). The great benefit of these endpoints is that they inform both upon fisheries and conservation priorities, and so can be applied under different situations. There are additional endpoints that may be valuable including those explicitly related to von Bertalanffy growth curves, or estimates of natural (M) or Total (Z) mortality rates. Generally, all these endpoints have more of a conceptual than practical value because, as described in this review, so many models lead to them. But once these endpoints are formally stated as the goal of the FPP framework, they can

be applied to many aspects of population and community demographic theory (including fisheries yields, extinction probabilities and life history evolution).

2. The FPP must not require, but should suggest and make available, the capacity for proponents to use the linkage models to estimate fisheries endpoints. While ecological models are still seldom used except for large projects, even a culvert installation requires fairly sophisticated engineering models and ecological modelling capacity is building among consultants (see Golder Associates' 2008 Habitat Suitability Indices for the Oil Sands Region). Further, there is a growing trend of engineering companies acquiring environmental consulting companies (e.g. Stantec Inc. purchase of Jacques Whitford Ltd. in 2008) to complete environmental permitting "in house". Last, the capacity for "quick and dirty" models to be easily run under free and common platforms (such as R), as well as the propensity of modelling in undergraduate courses is increasing in Canada. To respond to and capitalize on this recent accessibility of quantitative fisheries models and modellers, a set of preferred fisheries models (e.g. stage structured demographic models, production / biomass models) and guidance on their use should be provided. Under the FHMP a set of technical manuscripts were developed that taught the interpretation of fish habitat models (e.g. de Kerckhove et al. 2008) or how to run simple demographic models tailored to fish habitat in Canada (Minns 2003). These types of publications were of great use to consultants and proponents, yet were not widely adopted or endorsed by Fish Habitat Biologists under the FHMP. An excellent example of providing a range of preferred models for marine spatial planning (without explicitly requiring them) can be found at PacMara's Decision Guide (2011). This review alone found applicable and accessible models in BioEnergetics v3 (Hartman and Kitchell 2008), FISHMOVE (Radinger and Wolter 2013), ALADYM (Spedicato et al 2010) and the API (Minns 2003, Velez-Espino and Koops 2009).
3. An insight from Fulton and colleagues' (2005) analysis was that indicators worked better when data was aggregated by zones then across the entire system. This suggests that projects that span different ecozones should separate the management advice interpreted by the indicators by zone as well.
4. None of the indicators reviewed here were generally expected to show immediate and deterministic change with altered habitat. Nearly all indicators will experience substantial levels of environmental variation, time lags between the environmental state change and the biological response, and indirect effects from other ecological factors. Under the FHMP post-contruction monitoring was typically conducted over long time-scales (e.g. often 5 to 10 years for transportation infrastructure) for medium to large projects to ensure that new aquatic structures were able to become established. This same philosophy will be required for most indicators.
5. Under the FPP a standard set of indicators should be recommended for proponents to use, with clearly stated management interpretation of the qualitative change in the indicator and if possible, limit reference points), *in the absence of any other information*. This latter point ensures that the effects of:

    (a) density dependence,

    (b) community interactions, and

    (c) active harvest from CRA fisheries can be incorporated into the decision making processes if deemed necessary.

    A standard set of indicators will offer certainty for proponents to plan for Environmental Assessments but will not preclude the use of additional indicators if the proponent or FFP biologist find a reason to require them (e.g. based on Fisheries Management Objectives).

This review suggests a suite of indicators which should be used based on the size and intent of the project:

Small Projects

The emerging FPP framework will involve screening of small project such that the vast majority will not require extended scientific review or environmental assessments, and so indicators are generally not expected to be required. Further, the collection of a suite of indicators for projects that are expected to only contribute minimal (or cumulative) impacts to population productivity may quickly become more of an undertaking and expense than the project itself. Thus for small projects the type of indicators discussed in this review may not be appropriate. Instead, regional benchmarks of the existing population productivity of CRA fisheries would be more useful for habitat biologists. The type of indicators more suitable for these types of projects may be the type of habitat impact itself, however this idea is outside of the scope of this review.

Large Projects

Projects which are expected to contribute “serious harm” to CRA fisheries will require environmental assessments and fall under the emerging FPP framework developed by Bradford and colleagues (2013). This review (and others) has generally found that size-based indicators are more robust qualitative and quantitative estimators of fisheries productivity, are more commonly adopted across many management fields (i.e. stock assessment, conservation, habitat monitoring), and will be easiest to measure in the field, interpret change in indicators (see Figure 13 and Shin et al 2005) and to find suitable reference points (Fulton et al. 2004b, Patrick et al 2010). However, it is also clear from this review that the ambient temperature, species identification and size-based measurements for particular stages of development (e.g. juvenile and adult) will greatly add to the robustness of the estimate, sensitivity to environmental change and the interpretation of indicator change. Finally, although CRA Fisheries are the focus of the FPP framework, it is important to track the changes in forage and course species as well to understand how density dependence and biodiversity might be changing along with the indicators. Thus a suitable set of indicators that address these insights are recommended as a hierarchy with three levels (however note that the order of bullets within each level does not represent a preferred sequence of indicators):

Primary indicators:

- Body size (e.g. max length, all species)
- Total biomass (all species)
- Juvenile abundance for CRA species

Secondary indicators:

- Relative abundance of sentinel species (Index of Biotic Integrity approach).
- Size of maturity for CRA species

Tertiary indicators:

- Pathology/Stress
- Growth Rate (from time series)
- Juvenile Mortality (from time series)

- Fecundity
- Regional Fishing Effort
- Benthic Invertebrate Abundance
- Movement / Migration

The number of indicators selected from this list would likely be project specific, and should be prioritized within Primary, Secondary and Tertiary levels depending on what will work best operationally, although it is recommended that all the Primary Indicators be included as often as possible, as well as one Secondary and one Tertiary Indicator. This proposed set of indicators is not an exhaustive list and so additional Indicators within the Primary, Secondary and Tertiary levels could be selected as long as it could be scientifically demonstrated that they were of comparable sensitivity to the expected environmental change and statistical power under environmental variability to the existing Indicators on the list. Further if a target species was considered vulnerable by existing measures, other indicators (e.g. regional fishing pressure) could be applied to interpret the change in indicator value under additional ecological mechanisms (e.g. density dependence).

Offsetting Indicators

Indicators used for offsetting plans should generally agree with the indicators used for the environmental assessment to ensure that changes in population productivity are equivalent across projects. However, there are many reasons why additional indicators and additional effort might be beneficial to proponents at this stage. For example, assuming a proponent has to demonstrate that the offsetting plan was successful, the use of simple size based indicators could require long time periods before any change was noticed (e.g. 16 years for Mean Length in Nicholson and Jennings 2004). The use of more detailed indicators that require greater effort in the short term (e.g. juvenile mortality or juvenile growth rates) could demonstrate noticeable change much faster (Jones et al. 2003). This cost / benefit analysis should be left up to the proponent, yet it still requires that those types of more detailed indicators are understood under the FPP framework.

UNCERTAINTY IN INDICATORS

Most papers within the literature review attempt to address and reconcile different types of uncertainty in fisheries models. While uncertainty can occur at many scales (e.g. process, model, decision) the challenge for the indicators is to reduce the measurement errors surrounding the field based estimates of the metrics. In the previous sections some metrics were characterized as more reliable than others, but in reality all metrics will be unreliable if they have high coefficients of variation. The main source of variation that requires active management is the fisheries surveys. First, and at the very least, proponents must justify their sampling protocol rather than report it. The effort expended (e.g. in electrofishing seconds) must be justified by the area under investigation and the target fish species. Further, a qualitative or quantitative argument must be made to estimate the completeness of the fish survey. The most quantitative methods would include block nets, multiple passes and species accumulation curves. Jones and Peterman (2012) and Peterman (1990) demonstrate that it doesn’t take a lot of variation to guarantee that a reasonable change in an indicator will not be noticed over even a decade of monitoring. Second, all sources of variation in indicator values must be reported (e.g. population means must be accompanied by standard deviations or error estimates). Sources of error were less of a concern under the FHMP because generally habitat was quantified only by area. Indicators were not required because post-monitoring programs judged the effectiveness of a compensation project by it having been properly installed. If indicators will be used to track the effectiveness of protecting the ongoing productivity of CRA fisheries, understanding the potential sources of uncertainty will be critical. In the absence of error estimates, a precautionary approach should be followed.

In a 2012 report for DFO titled “A Guide to Habitat Loss and Gain Accounting Methods”, I demonstrate that the majority of habitat assessment models (under the FHMP) can be parameterized with a relatively small set of biophysical and fisheries survey information. For the 19 models I examined (almost identical to those in Minns et al 2011) the most common minimum survey requirements across all methods included (with their percentage of use):

1. Standard Physical Features (94%);
2. Site-Specific Assemblages (63%);
3. BACI (Before After Control Impact) Sampling Design (79%);
4. Population Size Estimates (42%); and
5. Individual Fish Morphometrics (47%)

In the same report I examined the few available provincial standard environment assessment protocols (AB, BC and ON) and found that “The provincial protocols address elements of Standard Physical Features, Site-Specific Assemblages, Population Size Estimates, and Individual Fish Morphometrics for which the specific data collected were largely shared across protocols. The protocols also offer instruction on the need for multiple sites, sub-sampling designs and reference sites and thus address the BACI design.” Fulton and her colleagues (2004b) extensive review of EBFM indicators also reported data needs, which were briefly summarized in this review in the previous section. An early recognition of the type of data collection methods that lead to robust estimates of indicator values would be very useful for the FPP program because in all likelihood they are not more stringent than many of the existing survey protocols. However, they do need to be highlighted so that proponents can avoid methods that will contribute too much variation to the indicator value.

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APPENDIX

Summary of potential productivity indicators proposed for the Fisheries Protection Program (FPP) framework.

Indicator	Impact	Assessment	Indicator Conclusion	Reference
abundance	fishery	FS	Catch Per Unit Effort is likely to stay high even as abundance declines in cod, flatfish and gadiformes	Harley et al 2001
abundance	canopy cover	FS	Although there was a 50% reduction in canopy cover there was no change in abundance	Ellis and Bell 2013
abundance	contaminants	FS	In a comparison of juvenile abundance between pristine and contaminated sites over three years a strong correlation was found (ie. r = -0.73)	Gilliers et al 2006
abundance	detect change	FS	The detection of a 25% decline in abundance over 5 years was not possible with a statistical power of 0.8, instead 15 years of sampling would be required, however including age and stage class would substantially improve the efficacy of the indicator.	Al-Chokhachy et al 2009
abundance	detect change	FS	The detection of a 50% decline in abundance over 50 or 25 years would take 11 to 69 years to observe (depending on the species)	Hatch 2003
abundance	detect change	FS	The detection of declines that would lead to a listings on the IUCN 1 Red List Criteria (>90%, >70% or >50% declines over 10 yrs or 3 generations - that's 6.7%, 11.3% and 20.6% per year) would require at least 10 years to observe	Maxwell and Jennings 2005
abundance	detect change	FS	The detection of a 5% per annum decrease in dolphins would require > 8 yrs to detect (although biennial surveys would work just as well)	Wilson et al 1999
abundance	eutrophication	FS	Changes in abundance correlates well with habitat change as long as an abundance of omnivorous fish is included (i.e. functional relationships are required to improve accuracy)	Argillier et al 2013

Indicator	Impact	Assessment	Indicator Conclusion	Reference
abundance	industrial development	FS	Negative associations were found between fish abundance and an index of inductrial development across roughly 40 and 70 sites for bull trout, whitefish and rainbow trout, but not for grayling and white sucker.	Scrimgeour et al 2008
abundance	contaminants	FS	No change in abundance was detected across a range of contaminated sites, even though an Index of Biotic Integrity did.	Bervoets et al 2005
abundance	habitat degradation	FS	This study was marked by generally high abundances, and while population growth appeared in decline over 5 years of observations, the confidence intervals around the annual predictions often suggested no observable change	Budy et al 2007
abundance	fishery	FS	Catch Per Unit Effort remained highly variable (CV = 100 to 121%)	Iserman 2007
abundance	acidification	FS	Changes in abundance were noted relatively quickly for declining forage fish populations (fathead minnow and sculpin), but less quickly for sport fish (lake trout and white sucker), suggesting that sport fish were reacting to a decline in their forage base, and so a time lag was introduced into the data	Mills et al 1987
abundance	industrial development	FS	Negative associations were found between the abundance of bull trout and an index of industrial development in over 50 sites	Ripley et al 2005
abundance	fishery	FS	To detect a change in age 3 mean abundance from increasing length limits from 203 mm to 229 mm or 245 mm in a crappie fishery would require 12 or 2 years, respectively under a statistical power of 50%.	Martin and Maceina 2004

Indicator	**Impact**	**Assessment**	**Indicator Conclusion**	**Reference**
abundance	salinity	FS	A negative correlation was found between abundance of young of year flounder and the salinity of their nursery habitat (r = -0.47). The change in biomass was related through to population production with P:B ratios.	Franco et al 2010
age at maturity	food availability	EX, TH	Low food could elongate time to move from a larval to a juvenile stage in a Chironomid midge. In this example a 11% change in age at maturity is expected to lead to a 5.6% change in population growth.	Charles et al 2004
age at maturity	fishery	FS, TH	Age at maturity and maximum age were better predictors of population growth rate than body size for three species of skates which had all decreased in abundance by over 90% since 1970.	McPhie and Campana 2009
age at maturity	fishery	MT	A significant correlation was observed across 63 stocks of commercial exploited fish such that late maturity appeared to contribute to low maximum recruit production. Late maturity is generally found in fish species with low fecundity, large body size and low body growth rates. Fecundity was not a good indicator for low maximum recruit production because it positively correlates with body size. However, no relationship with adult production was found.	Denney et al 2002
age at maturity	fishery	FS, TH	A shift of age of maturity in atlantic cod from 6 to 4 years would result in a 25-30% reduction in population growth.	Hutchings 2005
biomass	eutrophication	FS	Only biomass demonstrated a noticeable change to a high fertilization treatment with lake whitefish and remained high after fertilization ceased (unlike growth rate and survival at age 0).	Mills and Chalanchuk 1987
condition	temperature	FS	Across 10 atlantic cod populations condition (Fulton's K) is positively correlated with recruitment potential and affected strongly by temperature, weight at age 4 and fishing mortality.	Ratz and Loret 2003
condition	contaminants	FS	Condition in european juvenile flounder appears to be related to metal contamination of sediment. Sample sizes of 30 individuals across 6 sample sites demonstrated low standard deviation and significant differences from one year of study.	Henry et al 2012

Indicator	**Impact**	**Assessment**	**Indicator Conclusion**	**Reference**
condition	contaminants	FS	No correlation between condition and contaminant levels in the common sole's nursery grounds were found. High variation in the indicator reduces statistical power to detect change.	Gilliers et al 2006
condition	food availability	EX	Condition explains how fish change their age at maturity under different food levels better than length or age estimates alone. This study recommends measuring weight and length in fish populations.	Uusi-Heikkila et al 2010
condition	food availability	EX	Initial condition in cod were hypothesized to influence growth rates during recovery periods following extended periods of starvation. However, in the experiment mortality rates were high and so condition was found to correlated with age of maturity and mortality due to disease, but not with recovery in growth following 7 to 12 weeks of food deprivation.	Dutil et al 2006
condition	nutritional status	FS	Condition factor was found in bluegills to be a seasonably variable and qualitative measure which is influenced by too much by other factors to be a direct estimate of actual physiological condition	Copeland et al 2008
condition	food availability	FS	A good correlation between condition factor and declines in prey densities were found in chinook salmon but not lake trout. This is because variation in condition is contributed by variation in length at age, and so this study recommends the indicator only as a qualitative measure.	He et al 2008
condition	food availability	FS	Four measures of condition factor demonstrated good correlations with physiological condition of lake whitefish under varying prey densities in Lake Huron.	Rennie and Verdon 2008
condition	eutrophication	FS	Lake whitefish demonstrated a noticeable change to high fertilization treatments as an increase in the mean condition that was 3 to 4x the confidence interval in the 7th year of fertilization	Mills and Chalanchuk 1987
condition	sedimentation	EX	Condition decreased in high sedimentation over 28 days of a high sediment concentration treatment in coral reef damselfish, however, mortality also increased to 50% in this treatment as well. The decreases in condition was due to lower consumption rates under turbid conditions.	Wenger et al 2012

Indicator	Impact	Assessment	Indicator Conclusion	Reference
fecundity	fishery	MT	In the analysis of a 50 year California Cooperative Oceanic Fisheries Investigation larval fish record, it appears that truncated age distributions from fisheries can lead to unstable population dynamics through its effect on fecundity and recruitment.	Anderson et al 2008
fecundity	contaminants	EX	In cunner a 38.7% decrease in egg production resulted in an estimated 10% decrease in population growth rate over a 2 week exposure period to endocrine-disrupting chemicals. This indicator was considered robust for the type of environmental impact.	Gutjaur-Gobell et al 2006
consumption rate	food availability	EX, TH	As an indicator consumption rate is sensitive to lag times as populations of midge larvae still grew under low food density.	Charles et al 2004
growth rate	contaminants	FS	A strong relationship was found between the level of heavy metal contamination in the sediment and the growth rates of the common sole between sites over 3 years (r = -0.67).	Gilliers et al 2006
growth rate	contaminants	FS	Lower growth rates were found under higher levels of contamination for yellow perch.	Eastwood and Couture 2002
growth rate	contaminants	FS	Lower growth rates were found under higher levels of contamination for marine benthic fish.	Forrester et al 2003
growth rate	contaminants	EX, TH	Response of tilapia population growth rate to arsenic was examined in laboratory experiments which concluded that reduced feeding led to lower growth rates and higher stage specific mortality rates. Through bioenergetic modelling the highest arsenic trial was expected to decreases the instrinsic population growth rate from 1.0027 to 0.9935.	Liao et al 2006
growth rate	fishery	MT	A significant correlation was observed across 63 stocks of commercial exploited fish such that growth rate was a robust indicator of rates of adult production (r^2 = 0.4).	Denney et al 2002
growth rate	habitat degradation	FS	Growth rate of cutthroat trout were higher in more pristine, high elevation sites on the Logan River however a direct link to population productivity was not reported.	Budy et al 2007

Indicator	Impact	Assessment	Indicator Conclusion	Reference
growth rate	acidification	FS	Growth rate decreased for lake trout but increased for white sucker over an 8 year study of experimental acidification in a natural lake.	Mills et al 1987
growth rate	eutrophication	FS	Lake whitefish demonstrated a noticeable change to high fertilization treatments with increased growth rates at higher levels of eutrophication	Mills and Chalanchuk 1987
growth rate	sedimentation	EX	Growth rates decreased over a 28 day experimental increase in sedimentation for damselfish due to reduced consumption rates. Mortality rates were also high.	Wenger et al 2012
growth rate	fishery	TH	Large and positive selection differentials are found on growth capacity of northern pike under size-selective fisheries, which suggest that growth rate is a robust indicator under these conditions.	Matsumura et al 2011
growth rate	temperature	FS	40% of variablity in growth was explained by temperature for juvenile atlantic croaker, and other important factors were density dependence and selective mortality. The study was conducted over two seasons of recruitment.	Searcy et al 2007
GSI	contaminants	FS	GSI was determined no to be a great indicator of the effects of sewage on river populations of gugeon in Switzerland. The indicator displayed seasonal Coefficients of Variation of 50% in spring, and 20 to 40% in fall.	Faller et al 2003
GSI	contaminants	FS	It was difficult to observe any indicator response to estrogen-like compounds in gudgeon river populations in Europe. Interactions with other compounds was hypothesized to obcure changes in GSI.	Nadzialek et al 2011
GSI	contaminants	MT	GSI in females appears to be a robust indicator of pulp-and-paper mill effluents in Canada for a range of species such that in a meta-analysis 21 declines and 3 no changes were uncovered in relation to reference sites. Males demonstrated a more ambiguous result with 8 declines, 1 increase, and 8 no change.	Kovacs et al 1997

Indicator	**Impact**	**Assessment**	**Indicator Conclusion**	**Reference**
GSI	species differences	FS	GSI is not necessarily comparable across species because different reproductive strategies may be used. This study demonstrates that a predicted decline in somatic energy stores as fish age was only found in walleye and not lake whitefish	Johnson et al 2012
GSI	food availability	EX	GSI did not change under 7 to 12 weeks of food deprivation in atlantic cod.	Dutil et al 2006
histology	habitat degradation	FS	General patterns were discovered in smallmouth bass on the Potomac Drainage of increased lesions and fungal infections with increasing mortality rates and declines in abundance, however it is generally difficult to obtain quantitative estimates of disease rates (and its effects on population productivity) because of multiple stressors and a lack of standardization across jurisdictions.	Blazer et al 2010
histology	contaminants	FS	Under a suite of biomarkers liver histopathology appear to be a robust indicator of the effects of sewage treatment plant effluent on gudgeon in Europe (the coefficient of variation was 15% at all times)	Faller et al 2003
HSI	reproductive investment	FS	HSI is not a consistent indicator as it varies between sexes, populations and species in walleye and lake whitefish.	Johnson et al 2012
HSI	none	EX	HSI did not change under 7 to 12 weeks of food deprivation in atlantic cod.	Dutil et al 2006
migration	contaminants	TH	Migration rates can : 1) Lead to metapopulation dynamics that include patches where mortality is high , 2) Lead to the avoidance of patches with contaminants are then greatly reduce r, or 3) Redistribute biomass around a greater spatial scale than observed. These three outcomes all lead to observing a decline in population growth rates and are all good arguments to applying the influence of migration on indicators rather than including migration as an indicator.	Chaumot et al 2003 a,b

Indicator	Impact	Assessment	Indicator Conclusion	Reference
mixed-function oxidase	growth	FS	Including MFO in a regression model of growth including fish abundance, invertebrate biomass and stream depth only marginally improved fit (Mean Square Error from 0.1 to 0.11 and regression was r^2 0.55 between ln(EROD activity) and length at age 3)	Adams et al 1996
mortality	fishery	FS	In lake whitefish handling mortalities that were less than 50% would be difficult to detect because they had less than 40% statistical power using mark-recapture data.	Arnason and Mills 1987
mortality	fishery	TH, FS	Additional mortality on age-0 causes reduced population abundance and biomass, faster growth rates, and larger sizes at maturation in fish. Additional mortality on individuals over a particular size limit exert a small influence on population abundance and somatic growth, but reduces biomass and age / size at maturity. These results were modelled over 100 years, however it is not clear how quickly any of the changes would be detectable.	Dunlop et al 2007
mortality	habitat degradation	FS	Survival rates in cuttroat trout range from 40 to 80%, and generally increases with age, however in field studies it is very difficult to distinguish between loss of individuals through mortality or migration.	Budy et al 2007
mortality	relationship to growth	MT	A positive correlation with growth and mortality was found through meta-analysis but it is strongly modified by animal behavious and other environmental factors.	Griffiths and Harrod 2007
mortality	fishery	TH	An increase of a post-maturation additional mortality of 0.05 to 0.15 resulted in a roughly 10% decrease in adult atlantic cod biomass.	Kuparinen et al 2012b
mortality	eutrophication	FS	A noticeable change to rates of mortality in young of year lake whitefish under high fertilization treatments were observed but not for any older ages.	Mills and Chalanchuk 1987
mortality	fishery	EX	Over 25 years of study on atlantic silversides suggest that size selective fishing mortality can have a great impact on many traits that relate to overwintering mortality. A rapid decline in yield was observed after 4 generations of size-selective fishing.	Conover et al 2005

Indicator	Impact	Assessment	Indicator Conclusion	Reference
reproductive effort	relationship to growth	MT	Reproductive effort will be strongly correlated with size and age of maturity.	Charnov et al 2001
reproductive effort	fishery	TH	Selection differentials from sized-based fishing limits suggest that reproductive effort is a not a robust indicator of changes in the size distribution of a population.	Matsumura et al 2011
size	fishery	FS	Length of age-three walleye can be difficult indicator to interpret due to high variation. From an overall mean 440 mm, the year class value can differ as much as 140 mm.	Iserman 2007
size	habitat degradation	FS	A change in mean length was found for young-of-year forage fish individuals to be related to habitat conditions (discharge with $r^2 = 0.9$) however this same relationship was not found for older age classes.	Drake et al 2008
size	winter	FS	Juvenile slamonid fish can shrink in length by 10% under unfavourable conditions of winter. This change in length can be observed relatively easily and does not suffer from high levels of variation.	Huusko et al 2011
size	contaminants	FS	In evluating a series of biomarkers on the effects of sewage treatment plants on gudgeon in Europe, the coefficient of variation was found to be much lower for length (10%) than for weight (up to 60%), however overall the effects of sewage treatment plants was quite small.	Faller et al 2003
size	fishery	FS, SS	The size at maturity and maximum length of three skate species demonstrated a fair relationship to population growth rates.	McPhie and Campana 2009
size	fishery	MT	Declines in maximum length was the best indicator of the intrinsic rate of population growth (r2 = 0.44) over 63 stocks of commercially exploited fish species among maximum length, body growth rate, age at maturity and fecundity.	Denney et al 2002
size	fishery	FS	More than 5 years is needed to observe a change in the maximum mass of mature fish within size classes for dermsal fish species in the North Sea. The minimum size class that is included is an important factor.	Jennings and Dulvy 2005

Indicator	Impact	Assessment	Indicator Conclusion	Reference
size	fishery	FS	The recovery of a spawning aggregation of marine fish was observed using the size distribution of the individuals within the aggregation. In the 4th year of an 8 year study the improvements in recruitment could be observed and estimated. It is worth noting that there was a great deal of scatter surrouding the significant trends (r^2 = 0.12)	Heppell et al 2012
size	acidification	FS	Lake trout weight declined following acidification treatments because of the loss of prey species rather than a physiological response to lower ambient pH.	Mills et al 1987
size	fishery	TH	Large and positive selection differentials were found between size at maturity due to a size-based fishery.	Matsumura et al 2011
Spleno-somatic Index	contaminants	FS	Although the coefficient of variation was 25 to 50% the effect of sewage treatment plant was strong enough to be noticeable in gudgeon in European rivers.	Faller et al 2003