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Identification of Species and Habitats that Support Commercial, Recreational or Aboriginal Fisheries in Canada

Identification des espèces et désignation des habitats qui soutiennent les pêches commerciales, récréatives ou autochtones au Canada

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ABSTRACT

The changes to Article 35 of the *Fisheries Act* include a new phrase that contains undefined terminology, i.e., "No person shall carry on any work, undertaking or activity [w/u/a] that results in serious harm to fish that are part of a commercial, recreational or Aboriginal fishery, or to **fish that support such a fishery**". An ecological interpretation of the support functions of an ecosystem are those functions which are essential for sustaining the production of commercial, recreational or Aboriginal (CRA) fishery species within the bounds of natural variability (taking into account managed changes to many populations) over short- and long-term temporal scales. Theoretical and empirical approaches can be used to identify fish that support CRA fisheries by considering the ecological functions that allow CRA fish species to carry out their life cycles. We discuss support functions in terms of direct and indirect support roles. Key prey species and biogenic habitats are considered to be direct support functions while a number of species may indirectly affect the ongoing productivity of the CRA fish species. These include keystone species, wasp-waist species, highly-connected species, apex predators and environment-modifying species all of which have important roles in maintaining ecosystem structure and functioning and therefore indirectly supporting CRA fish species. Changes to the structure of "supporting fish" populations (e.g., in terms of abundance, size structure, spatial structure, genetic structure, distribution) brought about through w/u/a must have the potential to alter the capacity for support species to fulfill their corresponding supporting function(s) in a manner which affects the ongoing productivity of the CRA fish species. Support functions and "supporting fish" populations which affect the productivity of CRA fishery species may occur in areas outside of the distribution of the CRA fishery species and be connected to the CRA fishery species through such mechanisms as food webs, source-sink dynamics or migratory behaviour. When identifying the range of key functional roles played by species supporting CRA fisheries, we describe common characteristics of species fulfilling each role and provide a few examples of such species. We also discuss vulnerability to perturbation and the most common types of human-induced perturbations that will affect support species. For each function, we provide guidance on how to make defensible and consistent decisions on which instances meet our definitions for "support".

RÉSUMÉ

Les modifications apportées à l'article 35 de la *Loi sur les pêches* comprennent une nouvelle phrase qui contient une terminologie non définie : **«** Il est interdit d'exploiter un ouvrage ou une entreprise ou d'exercer une activité entraînant des dommages sérieux à tout poisson visé par une pêche commerciale, récréative ou autochtone, ou à tout **poisson dont dépend une telle pêche.** » Une interprétation écologique des fonctions de soutien d'un écosystème serait les fonctions qui sont essentielles pour soutenir la production d'espèces ciblées par les pêches commerciales, récréatives ou autochtones dans les limites de la variabilité naturelle (en tenant compte des changements subis par de nombreuses populations) à court et à long terme. Il est possible d'utiliser des approches théoriques et empiriques pour identifier les espèces de poissons qui soutiennent les pêches commerciales, récréatives et autochtones en tenant compte des fonctions écologiques qui permettent à ces espèces d'accomplir leur cycle de vie. Nous discutons des fonctions de soutien en ce qui a trait aux rôles de soutien directs et indirects. Les principales espèces de proies clés et les principaux habitats biogéniques sont considérés comme des fonctions de soutien directes, tandis qu'un certain nombre d'espèces peuvent avoir une incidence indirecte sur la productivité à long terme des espèces de poissons ciblées par les pêches commerciales, récréatives et autochtones. Parmi ces espèces, on retrouve notamment des espèces clés, des espèces « en taille de guêpe », des espèces fortement interreliées, des prédateurs dominants ainsi que des espèces qui modifient l'environnement. Toutes ces espèces jouent des rôles importants dans le maintien de la structure et du fonctionnement de l'écosystème et soutiennent donc indirectement les espèces de poissons ciblées par les pêches commerciales, récréatives et autochtones. Les changements de la structure des populations « soutenant les poissons » (p. ex., sur le plan de l'abondance, de la structure selon la taille, de la structure spatiale, de la structure génétique et de la répartition) attribuables à l'exploitation d'un ouvrage ou d'une entreprise ou à l'exercice d'une activité sont susceptibles de modifier la capacité de soutien des espèces à accomplir leurs fonctions de soutien correspondantes d'une manière qui nuit à la productivité à long terme des espèces ciblées par les pêches commerciales, récréatives et autochtones. Les fonctions de soutien et les populations « soutenant les poissons » qui ont une incidence sur la productivité des espèces de poissons ciblées par les pêches commerciales, récréatives et autochtones peuvent être observées à l'extérieur de l'aire de répartition de ces espèces et y être reliées au moyen de mécanismes comme les réseaux trophiques, une dynamique entre les populations sources et les populations puits ou un comportement migratoire. Au moment de définir la portée des principaux rôles fonctionnels joués par les espèces soutenant les pêches commerciales, récréatives et autochtones, nous décrivons les caractéristiques communes des espèces qui s'acquittent de chaque rôle et nous donnerons quelques exemples de telles espèces. Nous discutons également de la vulnérabilité à la perturbation et des principaux types de perturbations anthropiques qui auront une incidence sur les espèces de soutien. Pour chaque fonction, nous donnons des directives sur la façon de prendre des décisions uniformes et justifiables qui, dans ces cas, respectent nos définitions de « soutien ».

1. INTRODUCTION

Old article 35(1):

> 35. (1) No person shall carry on any work or undertaking that results in the harmful alteration, disruption or destruction of fish habitat.

New article 35(1):

> 35. (1) No person shall carry on any work, undertaking or activity that results in serious harm to fish that are part of a commercial, recreational or Aboriginal fishery, or to fish that support such a fishery.

Definitions Provided for Terms used in new article 35(1):

> Serious Harm: “For the purposes of this Act, serious harm to fish is the death of fish or any permanent alteration to, or destruction of, fish habitat”;
>
> Fish: “…(*b*) shellfish, crustaceans, marine animals and any parts of shellfish, crustaceans or marine animals, and (*c*) the eggs, sperm, spawn, larvae, spat and juvenile stages of fish, shellfish, crustaceans and marine animals”;
>
> Commercial: “in relation to a fishery, means that fish is harvested under the authority of a licence for the purpose of sale, trade or barter”;
>
> Recreational: “in relation to a fishery, means that fish is harvested under the authority of a licence for personal use of the fish or for sport”;
>
> Aboriginal: “in relation to a fishery, means that fish is harvested by an Aboriginal organization or any of its members for the purpose of using the fish as food or for subsistence or for social or ceremonial purposes”.

The changes to the Fisheries Act (referred to hereafter as “the Act”) introduced through Bill C-38 include a new phrase that contains undefined terminology, i.e., “**fish that support such a fishery**”. The term “fish” retains its original definition and fishery refers to a “commercial, recreational or Aboriginal” (CRA) fishery, leaving open the definition of “**support**” in this context. The objective of this paper is to provide science-based information for operational application of this clause.

Theoretical and empirical approaches can be used to identify fish that support CRA fisheries by considering the ecological functions that allow CRA fish species to carry out their life cycles. At the simplest level, this involves considering the species interaction(s) and habitats that are required for the nourishment, shelter, refuge, and movement of individuals at each life history stage, as well as interactions and habitat necessary for successful reproduction. The litmus test for whether or not fish support a CRA fishery, at least from an ecological perspective, involves considering the population-level impacts of losing the supporting population or species on the corresponding CRA fishery species, noting that impacts may be apparent immediately or at some future time (e.g., those associated with reproduction).

Our approach here has been to identify a non-exhaustive list of generic ecological functional roles, with examples, carried out by species in communities and ecosystems that could be considered key for supporting CRA fisheries in a region. We recognize that these functions will be carried out by different species in different areas, therefore, future assessments will require an evaluation of which species in an area may be providing those support functions. Over time and with increased knowledge gained through application, a database could be built which documents support species, their distributions and habitat usages at different stages, and their known or hypothesized functional roles to further enable an efficient and consistent implementation of Article 35 of the Act by the Department. Building blocks for this database are already in existence and could be modified for this specific application (e.g., FishBase (http://www.fishbase.org/search.php), Ontario Freshwater Fishes Life History Database (OFFLHD; http://www.ontariofishes.ca/home.htm), Lane et al. 1996 a,b,c; Minns et al. 2008).

We have developed a definition of “support” relevant to CRA fisheries and attempt to identify most of the key functional roles played by species that support CRA fisheries in aquatic communities and ecosystems across Canada. Companion papers defining ‘ongoing productivity’ (Randall et al. 2013) and ‘contribution’ (Koops et al. 2013) are also producing operational definitions for a related clause: ‘the contribution of the relevant fish to the ongoing productivity of commercial, recreational or Aboriginal fisheries’ (Fisheries Protection Revisions to Section 6.1 of the Fisheries Act), which relates to habitat- and ecosystem-level interactions that influence fish production. Koops et al. (2013) discuss various response functions between CRA fishery species productivity and metrics of support species but do so only at a high level to conceptualize the outcomes of different management decisions. Our paper extends that work by providing examples of key ecosystem functions that support fish production, which are essential for sustaining fisheries.

When identifying the range of key functional roles played by species supporting CRA fisheries, we describe common characteristics of species fulfilling each role and provide a few examples of such species. We also discuss vulnerability to perturbation and the most common types of human-induced perturbations that will affect support species. For each function, we provide guidance on how to make defensible and consistent decisions on which meet the definition for "support" and which ones may qualify, but may not be sufficiently linked to the ongoing productivity of CRA fisheries under the new Act.

2. INTERPRETATION OF “SUPPORT” FOR CRA FISHERIES

> 35. (1) No person shall carry on any work, undertaking or activity that results in *serious harm* to fish that are part of a commercial, recreational or Aboriginal fishery, or to *fish that support such a fishery*.

Ecological Interpretation:

1. The support functions of an ecosystem are those functions which are essential for sustaining the production of CRA fishery species within the bounds of natural

variability (taking into account managed changes to many populations) over short- and long-term temporal scales.

2. The fish that support the fish that are part of CRA fisheries are fish that provide those support functions; changes to the structure of “supporting fish” populations (e.g., in terms of abundance, size structure, spatial structure, genetic structure, distribution) have the potential to alter the capacity for support species to fulfill their corresponding supporting function(s) as well.

3. For many ecosystems, several species are likely to contribute to providing a particular function and in such cases decisions about which species support fish that are part of CRA fisheries will depend on the strength of ecological linkages among species.

4. In cases where multiple species contribute to fulfilling a supporting function (e.g., species assemblages, prey guilds), usually no single species can be considered to directly “support” the fish that are part of the CRA fishery, unless one species in the suite of species consistently provides a portion of the functional support that cannot be compensated for by other species.

5. Support functions and “supporting fish” populations which affect the productivity of CRA fishery species may occur in areas outside of the distribution of the CRA fishery species and be connected to the CRA fishery species through such mechanisms as food webs, source-sink dynamics or migratory behaviour.

6. In cases where the supporting species or habitats are on the edge of their distributions, support functions and roles may be somewhat different from those in the core range. Consequently the ability of the support species or habitats to withstand or recover from perturbation may also differ. Special reconsideration of support roles and increased sensitivities at distributional margins may be necessary.

3. KEY SUPPORT FUNCTIONS FOR CRA FISHERIES IN CANADA

Ecosystems are complex networks of intraspecific, interspecific and environmental interactions. The flow of energy through an ecosystem is moderated not only by predator-prey interactions, but also by other types of interactions including competitive, mutualistic, and commensal relationships within the framework of their environment. Within an ecosystem, support species can exert direct or indirect influences on CRA fishery species depending on ecosystem structure and the nature of interactions between the species. Although direct interactions (predator-prey, competition) are perhaps easier to detect and document, it is important to note that both direct and indirect interactions have the potential to limit the productivity of a CRA fishery species. In terms of the provisions of Article 35 of the Act, it is not necessary that a supporting species be directly linked to the fish that are part of CRA fisheries, but in order for a species to be considered a supporting species there has to be a necessary link between changes in the population structure and/or availability of the supporting species and expected reductions in productivity of CRA fisheries.

Certain generic, key functional roles are carried out by individuals and populations of species within all ecosystems with varying levels of influence on the dynamics of CRA fish populations, such as top-down control through predation, provision of food as prey (bottom-up control), and the formation or modification of habitats. The species that provide these key support functions can differ across ecosystems and can vary within ecosystems and over time. Further, some species can perform more than one functional role. Protecting the capacity to fulfill these key functional roles and the species essential to fulfilling them is important to maintain the support for CRA fishery species. Here, we outline some of the major direct and indirect support functions that we are likely to find in Canadian aquatic ecosystems (Table 3.1). We note that it will be rare for only one species to fulfill a support function for a CRA fishery species and therefore points 4 and 5 in Section 2 become highly relevant.

For each support function we provide further information including where possible: the types of human activities that are likely to cause important impacts to the support species performing the function; data requirements and examples of acquisition and analytical methodologies for identifying the support role and link to the CRA fishery species; information required when undertaking risk assessments of works, undertakings, and activities (w/u/a) on the support species; how thresholds could be determined for assessing serious harm to the supporting species; and operational guidance on ecological considerations when undertaking a scoping of the application of the new Act. For the latter, we consider whether there could be unique threats that would apply only to the supporting species and not to the CRA fishery species itself and whether these threats could occur in areas outside of the distribution of the CRA fishery species. This could occur when spatial distributions of the supporting species or species assemblage have different but overlapping distributions and where the populations of the supporting species are connected to the CRA fisheries through source-sink dynamics.

The fact that a functional role is described here does not mean that this role occurs in all ecosystems but certainly at least one of these roles will appear in every ecosystem and many are common. We consider the species or group of species which provide these functional roles to be ones to which particular attention needs to be paid in initial scoping and assessments of w/u/a, as they may be important for supporting CRA fisheries. We also note that there will be examples of communities where an additional CRA support function is considered important but is not outlined here. These are most likely to occur in highly specialized environments (e.g., microbial activity in high Arctic lakes). Should such support functions be identified then they should be treated in the same manner as the functions described here. The important aspect of the approach here is recognizing the needs of CRA fishery species in terms of promoting ongoing productivity (i.e., survival, growth, reproduction and movement) and identifying the ecological functional roles played by support species in meeting those needs.

Direct Support Functions

Two key support functions that arise from direct interactions with CRA fishery species are the roles of key prey species and any biogenic habitats that the CRA fishery species requires to complete its life cycle and contribute to the ongoing productivity of CRA fisheries.

Key prey species: Most support functions can be best understood in terms of the energy flow features of the ecosystem, which are strongly, but not solely, influenced by food web structure. Being prey of a CRA fishery species is the most common direct functional role. An important prey of a CRA fishery species provides energy required for that species to fulfill all or part of its life cycle. The flow of energy from the prey species to the CRA fishery species is modified through a functional response defined by the relative abundances and behaviours of the two species in the ecosystem. Functional responses can vary seasonally or among years according to the population dynamics and behaviours (e.g., migration) of the interacting species. CRA fishery species may be highly selective in their food choices, or may have broader diets and be easily able to switch foods (Holling 1959, Chesson 1983). For CRA fishery species with broad diets, identification of species that fulfill the functional role of key prey is more complex and requires establishing the relative importance of multiple species for the CRA fishery species (see Section 3.1 below). In such cases, no one species may provide the "supports" role according to the intent of the Act.

Structure-providing species: Many species provide three-dimensional structures or habitats which are important for one or more life history stages of a CRA fishery species or its food supply. Examples of "fish" species under the Act which provide these support functions include mussel beds and marine coral and sponge reefs. These examples are all biotic (or biogenic) habitats, that is, habitats created by living species. Biogenic habitat created by one or more support species may be essential for sustained production of a CRA fishery species (see Section 3.2 below). It is well known that plants can be key structure-providing species with established links to fish production. These are separately considered in Section 3.8.

Indirect Support Functions

As part of complex ecosystems CRA fishery species are subject to changes not only in the direct support functions provided by other species but also to indirect ones where the link between the CRA fishery species and the indirect support species is not necessarily obvious. We have outlined a few important indirect supporting functions for CRA fishery species that may not be present in all Canadian aquatic ecosystems but when they are present they should be considered in assessing the potential impacts of a w/u/a. It is more likely that a w/u/a's immediate influence on the productivity of CRA fisheries will be mediated through disruption of direct rather than indirect support functions.

Keystone species: Those species which have an impact on ecosystem structure and functioning disproportionate to their biomass or abundance in the ecosystem (e.g., sea otter) are called keystone species. Keystone species do not have to be found at the highest trophic level. The criteria for a keystone species are that the species exerts top-down influence on lower trophic levels and prevents species at lower trophic levels from monopolizing critical resources, such as competition for space or key producer food sources. They maintain community diversity by preying selectively on competitively superior prey taxa, thereby preventing the exclusion of relatively weak competitors.

Wasp-waist species: These species usually occupy an intermediate trophic level and are expected to play a critical role in regulating the transfer of energy from primary and secondary producers to the higher trophic level species in the ecosystem (e.g., capelin).

Apex predators: Apex predators occupy the top trophic position in a community; these are often large-bodied and specialized hunters that have no predators of their own within their ecosystems. They can have a controlling influence on the structure of lower trophic levels, referred to as “top-down” control (e.g., large sharks).

Highly-connected species: a species with a high proportion of links to other species in a food web compared with the average number of links between species (e.g., krill).

Environment-modifiers: An organism that directly or indirectly modulates the physical environment in which it lives in ways that create resources for CRA fishery species and their prey. The activities of these organisms provide abiotic habitats that would not otherwise be available, often by means of disturbance to the physical environment (e.g., walrus).

Whether or not support functions are provided through direct or indirect interactions, there is the possibility that the support species and CRA fishery species occupy areas that are spatially separated, requiring the need to consider the life cycles and habitats of both support and CRA fishery species. Further, as support functions become more indirect, there is an increased likelihood that multiple species will fulfill the functional role. In such cases, no one species may provide the “supports” role according to the intent of the Act.

Table 3.1. Examples of functional support roles for CRA fishery species. The key steps in this ecosystem-based approach are to identify the necessary functions to maintain and support healthy and productive populations of CRA fishery species as well as the functions necessary to maintain the structure of the ecosystems that support such species. The table may not be complete in the identification of all required functions though it should cover most cases. Under some circumstances a function that is not mentioned here may also be considered a key support function and arguments should be made for such cases.

Name	Main function	Characteristics and identification	Potential link to CRA fisheries	Canadian examples	Non-Canadian examples	References
Key Prey Species	A lower trophic level species that is important for energy flow to higher trophic levels.	Small size, low trophic level	Direct prey of CRA species. Comprises a large percentage of the CRA fishery species diet for a large portion of the foraging year or during a key period required for CRA fish production.	Marine: capelin, sandlance, herring, eulachon, shrimp. Freshwater: alewives, shiners, minnows. mayfly larvae and adults (Ephemeroptera), caddisfly larvae and adults (Tricoptera).		Species specific and relatively well studied.
Structure-Providing Species	Provides structure for other species directly through its presence.	Usually sedentary, sometimes calcareous. Sites of organization, congregation, and high diversity and production.	Provides spawning, nursery or adult (migratory, feeding, over-wintering) habitat. May be feeding locations due to increased biodiversity particularly of invertebrates which are important dietary items for most juvenile fish. Often provides refuge and attachment points.	Marine and Freshwater mollusc beds; Deepwater corals and sponge reefs (BC).	Oysters and oyster beds, USA; horse mussels, Scotland.	Zu Ermgassen et al. 2012; Hiscock and Marshall 2006

Name	Main function	Characteristics and identification	Potential link to CRA fisheries	Canadian examples	Non-Canadian examples	References
Keystone Species	Promotes co-existence and increased diversity.	Has a disproportionate influence on community structure, e.g., predator of a highly fecund and competitive species that has dominance potential.	Prevents over-dominance of any one CRA fishery species or supporting species. Their depletion can quickly lead to cascading effects through the food web / environment.	Sea otters BC coast.	Starfish in intertidal Washington; urchin grazing on coral reef algae; salamanders	Paine 1995; Estes and Duggin 1995; McCook 1999; Davenport & Chalcraft 2012
Wasp-Waist Species	Funnels energy / controls energy flow from lower to higher trophic levels.	A species dominating biomass at an intermediate trophic level and which is important prey of higher trophic levels and an important predator on lower trophic levels. They are the transition point between top-down and bottom-up control. If fish, they are usually schooling species with dense local aggregations available to predators but patchy over the entire area. Mostly a phenomenon of marine ecosystems.	Essential for providing energy flow to higher trophic levels where most CRA fisheries occur.	Capelin NL; Sand eel Grand banks; herring southern Gulf of St. Lawrence.	Sand eel NorthSea; Anchovies and Sardines southern Benguela ecosystem; Antarctic krill.	Bakun 2006, 2009a ,b; Frederiksen et al. 2007; Daunt et al. 2008; Shannon et al. 2004; Swain et al. 2000.

Name	Main function	Characteristics and identification	Potential link to CRA fisheries	Canadian examples	Non-Canadian examples	References
Highly-Connected Species	Provides an important primary energy flow and potentially a redundant energy flow in food web thus bringing resilience to an ecosystem faced with perturbation.	Significantly greater than average number of links in a food web analysis. Only easily defined for well-studied ecosystems.	Could potentially be an important alternate prey when the primary prey is less abundant; Could interact indirectly to maintain ecosystem stability.	Zooplankton prey of Key Prey Species.		Scheffer et al. 2001
Apex Predators	A high trophic level or top predator that maintains lower abundance of other species, thereby decreasing dominance (i.e. top-down control).	Large size, high trophic level, highly mobile, often ranges widely	Can maintain lower abundance of predators on CRA fishery species. Also can limit parasite loads in CRA fishery species.	Large sharks off Sable Island which consume pinnepeds, thus indirectly reducing cod predation and limiting seal worm infestation. FW examples other than the fishes themselves would be fish-eating birds / mammals.		Brodie and Beck 1983

Name	Main function	Characteristics and identification	Potential link to CRA fisheries	Canadian examples	Non-Canadian examples	References
Environment Modifiers	Species that modify the physical or chemical environment thus creating habitat for other species.	Usually a bottom dwelling species, creates or destroys natural physical structures and the result is important for other species. Can also be a species such a bioturbator that mixes organic material, oxidizes deep layers and flushes sulphides etc. or a filter feeder that clarifies the water.	Can create / modify a habitat for CRA fishery species especially during early life stages.	Salmon, bass (nest building), walrus, burrowing clams, freshwater mussels, sand dollars.		Jones et al. 1994

3.1 DIRECT SUPPORT FUNCTIONS: KEY PREY SPECIES

The most obvious example of species that support CRA fishery species is the suite of species that are essential food items. These may include both animals and plants (prey and forage). Herbivorous fish (those whose food constitutes more than 50% plant material by weight or volume, at least in some period of their life) are common amongst species in the family Cyprinidae in Canada. All forage species may be protected as fish habitat of CRA fishery species, if their status affects the productivity fish that are part of a CRA fishery. Key prey can be distinguished from general prey items in that the abundance, diversity, availability and/or nutritional value of key prey <u>limit</u> the productivity of a CRA fishery species.

Identification of the principal dietary items of a CRA fishery species is usually achieved by examination of the stomach contents at various life history stages although alternate methods have been developed (e.g., stable isotope, genetic identification, and fecal analysis). Diets are relatively well known for some commercial fish species in some ecosystems and the relative trophic positions for many species and their ontogenetic stages are classified or quantified. Stomach contents remain the main source of data for identifying the numbers, sizes and types of species consumed by fish and other predators (mammals). DNA technologies and stable isotope analysis have been introduced to assist in identification of contents, and detailed data collection manuals are available for standardization of data recording and procedures. Fatty acid composition has also been used as supporting evidence of diet for marine mammals (e.g., Iverson et al. 1997).

Fish diets change as individuals grow: in general ichthyoplankton feed on zooplankton, juveniles feed on benthic or pelagic invertebrates and adults have either a mixed diet of fish and invertebrates and/or plants or eat only one of these groups. Some species have very specialized diets at one or more life stages whereas others are opportunistic and omnivorous feeders. Some species may also have sex-specific diets. Further, key prey species can be geographic or population-specific, and prey consumed may vary according to seasonal changes in abundance and availability. Even when diets are generalized and the CRA fishery species is an opportunistic feeder, the presence of certain prey may influence the productivity of the CRA fishery species through their nutritional value.

There are few examples of *extremely* specialized diets in northern temperate aquatic species; ocean sunfish (*Mola mola*) feed almost exclusively on gelatinous plankton (jellyfish, salps ctenophores) and need very large quantities daily to support their mass (Dewar et al. 2010). The southern resident killer whales (*Orcinus orca*) specialize on Chinook salmon (*Oncorhynchus tshawytscha*), and availability of Chinook has been linked to killer whale fitness (Ford et al. 2010). Examples from elsewhere, include the Crested Bullhead Shark (*Heterodontus galeatus*) found in tropical waters off Australia which feeds almost entirely on red sea urchins. When such extremely specialized diets are known to occur it is also extremely important to protect the prey/forage items of that CRA fishery species.

There are many examples of *highly* specialized diets. For example, narwhals (*Monodon monoceros*) have a relatively restricted and specialized diet particularly in late fall. Examination of 121 stomachs from the Canadian High Arctic and West Greenland in summer, late fall and winter has shown that their prey is predominantly composed of

Greenland halibut (*Reinhardtius hippoglossoides*), polar cod (*Arctogadus glacialis*), Arctic cod (*Boreogadus saida*), and squid (*Gonatus fabricii*). The diet was most diversified in summer with all species except Greenland halibut found in the stomachs. However in fall, only squid were found and winter feeding was mostly on Greenland halibut and squid (Laidre et al. 2004). In such cases when only a few alternate prey choices are available, the total prey availability should be assessed in consideration of seasonal and spatial patterns of feeding.

In freshwater ecosystems, the size of CRA fishery species can be strongly affected by the size structure and type of prey available. For example, adult lake trout (*Salvelinus namaycush*) are opportunistic feeders, eating a wide variety of prey including alewife, smelt, sculpins, minnows, zooplankton, insects and crustaceans depending upon availability. Comparative studies of fish from different lake foodwebs showed that lake trout that did not eat fish grew much slower and remained much smaller than piscivorous lake trout populations (Kerr and Martin 1970, Pazzia et al. 2002). Prey size restrictions are very common.

Also fish may specialize on prey belonging to a particular ecosystem component or prey guild (fish, sharks, krill etc.). For example, the specialized filter-feeding mechanism of baleen whales (such as fin, blue and humpback whales) enables them to feed primarily on zooplankton and schooling fishes. These food sources are often encountered in large swarms or schools, which is important as large baleen whales eat about 4% of their body weight each day during the feeding season and so the *amount* of food available may be more important than the *species composition*, in this example.

When determining which prey species, if any, are important support species for a CRA fishery species the following should be considered:

i) Whether the diet of the CRA fishery species is highly specialized by species or size class in at least some seasons or life-history stages. Evidence that a prey item is consistently common in predator diets is weak evidence for diet specialization. The evidence for specialization is stronger when the prey on which the predator specializes is more common in the diet of the predator than it is in the ecosystem (considering other species of similar size and general spatial and temporal availability to the predator);

ii) Some components of productivity of the CRA fishery species (reproductive success, growth rates etc.) co-vary with the abundance of the prey population, particularly for relatively high or low abundances of the prey. What would be the likelihood and magnitude of population-level impacts on the CRA fishery species if the prey abundance or availability were reduced?

Breadth of diets can be assessed using a wide variety of analytical methods including (but not limited to) species accumulation curves (e.g., Cook and Bundy 2010), k-means clustering (e.g., Rice 1988), and stable isotope analyses (e.g., Vander Zanden et al. 1999). For example, on the Scotian Shelf, an analysis of the diets of 30 marine fish species showed that diet breadth ranged from 11 to 80 prey groups, with an average of 37. However, most diets were dominated by 2-3 species (Figure 3.1.1).

Often detailed diet data are not available but specialization in the diet and the main prey group can often be inferred from just a few morphological characteristics evolved to this specialization (Douglas and Matthews 1992). For example, specialized plankton feeders such as herring have narrowly spaced gill rakers which reflect their main diet item. The spaces between the gill rakers provide some evidence of the minimal size of plankton targeted. Intestinal length, mouth size and mouth orientation may be good indicators of evolved diet preferences and specialization (Ibañez et al. 2007).

Ideally, a key or dominant prey should be identified by its overall energetic contribution to the predator diet after assimilation. Typically, a way of approximating this is by identifying key or dominant prey by their contribution in weight to the diet, together with the number of stomachs containing the prey species (frequency of occurrence), and the stability of these observations over space and time. For example, the average diet of Alaska plaice consists of almost 60% polychaetes (worms) by weight. Species composition may vary but polychaetes as a group are important prey items.

In these kinds of stomach content-based approaches attention should be paid to those cases where the prey species has high calorific value disproportional to its representation in diets, or conversely, indigestible material that has elevated or reduced the importance of the weight in the stomach. Also the prey species might be selected for certain essential nutrients that it contains (e.g., essential fatty acids). Another important consideration when using stomach content data is that, unless the stomachs have been gathered from large-scale, appropriately designed sampling schemes that account for diel, seasonal, inter-annual or stage-specific variation in prey availability and the distribution of the CRA fish predator, there are serious risks of biases in the results associated with clustered sampling, biased representation towards local prey fields due to the small time window represented by the stomach content.

Similar sampling/representation issues may arise with other methods for identification of key or dominant prey (e.g., stable isotopes do not identify prey species *per se*, fatty acid analysis is dependent on the quality of the fatty acid library being used, as well as on specific considerations of the metabolisms of fatty acids). Overall, the sampling design underlying the data collection, as well as the intrinsic biases and issues of the analytical method employed, should be considered commensurate if the diet analysis results are reliable for fully quantitative assessments, or they can only be used in an indicative, semi-quantitative way.

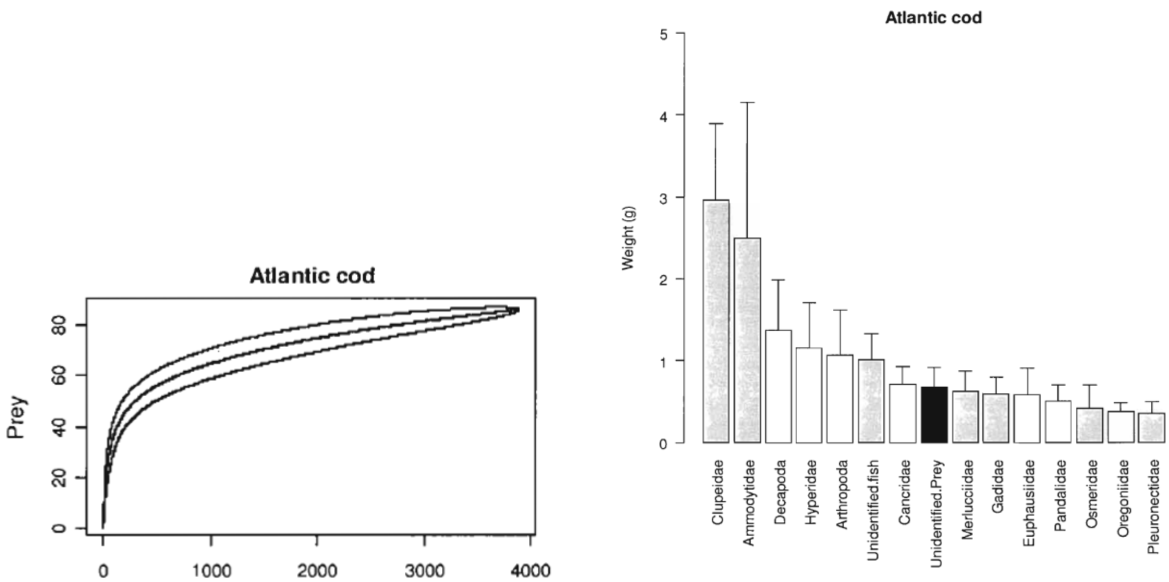


Figure 3.1.1. (left) GS species accumulation curves for Atlantic cod sampled from the Groundfish Surveys. (right) Mean (standard deviation) diet of Atlantic cod captured during RV surveys between 1995 and 2008. Grey shading represent fin fish species, white -invertebrates and plants and black-unidentified prey. Extracted from Cook and Bundy (2010).

A complementary way of identifying key or dominant prey is through predator-prey modeling. Even though quantitative assessment of the diet may not be readily available, the use of predator-prey or food web modeling can be used to test if the changes in abundance/biomass over time of a particular prey can be linked to changes observed in a predator that is part of a CRA fishery. There is a large and diverse set of approaches that can be used to develop predator-prey or food web models; but major constraints would be imposed by the data availability, the precise nature of the question posed, the urgency of the answer, and the amount of resources (both human, and logistic) allocated to model development. In addition, there are multiple ways to formulate how models represent predator choice among multiple prey and the functional feeding responses between predator and prey and choices made in model formulation can affect model results strongly. Most likely, this type of modeling approach to the identification of key or dominant prey would be practical if there are pre-existing models for the system under study that can be leveraged for addressing this question. In the long term, predator-prey, food web, and ecosystem modeling can be developed as strategic tools because they can contribute to addressing multiple inter-related questions (e.g., key or dominant prey, keystone species, wasp-waist species) and inform decisions on managing trade-offs (e.g., in cases where the presence of one species benefits a CRA fishery species but not another CRA fishery species within the same management area). They are likely to be of value as a complement to actual data on diets of fish that are part of CRA fisheries. On their own, however, only very well validated food web models (including parameterization to local conditions) are likely to provide stand-alone evidence that a particular prey meets the requirements for being a *key* prey.

All of the above methods have strengths and weaknesses. A useful method will provide clear evidence regarding the strength of the factors listed above (i.e., i) and ii)), and be feasible with the types of information likely to be available on diets and predator-prey interactions.

Predator-tolerant Prey Species in an Ecosystem Context

In terms of the Act *we define* Key Prey in a *species context*. However this term is also used in the scientific literature to describe a trophic position in food webs specifically a preferred prey species that is able to maintain its abundance in the face of predation (e.g., via a high reproductive or survival rate) and that can affect community structure by sustaining the density of a predator (a potential CRA fishery species), thus reducing the density of other prey (Holt 1977). Others refer to such species as "predator-tolerant prey". In order to avoid confusion between these two concepts we use the term '*Predator-tolerant Prey Species'* to describe such species. Predator-tolerant Prey Species in this context may be Key Prey of a CRA fishery species and so serve a direct link to the CRA fishery species or they may be indirectly linked to the ongoing functioning of the ecosystem in which the CRA fishery species lives. When Predator-tolerant Prey Species are found in the ecosystem in question but are not Key Prey of CRA fishery species as defined above, they should also be assessed against the criteria set in i) and ii), in order to be relevant to the Act.

Methods used to determine prey population presence and (sometimes) relative abundance, include net and acoustic surveys, traps, visual survey methods (e.g., stream walks or cameras or aerial surveys) plankton counter technology for zooplankton, benthic sampling methods (e.g. ponar grabs, kick sweeps), to name a few.

3.1.1 Assessing Impacts on Key Prey Species

Key prey species are vulnerable to a suite of human activities including fishing (either directly through removals in a targeted CRA fishery species itself or indirectly through bycatch of CRA fisheries), release of toxic chemicals, introduction of non-native species, and loss/exclusion of habitat. Mortality can also occur through turbines, water intakes, blasting, etc. CRA fisheries directed at key prey species can threaten other CRA fishery species through competition by fishing for the same food that the other CRA fishery species depend upon for survival, growth and reproduction.

3.1.1.1 Operational Implications

There are no unique classes (or types) of threats that would apply only to the prey of fish that are part of a CRA fishery and not to the CRA fishery species itself, unless the distribution or the habitat usage of the prey and CRA fishery species are different and the threat is specific to the location of the prey and not the predator. As examples, prey could be affected in a location during the time of year when a CRA fishery species is absent through migratory habits; prey may occupy different habitats than the CRA fishery species and only overlap as prey during critical feeding periods; or the viability of prey populations that overlap in distribution with CRA fishery species may depend on dispersers from distant but connected populations. Also the dependence of either the CRA fishery or the supporting species on particular habitats could mean that the impact would be manifested differently even though the species' ranges overlap. In the incompletely overlapping case, natural and artificial barriers that would prevent the prey and CRA fishery species interacting should also be considered.

Once key prey species are identified it will be important to assess the potential for such temporal and spatial separations against the spatial and temporal scales of the activity. There will be no generic answer to how far away (distant) impacts must be considered, as these will depend on the functional role, ecology, and behaviour of the support species as well as the nature of its interaction with the CRA fishery species. In some cases activities may affect whole watersheds (upstream flow changes or contamination), while in other cases impacts may be restricted to a single lake, tributary, or portion of a coastline and be more localized (see Section 4.4 below). Nonetheless, cumulative impacts and multiple stressors still need to be considered in the examination of stability and resilience of the CRA fisheries.

Therefore, with regard to spatial scoping of application of the Act, it is likely that most areas that would be considered for coverage under the Act due to presence of supporting prey would have already been scoped in as areas where the Act applies due to presence of the species that are part of the CRA fisheries. There is an important exception to this generalization, though, when an important prey may spend some of its life history in areas never frequented by the predator; for example aquatic invertebrates or small fishes in headwaters of rivers where CRA fisheries occur downstream. Such areas could be included for application of the Act, if the best available information suggests that the prey imported into the range of the predator from outside its range are indeed the prey that support the fish in the CRA fishery (i.e., source areas). In all areas deemed as fish habitat for the purposes of application of the Act, the status of the important prey in the area, and the habitats they require, become part of the evaluation process for self-assessment or authorization review.

3.1.1.2 Information Required for Performing Risk Assessments

The evaluation of risk associated with impacts to key prey species would be improved if it were possible to assess:

i. the intensity or severity of the impact on the prey species being affected;

ii. the sensitivity/vulnerability of the CRA fishery species to the impact created through the loss of the prey species;

iii. the ability of the CRA fishery species to recover from impacts, and the rate of such recovery;

iv. the extent to which ecosystem functions may be altered by the impact.

The assessment of the above bullets can be done using a diversity of approaches (e.g., prey availability indices, predator-prey, food web, and ecosystem modeling, analysis of trends, or other statistical tools) depending on the availability of science capacity and/or data specific to the ecosystem. The section on data poor methods in the working paper on using the Precautionary Approach (Koops et al. 2013) provides a basis for considering what alternative approaches are available, and which ones might be appropriate in particular contexts.

3.1.1.3 Development of Threshold Levels

The interactions between species in a food web are complex and dynamic. The ICES-JRC report on Food Webs (Rogers et al. 2010), noted that “changes in species relative abundance in an ecosystem will affect interactions in several parts of a food web, and may have an adverse effect on food web status”. Food webs will therefore require that populations of selected food web components occur at levels that are within ranges that will secure their long-term viability (Chesson 2000). The same will apply to key prey species; in the context of support CRA fishery species, thresholds should correspond to the abundance and availability of the support species required to ensure on-going productivity of the CRA fishery species, as well as to abundances necessary for maintaining viable populations of support species. Setting thresholds for CRA fishery species and key prey species is considered in Koops et al. (2013), where potential relationships between productivity and habitat features are considered in detail. Timelines for recoverability from disturbance must be placed in context of the generation time of the prey species. Short-lived organisms often recover swiftly and securely from some forms of perturbation and may even be adapted to naturally disturbed environments. Traits characteristic of short-lived species include high fecundity, small body size, early maturity onset, short generation time, and the ability to disperse offspring widely. However, caution is merited especially for such species because even short periods of poor recruitment can lead to population depletion because spawners are not in the population for long (King and McFarlane 2003).

Connectivity among spatially-structured populations stabilizes metapopulation dynamics, and dispersers, whether juveniles or adults, provide an important ecological function in terms of population rescue when a population is locally extirpated or depleted (e.g., Dolly Varden (Koizumi 2011)). Population recovery depends on the spatial extent and relative connectivity of the aquatic area to be affected (i.e. open or closed system) and the ability of neighbouring metapopulations to recolonize the area should be considered.

We recommend impacts lasting more than one generation time for key prey/forage species to be evaluated as potentially permanent. Depending on the connectivity of prey meta-populations and the severity of the impacts longer impacts may mean there is no residual population to recover following removal of the pressure. This rationale means that the generation time guidance should be considered only a general guideline. The extent of mortality or exclusion from access to key resources associated with the w/u/a, and the availability of potential colonists to the impacted area are all relevant to consider. Ready availability of colonists or partial access and use of the areas throughout the period of the w/u/a could result in the consequences of even long-lasting w/u/a not exerting permanent impacts.

Fisheries reference point methods could be used to identify lower limit reference points for fish species identified as key prey. Such reference points may exist if the prey are themselves part of a managed fishery, (e.g., shrimp, Pacific herring), but for most prey it is anticipated that these would need to be developed. In some cases, existing reference points may need to be adjusted to incorporate the relevance of the managed species as key prey for a CRA fishery species. Reference points may also be set for properties such as growth rates, condition factor, distribution, size structure, or reproductive output of key prey, especially where such factors can be monitored more reliably than the biomass, exploitation rate, and other properties of the key prey species itself.

In some cases, reference points may be developed for an assemblage of key prey which amongst themselves display replacement dynamics. That is, sometimes key prey may vary between two or three species over a period of several years where only one of the prey species is ever abundant at any one time. A reference point approach to the prey should therefore account for the assemblage, recognizing this annual variability but also attempting to keep prey from falling below historically observed minimum abundances.

Bioenergetic models are useful but uncommon tools for extrapolating from stomach contents data to estimate prey consumption through time and inform decisions on thresholds for prey species. For example, bioenergetic models have been used to estimate growth and daily ration for common age and maturity groups of sockeye, chum, pink, and coho salmon caught in the central North Pacific Ocean and Bering Sea in summer. Calorimetry results indicated that small medusae had low caloric density. Copepods, euphausiids, hyperiid amphipods, pteropods, and appendicularians had values in the middle range for caloric density, and squid and fish were calorically dense (Davis and Ishida 1998). Model simulations performed by Davis and Ishida (1998) indicated that salmon were feeding at a rate close to their physiological maximum and any decrease in daily ration could cause significant decreases in growth over a time period as short as two months. Such information can be very useful in establishing thresholds of prey abundance that are needed for sustaining production in CRA fishery species. Similarly bioenergetic or other population modelling techniques have been developed for freshwater fisheries and used to quantify thresholds (e.g., Kitchell 1977). Some have begun to incorporate habitat linkages to population dynamics through bioenergetic connections (e.g., Hayes et al. 2008). (See also discussion in 3.2.1.2 below regarding recovery timescales for ecosystem-level impacts).

Only for the most intensively evaluated projects is it likely that such bioenergetic factors will be explored even semi-quantitatively. For most proponents, using self-assessments or time-limited decisions by field officers, decisions will have to be based on general descriptive information (for example, a key prey species list assembled and made available by the Department) regarding what key prey are likely to be important to CRA fishery species in the general area and the current state of those prey populations in the short term. This is likely to be the case for most systems, until further investment is made in more rigorous management tools to be used at appropriate scales. That information would then be used in the contribution framework described in Section 3 of Koops et al. (2013).

In the short term, a possible follow-up project would be a feasibility study of preparing tabulations or a database of key prey for fish that are part of CRA fisheries, based on literature and existing data sets, on spatial scales of watershed or larger. Existing knowledge on trophic status can be used but gaps will need to be identified for data poor situations where extrapolation is not possible.

3.2 DIRECT SUPPORT FUNCTIONS: STRUCTURE-PROVIDING SPECIES

Biogenic habitats are habitats created by plants and animals. Under the Act, plants are considered as part of fish habitat, whereas biogenic habitat created by animals may also be considered as ‘fish that support’ the CRA fishery species. This may be the organism itself, such as a bed of mussels, or arise from an organism’s activities or skeletons, such as the mounds created by dead corals or sponges. Animal-based biogenic habitats are very diverse in size and structure and occur in both marine and freshwater environments. Similar to some abiotic substrates, these biogenic substrates provide three-dimensional habitats for a large variety of species. Juvenile fish feed on small invertebrates which often are strongly associated with such habitats, either actively seeking refuge from predators or passively being more abundant because of differences in predation rates due to accessibility or other reasons.

In Canadian marine ecosystems there are many types of structure-forming species including corals and sponges, that create such biogenic habitats as forests of deepwater corals, sponge reefs, bryozoan beds, tunicate fields, sea pen fields, polychaete worm reefs, sea grasses, kelp beds, marsh grasses, maerl beds, mussel and oyster beds, amongst others. Freshwater mussel beds in some areas form biogenic habitats.

It is important to establish linkages between the structure providing species and the CRA fishery species. Coldwater marine corals have been shown to directly support marine CRA fishery species. They contribute to vertical relief and increase the availability of microhabitats (Tissot et al. 2006). Increasing complexity provides feeding opportunities for aggregating species, a hiding place from predators, shelter from high flow regimes, a nursery area for juveniles, fish spawning aggregation sites and attachment substrate for fish egg cases and sedentary invertebrates (Reed 2002, Fosså et al. 2002, Etnoyer and Morgan 2003, Etnoyer and Warrenchuk 2007), all of which have been reported for deepwater coral habitats. In general, coral habitats in deep water represent biodiversity hotspots for invertebrates (Reed et al. 1982, Jensen and Frederiksen 1992, Reed 2002, Freiwald et al. 2004, Mortensen and Mortensen 2005), and commonly support a high abundance of fish (Koenig 2001, Husebo et al. 2002, Krieger and Wing 2002, Costello et al. 2005, Tissot et al. 2006). Other marine invertebrates also form complex habitats which support fish production. For example stalked tunicates are often found in groups where they form significant habitat. In the North Pacific, they are known to provide habitat to small juvenile red king crab (*Paralithodes camtschaticus*) (McMurray et al. 1984, Stevens and Kittaka 1998).

CRA fishery species and their prey may depend on different habitats as they pass through different life stages. The types of habitats available, their quality, and their relative size and location, can affect how many fish survive through to the adult population. For juvenile fish especially, there may be ‘habitat bottlenecks’, where the limited availability of specialized habitats may constrain fish production, and ‘habitat chains’, may exist whereby spatially discrete habitats may be connected through ontogenetic shifts in habitat use during the life cycle of a CRA fishery species (or its key prey). When habitat chains are in operation, impacts at one location may influence a CRA fishery species at far distant locations (Rosenfeld and Hatfield 2006). As one example, degradation of nursery habitats may reduce the abundance of distant adult sturgeon stocks in the future (Collins et al. 2000).

There may be unique threats that would apply directly to the key structure-providing species which support CRA fishery species but only indirectly (through habitat impacts), if at all to the CRA fishery species itself. This will be especially true for small scale point source impacts where the sessile habitat-providing species are not able to move and therefore succumb, while the CRA fisheries species are able to move temporarily to avoid the stressor (e.g., by passing through the meshes of fishing gears). Other likely situations involve the different stressors for plants that do not affect fish either at all, or to the same degree. For example, changes to light and nutrient regimes (see Section 3.8 for a general discussion) may impact plants but have no direct impact on the fish. It is also possible for the structure-providing habitat to be affected in a location during the time of year when a CRA fishery species is absent through migratory habits, or for source populations for the habitat to be affected outside of the distribution of the CRA fishery species (e.g., natural or artificial barriers to fish that nevertheless allow dispersal of gametes or larvae to other populations).

3.2.1 Assessing Impacts on Structure-Providing Species

Human activities can damage three-dimensional biogenic habitats and the fish they support, sometimes resulting in the reduction of the spatial extent of these habitats and in some of those cases reducing the productivity of CRA fishery species. For example, strong links have been shown between bryozoan beds and juvenile fishes. Their fragility and exposure above the sea floor makes them susceptible to damage caused by bottom trawling of the seabed. Saxton (1980) and Bradstock and Gordon (1983) recorded the effects of the systematic destruction by trawlers of the bryozoan beds in Tasman Bay, New Zealand, which provided habitat for juvenile snapper and tarakihi which subsequently declined (Saxton 1980). These bryozoan beds had not recovered ten years later and the loss was believed to be permanent (Jones 1992). In coastal and freshwater areas land-based and aquatic stressors include shoreline modification, infilling, water level manipulation, sedimentation, contamination, navigational dredging, contaminants, invasive species and eutrophication arising from excessive nutrient run-off, which can all alter biogenic habitats.

Habitat modification, fragmentation and loss are widely considered to be some of the most serious threats affecting aquatic species (Brinson and Malvarez 2002). Habitat loss is particularly severe in freshwater and coastal ecosystems, where human activities have been historically concentrated. Estuarine and coastal landscapes have been deeply modified and transformed over time such that many of these habitats are already severely degraded. Degradation is, however, difficult to measure because it represents a decrease in condition or quality, not a change in distribution (i.e., habitat loss); nonetheless, indices have been developed to measure the quality or condition of different habitats, especially biogenic ones (e.g., Index of Biological Integrity (IBI); Faunal Index; WQI (Croft and Chow-Frawer 2007); HSI-approaches (Minns et al 2001)). Habitat fragmentation occurs when previously continuous habitats become patchier and can influence the quality of the habitat for the CRA fishery species. Degradation is particularly difficult to measure at the regional, national and multinational levels and cumulative impacts of habitat degradation should be considered when evaluating impacts at the scale relevant to the CRA fishery species (see Section 4.3 below).

As for key prey species (see Section 3.1 above), the link between the CRA fishery species and the biogenic habitat can range from essential to facultative or it may only be a preferred location, and the importance of habitat will vary seasonally and across life history stages. Obligative associations are more common in tropical waters. For example, the coral-dwelling gobies of the genus *Gobiodon* exhibit an obligate association with branching corals from the family Acroporidae such that the population dynamics of the gobies and the corals are closely linked. In temperate waters, biogenic habitat associations may be less specialized but still important in supporting the productivity of CRA fishery species.

The spatial scale of any potential damage to such habitats should be considered in light of the local distribution of that habitat, the ability of the habitat to recover, and the strength and nature of the association of the habitat with the CRA fishery species and supporting species, noting that some biogenic habitats have higher ‘values’ than others, depending on how well they provide for the needs of associated species. For example, juvenile fish tend to congregate more densely in subtidal seagrass meadows than over horse mussel beds, though both habitats play important nursery roles. Further, habitats formed by structure-providing species may be formed by a single species (e.g., marine mussels, *Mytilus edulis* in marine coastal areas) or by species assemblages (e.g., deep sea sponge communities). Assessing impacts on habitats formed by multiple structure-providing species will require consideration of impact and recovery trajectories for all component species and an assessment of whether impacts that differentially impact one such species over another would change the capacity of the community to support production of the CRA fishery species.

3.2.1.1 Operational Implications

Structure-forming species will rarely result in new areas being included under the Fishery Protection Provisions of the Act, because the relevant habitat provided by structure-providing species would be used by some life stage of the fish that are part of a CRA fishery. However, the functions of structure-forming species in relation to CRA fishery species should be considered on a case by case basis for each w/u/a.

It is important to assess whether structure-providing species may depend on source populations in areas never frequented by the CRA fishery species itself. Such areas could be included for application of the Act, if the best available information suggests that the viability of the population of structure-providing species is dependent on migration of individuals (or their reproductive products) from populations that fall outside the range of the CRA fishery species (i.e., source areas).

3.2.1.2 Information Required for Performing Risk Assessments

Risk assessments can be conducted at regional and sub-regional scales to assess whether an activity will cause serious harm to fish in CRA fisheries. The ICES-JRC Task 6 report (Annex 2; Rice et al. (2010)) drawing on FAO (2009) criteria, identified six elements required to perform such assessments (Annex 2):

i. the intensity or severity of the impact(s) at the specific site being affected;

ii. the spatial extent of the impact(s) relative to the availability of the habitat type affected;

iii. the sensitivity/vulnerability *vs*. the resilience of the area to the impact;

iv. the ability of the area to recover from harm, and the rate of such recovery;

v. the extent to which ecosystem functions may be altered by the impact (and their consequences); and,

vi. where relevant, the timing and duration of the impact relative to the times when the area serves particular functions in the ecosystem (shelter, feeding, etc) (FAO 2009).

Also important to consider:

vii. Whether the abundance or availability of a structure-providing species limits survival or growth of one or more life history stages (i.e. is a limiting factor), or limits reproduction in adults;

viii. What would the population-level impacts of altering the amount, quality or integrity of the structure-providing species be on the CRA fishery species?

Fortunately, for most of the key structure-providing habitats that would be encountered in Canada, there is a good body of knowledge on their role, life-history and in some cases recoverability or offsetting requirements. This is because their importance in supporting fish populations is widely recognized. While data may not be available for the precise location of the activity, it is likely that data are readily available from comparable systems to assist in assessment. As for Key Prey Species (Section 3.1 above) it is important to view alterations due to human activities in light of natural variability in the system. Adopting a precautionary approach to managing structure-providing species would help ensure resilience to additional stressors (i.e., invasive species, climate change, pollution etc.) and cumulative impacts.

For follow-up activities it should be feasible, on scales of watershed or groups of similar watersheds, and for intermediate scales of marine and coastal areas, to prepare tabulations of the considerations listed above based on literature and expert knowledge. Work already done on Pathways of Effects should provide the basis for operational guidance on element i) for major types of w/u/a (e.g., Clarke et al. 2008). Then the self-assessments or time-limited evaluations by field officers would only have to evaluate project specific elements (ii and vi).

3.2.1.3 Development of Threshold Levels

In order to establish threshold levels for structure providing species, it is necessary to determine when damage is temporary or permanent. Jones and Schmitz (2009) tested the prediction of irreparable harm to ecosystems using a synthesis of recovery times (return to pre-disturbance state) compiled from 240 independent studies reported in the scientific literature. They found an equal likelihood of recovery or not for all variables, meaning that permanent damage was done in about 50% of the studies. In those studies that documented recovery, Jones and Schmitz (2009) showed that in aquatic ecosystems globally, recovery times were on the order of twenty years or less and that the aquatic plant communities were slower to recover than the animal communities. Ecosystem functions (defined as above), although only measured by a few variables, recovered rapidly in brackish and benthic marine systems and slowest in freshwater benthic systems. When all categories were considered, forests took longest to recover, while aquatic systems required less recovery time than terrestrial systems. Trawling on aquatic plant communities required more than a century to recovery (Figure 3.2.1.3.1). The slowest recovery times were found when multiple stressors were involved (Jones and Schmitz 2009).

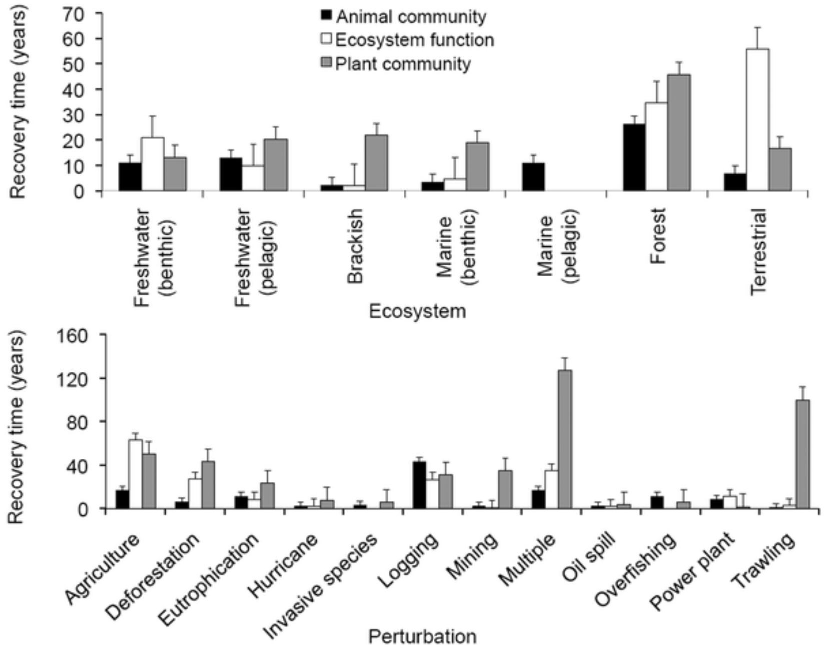


Figure 3.2.1.3.1. Average recovery times across ecosystems (top) and perturbation type (bottom). Variables are separated by animal community (black), ecosystem function (white) and plant community (gray) types. Bars represent mean ± one standard error. Extracted from Jones and Schmitz (2009; their Figure 2).

The FAO Deep-Sea Fisheries Guidelines (FAO 2009; Articles 19 and 20) in the context of damage to vulnerable marine ecosystems caused by bottom fishing, provide a definition of "temporary impacts" that we feel is useful in evaluating whether serious harm has occurred:

> "19. Temporary impacts are those that are limited in duration and that allow the particular ecosystem to recover over an acceptable time frame. Such time frames should be decided on a case-by-case basis and should be in the order of 5-20 years, taking into account the specific features of the populations and ecosystems.
>
> 20. In determining whether an impact is temporary, both the duration and the frequency at which an impact is repeated should be considered. If the interval between the expected disturbances of a habitat is shorter than the recovery time, the impact should be considered more than temporary. In circumstances of limited information, States and RFMO/As should apply the precautionary approach in their determinations regarding the nature and duration of impacts." (FAO 2009).

However, as discussed by Randall et al. (2013) and Koops et al. (2013), the definition of temporary must be placed more directly in context of the generation time of the species forming the habitat. In the case of the FAO Deep-sea guidelines the 5-20 years applied to organisms with life-spans of decades to centuries. Short-lived species, being highly fecund, may have the ability to increase rapidly under good environmental conditions; however, some short-lived species do not have this capability which should be considered when applying definitions of permanence. Recovery also depends on the spatial extent and relative connectivity of the aquatic area to be affected (i.e. open or closed system) and the ability to recolonize the area (itself a function of disturbance size and reproductive traits of the structure-providing species) should be considered.

As a general guideline, effects lasting greater than one generation for longer-lived species (> 5 years) that are key structure-providing species for CRA fishery species, should be considered permanent. We also support the guidance of the FAO (2009): “If the interval between the expected disturbance[s] of a habitat is shorter than the recovery time, the impact should be considered more than temporary.”

We recommend impacts lasting more than one generation time for key structure-providing species to be considered permanent, because longer impacts may mean there is no residual population to recover following removal of the pressure. This rationale means that the generation time guidance should be considered only as a general guideline. The extent of mortality or exclusion from access associated with the w/u/a, and the availability of potential colonists to the impacted area are both relevant considerations. Ready availability of colonists or partial access and use of the areas throughout the period of the w/u/a could result in the consequences of even long-duration w/u/a not being permanent.

The recovery trajectories of structure-providing species will vary according to the disturbance regime, as well as life history characteristics of the species, their abundance and spatial configuration. As noted in Section 4.4 below, most anticipated activities occur at local spatial scales and their impacts will also be felt at those same scales (but see Sections 4.2, 4.3). Table 3.2.1.3.1 provides a few examples of biogenic structure-forming species ranked according to their ability to recover from perturbations. It provides details of some of the characteristics the species that influence its ability to recover.

Table 3.2.1.3.1. Examples of Biogenic Habitats which May Provide Support for CRA Fishery Species Ranked by their Relative Ability to Recover from Perturbations.).

Structure-providing Species Categories	Characteristics of Structure-providing Species	Examples	Relative Recovery Potential	Comments
Mussel Beds	Annual reproduction; rapid growth rates; moderate life spans.	Blue mussels (*Mytilus* spp.)	High	Likely not of concern in marine environments.
Submerged Semi-infaunal Mussel Reefs/Bioherms	Massive, nearly continuous beds often forming true bioherm mounds. Habitat is created by interaction between biotic and abiotic components.	Horse Mussel Beds (*Modiolus* spp.), Oyster reefs	Low	Recoverability of habitat is low due to long time scales to create bioherms. The bioherms described in the Bay of Fundy have low densities of mussels (4-78 / m^2) but are frequently up to 3m high. Subject to non-linear responses due to erosion of habitat.
Deep-sea Sponges	Very long-lived (decades), slow growing; episodic recruitment; low recruitment; short distance dispersal.	Glass sponges (*Vazella pourtalesii*) M; Round massive sponges (*Geodia* spp.) M; Hexactinellid sponge reefs in Pacific Region.	Low	Considered components of Vulnerable Marine Ecosystems with poor recovery trajectories.
Deep-sea Corals	Very long-lived (decades to centuries), slow growing; episodic recruitment; low recruitment; short distance dispersal.	Bubblegum coral (*Paragorgia* spp.) M; Sea corn (*Primnoa resedaeformis*) M; Bamboo coral (*Keratoisis* spp.) M.	Low	Considered Vulnerable Marine Ecosystems with poor recovery trajectories.

3.2.1.4 <u>Determination of the Percentage of Structure-Providing Biogenic Habitats Necessary to Maintain Ecosystem Function and Productivity of CRA Fishery Species and Supporting Species</u>

In order to establish thresholds for determining key transition points on an axis of habitat change, there are two overarching questions to be addressed:

1. How much and what spatial arrangement of the structure-providing species are needed to maintain the ecosystem functions that support CRA fishery species?

2. How much and what spatial arrangement of the structure-providing species must be maintained if the structure-providing species is to sustain itself (and thus provide ecosystem services to other species)?

The second overarching question has received far less attention, though there has been some discussion of whether major aggregations are self-sustaining or dependent on larval or seed supply from smaller patches upstream.

Future work could be undertaken linking thresholds directly to operational objectives for managing CRA fishery species with respect to these two questions. While there is no substitution for rigorous empirical data coupled with statistical modelling for defining such relationships, we may be able to use simple rules of thumb developed in relation to “critical habitat” for sustaining viable populations in combination with population distribution models (e.g., minimum area for population viability (MAPV) (Vélez-Espino and Koops 2008a, 2008b, 2008c), or we can use more quantitative habitat-based population dynamics models to set some thresholds for more data-rich species. Regardless of the approach(es) we identify here the need for setting thresholds and stress that the thresholds and methods used need to be logically linked to some clearly articulated operational objectives.

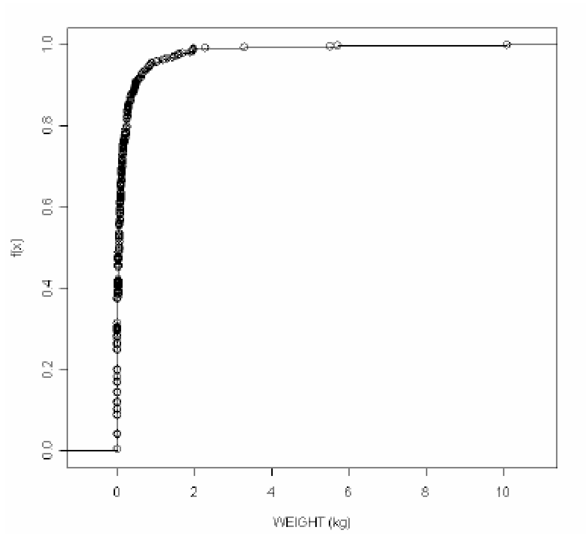
3.2.1.5 <u>Identifying Habitats of Structure-Providing Species</u>

In conducting assessments of the impacts of human activities on structure-providing species that support CRA fishery species, it may be important to distinguish between the presence of individuals of a species and the presence of the biogenic habitats. Two quantitative methods have emerged through the work of the North Atlantic Fisheries Organization (NAFO) to identify significant concentrations of corals and sponges which could be applicable to other ecosystems where similar (trawl survey) data were available. These methods are most suited to highly gregarious species.

i) Use of Cumulative Catch Curves

For sea pens, small gorgonian corals and large gorgonian corals NAFO used the cumulative catch distributions from research vessel catches to select thresholds for mapping significant concentrations. For each group, the majority of catches were small, with only a few very large catches (Figure 3.2.1.5.1). The location of those catches were then mapped and used to identify the biogenic habitats. There is no strong biological basis for selection of threshold values but NAFO considered that the 97.5% quantile (upper 2.5% of catches) was an appropriate level for sea pens and small gorgonians, while the 90% quantile (upper 10% of catches) was chosen for large gorgonians. The lower quantile for the large gorgonians was justified on the basis that these organisms are easily broken and many of the smaller catches may be composed of broken larger colonies. Selection of a lower threshold was considered precautionary for that conservation unit (NAFO 2008). Catches less than these thresholds were considered to be from outside the biogenic habitat (fields, grounds etc.) formed by the aggregations of individuals. However, no information is available to determine if impacts on concentrations of biogenic features below the 97.5% (or 90%) benchmarks might result in reductions of productivity of fish that are part of CRA fisheries. This method has potential to provide the thresholds if a link can be made between fish production and habitat area. Substantial work in freshwater systems suggests that productivity begins to decline when between 10 and 15% of typical used habitat has been lost (see Koops et al. 2013 for examples from a number of studies).

Using this approach where data are available would allow for an operational definition for the identification of biogenic habitat of aggregating species using research vessels. These areas can then be mapped and considered to be critical fish biogenic habitat. This has already been done for sea pens, small gorgonian corals and large gorgonian corals as well as sponge grounds in the marine waters of Atlantic Canada and the Eastern Arctic (Kenchington et al. 2010).



Weight Quantiles

100.0 %	maximum	10.116000
99.5 %		4.718880
97.5 %		**1.602200**
90.0 %		0.484600
75.0 %	quartile	0.170000
50.0 %	median	0.049000
25.0 %	quartile	0.010500
10.0 %		0.003094
2.5 %		0.001000
0.5 %		0.001000
0.0 %	minimum	0.000100

Figure 3.2.1.5.1. Cumulative catch distribution of sea pens collected in research vessel surveys in the NRA with associated weight quantiles (extracted from NAFO 2008).

ii) Quantitative Assessment of Patch Size

NAFO developed a spatial approach to identify significant concentrations or patches of sponge, that is, sponge grounds (Kenchington et al. 2009, NAFO 2009). Essentially, the research vessel data on sponge bycatch were used to create a biomass map (kg/km^2). A kernel density function was used to interpolate the data as this method is superior for identifying concentrations (over other smoothing methods such as Kriging or IDW). The biomass raster is then contoured by areas of equal or greater bycatch, creating polygons for each bycatch “threshold”.

This method worked well because the sponges not only had a catch distribution as described above, with few medium-sized catches, many small ones and few large ones, but the location of these larger catches were highly aggregated. These two properties allowed for the identification of sponge grounds by comparing the relative increase in area with increasing bycatch weight threshold. The area occupied by the largest catches did not increase very much as smaller catches were included - up to a point (Figure 3.2.1.5.2). Once the catches were outside of the sponge ground the area expanded rapidly. Through evaluation of the performance of this technique, confidence was gained in selecting the threshold that best defined the sponge grounds for these *Geodia* sponges and their associated sponge fauna (NAFO 2009). The area of the polygon encompassing the sponge grounds is an estimate of sponge habitat, which gives this approach a more direct biological basis for selection of thresholds. Further, Kenchington et al. (2012) proposed a number of state indicators for monitoring coral and sponges in the Arctic based on their spatial configuration, that is, the spatial properties and arrangement, position, or orientation of habitat patches within the broader survey area.

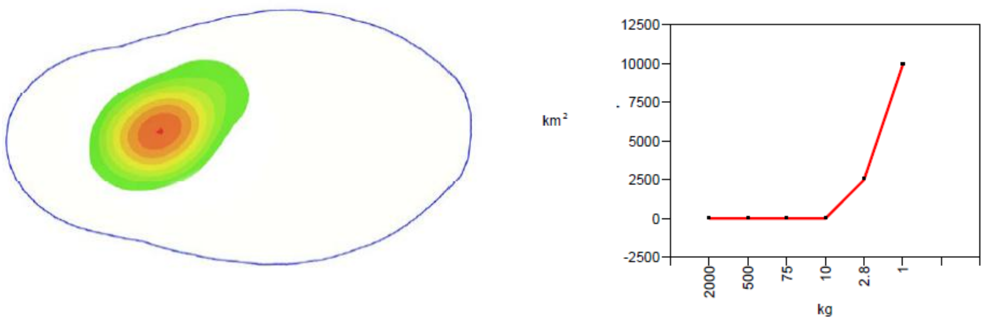


Figure 3.2.1.5.2. Relationship between area occupied and catch biomass used to identify structure forming biogenic habitats in the NAFO regulatory area of the high seas.

This approach is scientifically rigorous, but requires good spatially resolved trawl survey data and a large investment of time by experts in spatial statistics and computer mapping technique, and availability of geo-referenced habitat data from the area of concern. Therefore its application is likely to be restricted to high priority structure-forming species in areas where surveys have been conducted. Predictive habitat modelling can extend the applicability of the approach to unsurveyed areas where basic data on habitat features such as depth, temperature regime, substrate etc are available, and described (Boutillier et al. 2010, DFO 2010). However such methods are again labour intensive, and the resultant maps are often both coarse scale (larger than the habitat patches) and need some calibration and verification when applied to new areas. Predictive habitat models have been developed for both marine (e.g., DFO 2010) and freshwater habitat features (e.g., Minns et al. 1999, Mortsch et al. 2006). Fine to medium-scale models applied to freshwater plant community types (i.e. not species level predictions) used generalized percent cover or presence estimation (Narumalani et al 1997, Seifried 2002, Leisti et al. 2012). This may be adequate enough resolution for habitat assessments. Such models could be enhanced through comparisons with predictive models for fish species distributions (Porter et al. 2000) to identify habitat associations. For many small-scale projects in freshwater and/or coastal areas the proponent self-evaluation or evaluation by a field officer will have to depend on simple visual inspection of the extent of aquatic vegetation present in the area of the w/u/a with simplified model to be used for post-impact assessment.

3.3 INDIRECT SUPPORT FUNCTIONS - KEYSTONE SPECIES

Keystone species are those which have an impact on ecosystem structure and functioning disproportionate to their biomass or abundance in the ecosystem (Paine 1966, 1995). The criteria for a keystone species are that the species exerts top-down influence on lower trophic levels and prevents species at lower trophic levels from monopolizing critical resources, such as competition for space or key producer food sources. They maintain community diversity by preying selectively on competitively superior prey taxa, thereby preventing the exclusion of relatively weak competitors. However, keystone species may not act as controlling agents in all parts of their distribution under all environmental conditions (Power et al. 1996).

The initial concept of the keystone species was developed on the basis of observations from rocky intertidal communities on the Pacific coast of the USA and was described in terms of a two species interaction between starfish and mussels (Paine 1966). Mussels have huge colonization potential through their fecundity and pelagic larval dispersal via currents but they are space limited at settlement. Mussels left unhindered by predators can dominate the intertidal zone to the exclusion of most other sessile plants and animals. The result is decreased diversity, resilience and production all of which can be important for other species, including CRA fishery species. Starfish in this system control the mussels' spatial domination through predation and thus open up areas for colonization by other species. Removal of starfish from the system results in a mussel dominant zone with lower diversity.

An example in Canadian waters occurs on the BC coast. Sea otters eat sea urchins which are grazers, feeding on kelp. If predation rates are reduced on urchins (e.g. through reduction in sea otter abundance), urchin abundance increases and populations have the potential to significantly reduce the abundance of kelp forest by eating through holdfasts by which kelp attach to the sea floor. Urchins are keystone species for their impact on the ecosystem brought about through their grazing. However, they are top-down controlled by the sea otter which feeds on the urchins, maintaining populations at low numbers which prevents them from removing all of the kelp. Therefore, when the sea otter is present it is the keystone species in the system, replacing the sea urchin. The carrying capacity (maximum biomass in absence of exploitation) for sea otters on the BC coast has been estimated at between at only about 12,000 and 60,000 animals (Gregr et al 2008) yet their presence or absence considerably changes the composition of the BC nearshore community (Watson and Estes 2011). Recently there has been an increase in predation on the sea otter by the killer whale which has led to a sharp decline in the numbers of sea otters, an increase in the numbers of sea urchins, and a decline in the kelp. The killer whale has emerged as the new keystone species in the system of kelp forest, sea urchins, sea otters and killer whales in some areas.

Power et al. (1996) lists species that they believe meet the criteria for being keystone species in marine and freshwater habitats (Table 3.3.1). The definition of keystone used by Power et al. (1996) in creating their table would appear to be more inclusive than the original definition of Paine (1966) and the definition we have used here. We have not updated their list but reproduce the aquatic examples in order to provide examples of the types and numbers of keystone species that have been identified. Note that the keystone species may be a CRA fishery species itself (e.g., steelhead, *Oncorhynchus mykiss*). In future it would be possible to update this list based on scientific advice from experts on different ecosystems across Canada.

Table 3.3.1. Non-exhaustive list of examples of demonstrated keystone species found in aquatic environments and the target species affected (from Power et al. 1996).

Ecosystem	Keystone species	Target of direct effect
Marine		
Rocky Intertidal	*Pisaster ochraceus* (starfish)	mussels
	Nucella lapillus (whelk)	mussels
	Haematopus spp. (black oystercatcher)	limpets
	Concholepas concholepas (whelk)	mussels
Rocky Subtidal	*Enhydra lutris* (sea otter)	sea urchins
Pelagic	*Balaenoptera* spp. (baleen whales)	krill
	Theragra chalcogramma (walleye pollock)	zooplankton, small fish
Coral Reef	*Diadema antillarum* (sea urchin)	seaweeds
	Acanthaster plauci (starfish)	corals
	Stegastes fasciolatus (damselfish)	schooling parrotfish and surgeonfish
Soft Sediments	*Urolophos halleri* (ray)	amphipods
	Myhobatis californica (ray)	amphipods
	Eschrichtius robusta (gray whales)	amphipod mats
	Enhydra lutris (sea otter)	bivalves
Fresh Water		
Lakes and Ponds	*Alosa pseudoharengus* (fish)	zooplankton
	Cichla ocellaris (fish)	fish
	Micropterus salmoides (fish)	fish
	Notophthalmus viridescens (salamander)	anuran tadpoles
Rivers and Streams	*Micropterus salmoides* (fish)	algivorous minnows
	M. punctatus	algivorous minnows
	Castor canadensis (beaver)	trees
	Oncorhynchus mykiss (trout)	benthic invertebrates; anuran larvae
	O. mykiss (steelhead)	invertebrates and fish fry
	Hesperoleucas symmetricus (minnow)	invertebrates and fish fry

The definition of keystone provided here conforms mostly to the early empirical definitions. Species that might be deemed “important” and therefore “key” are covered elsewhere (prey, structure providers, highly-connected, apex predators etc.) and do not need to be included here. We also exclude ubiquitous species such as some bacteria that provide certain critical chemical environments. Essentially a keystone species is likely to be a mid- to upper-trophic-level species that plays an identifiable and key role of support in terms of preventing out-competing or over-domination of a CRA fishery species. Keystone species are usually not dominant in biomass and not so important in the direct flow of energy from lower to higher trophic levels.

Identifying keystone species can be problematic and attempts to develop a set of species traits that would identify keystone interactions have proved elusive. A variety of approaches have been used, either singly or in combination, including experimental manipulations, modeling approaches, comparative studies and natural experiments. Experimental removal is the most convincing but is frequently impractical for logistical, technical, social and/or ethical reasons.

3.3.1 Assessing Impacts on Keystone Species

Power et al. (1996) recognized the need for an operational definition of a keystone species and devised a measure of community importance (CI) which is the change in a community or ecosystem trait (e.g., productivity, nutrient recycling, species richness or the abundance of one or more functional groups of species or of dominant species) per unit change in the abundance of the species. A keystone species would have very large community importance values, and the effect on the ecosystem would also have to be greater than that of natural variation. Jordán et al. (1999) produced a “keystone index” applicable only to food webs and cases of trophic interactions.

A species meeting the keystone criteria should be well-protected from anthropogenic impacts because of potential amplification of the impacts due to their loss and subsequent cascading effects. Local impacts of keystone loss are likely the first to be observed quickly. For example in the sea otter case, removal of sea otters from a bay could rather quickly lead to increased urchin grazing and abundance and then loss of kelp beds. The loss of the kelp beds could then lead to a cascading loss of almost anything that depends on kelp beds. So for example, young salmonids or groundfish that may take refuge in kelp beds would be more exposed to predation leading to higher early mortality in the CRA fishery species. Depending on the degree of population connectivity through migration and reproduction, local keystone loss could spread to larger areas relatively quickly. Even though the removal of keystone species is expected to cause dramatic changes to the ecosystem, it does not necessarily follow that CRA fishery species will be affected. It is also important to consider:

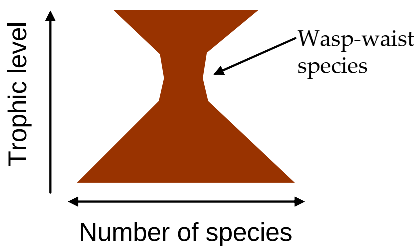
i) Whether the abundance or availability of a keystone species limits survival or growth of one or more life history stages (i.e. is a limiting factor), or limits reproduction in adults of CRA fishery species;

ii) What would the population-level impacts of altering the abundance, availability of the keystone species be on the CRA fishery species?

3.3.2 Operational Implications

Given the relative rarity of keystone species as defined here, but recognizing the importance of such species in ecosystems where they occur, we suggest that if keystone species are expected to occur and function as keystones in the ecosystem occupied by CRA fishery species and that the presence of the keystone has an impact on the CRA fishery as per i) and ii), then the abundance of the keystone species must be maintained at levels where the control function it provides to the system can be expressed effectively for the benefit of the CRA fishery species. Potential keystone species can be identified using the criteria in Section 3.3 and further assessed using methods discussed in 3.3.1 with the strongest evidence produced by experimental manipulations. Species identified as being keystone in other ecosystems (Table 3.3.1 and others) have a high probability of having keystone roles in equivalent unstudied systems; congeneric species of known keystones should also be evaluated given shared traits at that level of organization.

3.4 INDIRECT SUPPORT FUNCTIONS - WASP-WAIST SPECIES



In a "wasp-waist" ecosystem, an intermediate trophic level (usually a fish species) is expected to control the abundance of predators through a bottom-up interaction and the abundance of prey through a top-down interaction. Such ecosystems are commonly associated with marine upwelling areas (e.g., Cury et al. 2000) but they occur in many other areas as well. Wasp-waist species usually exist at a trophic level where they are one of only a few species that are transferring the energy and as such they live within this constriction (hence "wasp-waist") in the energy flow where there is little functional redundancy in the form of other species that can do the same task. They experience high predation mortality. Bakun (2006) notes that in marine systems these species can have life histories that result in radical variability that may propagate to both higher and lower trophic levels of the ecosystem (e.g., the well-known fluctuations in anchovies and sardines). In addition, Bakun suggests that populations of these species have two key attributes: "(1) they represent the lowest trophic level that is [actively] mobile, so they are capable of relocating their area of operation according to their own internal dynamics; (2) they may prey upon the early life stages of their predators, forming an unstable feedback loop in the trophic system that may, for example, precipitate abrupt regime shifts".

Wasp-waist species are usually CRA fishery species themselves or else direct prey of CRA fishery species and thus support those fisheries on that basis alone; CRA fisheries may also operate below the "waist". Less frequently, however, a wasp-waist species may not be a direct prey of a CRA fishery species yet the continued production of the CRA fishery species still will have an important dependence on the wasp-waist species.

Identifying wasp-waist species is usually not difficult because they are prevalent in most surveys or censuses, thus they are a dominant species in their trophic-level or size category. They are often also common in predator stomachs, when diet data are collected. Moreover if their abundance is highly variable due to oceanographic conditions, which is the case for many of the small pelagic species of this type, this variation is often present and amplified in the diet data as well. Predators focus on them when they are abundant, and switch largely to alternative diets when the wasp-waist species are at low abundance.

Wasp-waist species are often small pelagic species which eat zooplankton and in turn are eaten by larger predators; anchovies and sardines are the classic models however there are other important wasp-waist species. A well studied northern temperate marine example of a wasp-waist species is sandeel (*Ammodytes*) in the North Sea. Sandeels there are important predators of small crustaceans and are themselves important prey of most North Sea demersal fish and seabirds (Frederickson et al. 2007). Capelin is another such example of a wasp-waist species in northeastern Canada. One of the hypotheses for lack of recovery for Atlantic cod off Newfoundland is that capelin abundance was too low around the collapse period to support a recovery of cod (Rose and O'Driscoll 2002). An example of wasp-waist species displaying the predation on juveniles of their predators is herring in the southern Gulf of St. Lawrence (sGSL) (Swain et al. 2000). In that system, when herring are abundant, they can exert a large predation mortality on cod juveniles, while adult cod are predators of the herring. The result is that cod recruitment and recovery potential in sGSL is strongly dependent on the state of herring populations.

Wasp-waist systems appear to be uncommon in freshwater systems compared to marine systems. Wrona et al. (2005) discuss top-down and bottom-up control with respect to Arctic freshwater ecosystems but do not mention wasp-waist systems. Several examples of top-down control in freshwater systems are provided under the Apex Predator function below (Section 3.6). The only example of wasp-waist control that we could find was in the limnetic communities of Laguna Galarza and Laguna Iberá, Argentina. There, a wasp-waist food chain operates in two shallow sub-tropical lakes in which the zooplanktonic trophic position, in the middle, has crucial importance in defining the trophic control (Cózar et al. 2003). However, it is important to distinguish a species or restricted assemblage definition of the wasp-waist role from a classic trophic level role, e.g. the entire zooplankton trophic level. Wasp-waists should be defined by one or at maximum a few species. Broader grouping of species would suggest that key prey or highly-connected species may be more suitable functional definitions.

The wasp-waist function is most likely to be found in large systems where there are four or more trophic levels of secondary producers. In Canada, these kinds of systems are almost exclusively marine.

Wasp-waist species such as capelin and herring are recognized by DFO as important forage species. The DFO Policy on New Fisheries for Forage Species defines forage species as:

> "A forage species is a species which is below the top of an aquatic food chain, is an important source of food for at least some predators, and experiences high predation mortality. From the perspective of fisheries management, the species will fully recruit to the fishery at ages which still experience high mortality due to predation. Forage species often undergo large natural fluctuations in abundance in response to environmental factors, on time scales comparable to or shorter than a generation. Forage species also usually form dense schools for at least a part of the annual cycle, are relatively short lived and have a coastal distribution for at least a part of the year." (DFO: http://www.dfo-mpo.gc.ca/fm-gp/peches-fisheries/fish-ren-peche/sff-cpd/forage-eng.htm).

This definition does not capture the energy flow aspects of wasp-waist species explicitly, but fish which qualify as forage species may also be wasp-waist species in our context. A forage species would not qualify as a wasp-waist species if its function could be replaced by a number of other species in the ecosystem at the same trophic level.

3.4.1 Assessing Impacts on Wasp-Waist Species

Marine fisheries exploiting the wasp-waist species and occurring at intermediate trophic levels, have a potential disrupting effect on the stability of marine ecosystems (Vasconcellos et al. 1997) similar to those of highly-connected species, which could affect other CRA fisheries and their support systems. Because wasp-waist species tend to be quite abundant and their dynamics often variable, wasp-waist species can often withstand a certain amount of perturbation that will not unduly increase their variability and perturb CRA fishery species dynamics. There can be local effects however, that are quite important. In the North Sea, seabird colony health requires that sandeel banks are not too far from the colonies so that energetic expenditures of the sea birds are not so great to get the sandeels that there is no net energetic gain through predation. In response to studies linking low sandeel availability to poor breeding success of kittiwake, all commercial fishing in the Firth of Forth area off Scotland has been prohibited since 2000, except for stock monitoring purposes (ICES 2010).

The ratios of pelagic:demersal catch and biomass have been used successfully to track changes and detect collapses in small pelagic fish such as wasp-waist species (Shannon et al. 2009). The trophic indicators reviewed by Cury et al. (2005) and discussed under the Highly-Connected Species support function below could also be useful here. Food web models, where they exist or can be readily developed, can be used with environmental data to simulate conditions that would cause collapse of wasp-waist species.

As for other support functions, it is important to first identify whether the function is operative in the system under consideration, and then to ask:

i) Whether the abundance or availability of a wasp-waist species limits survival or growth of one or more life history stages (i.e. is a limiting factor), or limits reproduction in adults of CRA fishery species;

ii) What would the population-level impacts of altering the abundance or availability of the wasp-waist species be on the CRA fishery species?

3.4.2 Operational Implications

Wasp-waist species are ecologically important and can also have naturally fluctuating abundance which in most cases would be greater than perturbations caused by any small scale impact of a w/u/a. Given that examples from freshwater systems are rare, such impacts on open systems may be even less likely to affect CRA fishery species. However, attention should be given to *populations* of wasp-waist species that might have stronger links to other trophic levels (e.g., nearshore populations and birds). Because of their importance in the food web, w/u/a that have the potential to impact wasp-waist species should ensure that local depletion of populations do not occur. Local depletion of the wasp-waist species could result in food shortage for the dependent predators, even if the overall mortality of the wasp-waist species was sustainable.

Where the wasp-waist species are also CRA fishery species then the abundance of the wasp-waist species must be maintained at levels which allow the energy transfer function to be maintained, using fisheries reference points for managed stocks. Because year-class strengths of wasp-waist species can vary greatly, and sizes of year-classes often are correlated over several years, a fixed exploitation rate is unlikely to ensure that adequate spawning biomass will always be protected. Maintaining a minimum spawning biomass is also likely to be necessary as a management strategy. Human predators should be considered with all other predators in assessing cumulative impacts on the stocks. Therefore, the biomasses of wasp-waist species used as limit reference points in management should ensure both that future recruitment of the target species is not impaired, and that food supply for predators is not depleted.

For wasp-waist species that are prey of CRA fishery species then they should be treated under the considerations outlined for Key Prey and Forage Species in 3.1 above.

3.5 INDIRECT SUPPORT FUNCTIONS - APEX PREDATORS

Apex predators occupy the top trophic position in a community; these are often large-bodied and specialized hunters that have no predators of their own within their ecosystems. They can have a controlling influence on the structure of lower trophic levels, referred to as “top-down” control. When there is a reduction in the abundance of apex predators there can be an increase in mesopredators (smaller predators occupying trophic positions below apex predators), which causes a decline in prey populations. Apex predators can keep the smaller predators in check, keeping important prey populations abundant and maintaining healthy populations by removing weak individuals from prey populations (Ritchie and Johnson 2009, Baum and Worm 2009). By preventing one species from monopolizing a limited resource, predators increase the species diversity of the ecosystem, increasing the stability of an ecosystem.

Removal of apex predators from an ecosystem can cause trophic cascades (see glossary). In marine ecosystems trophic cascades can lead to alternative ecosystem states (Salomon et al. 2010). Similar effects may occur for Arctic freshwater food webs many of which show evidence of top-down control (Hershey 1990, Goyke and Hershey 1992, Hanson et al. 1992, O’Brien et al. 1992, Jeppesen et al. 2003). Estes et al. (2011) suggest that removal of apex predators may have additional unanticipated impacts of trophic cascades on processes as diverse as the dynamics of disease, carbon sequestration, invasive species, and biogeochemical cycles.

In the marine environment, apex predators include sharks, killer whales, sperm whales and polar bears, while in freshwater, lake trout, large-mouthed bass, walleye, northern pike, muskellunge, smallmouth bass, burbot, snapping turtles, bears and fish-eating birds, amongst others, can be apex predators. Some apex predators have complex social structures (e.g., toothed whales), social hunting and display group or kin selection types of behaviour and therefore group dynamics can be as or more important as abundance, and may need to be considered when evaluating their support role for CRA fishery species. Most apex predators that are fish (as defined in the Act) are likely to also be CRA fishery species.

3.5.1 Assessing Impacts on Apex Predators

Richie and Johnson (2009) conducted a review of 94 studies of the effects of vertebrate apex predators on mesopredators and prey communities in terrestrial and marine ecosystems (with one freshwater example). In all ecosystems changes in the abundance of apex predators had disproportionate (up to fourfold greater) effects on mesopredator abundance, although outcomes of interactions between apex predators and mesopredators depended on resource availability, habitat complexity and the diversity of predator communities.

Apex predators are less abundant than lower trophic level species and often migrate large distances relative to the scale of ecosystem. In freshwater environments movement may be constrained by landscape topography, but apex predators would still often be among the most mobile species in the ecosystem. There are a number of well-established methods for estimating the abundance of apex predators. Visual flight surveys have been used to quantify the abundance of polar bears, large sharks, and whale species. Acoustic techniques which record underwater sounds made by the animals have been used to complement visual efforts and to assess population size of sperm whales (Leaper et al. 1992). Trends in abundance of most apex predators can be followed using population assessment models, and tracking trends can sometimes be improved if the models consider habitat/environmental influences. However it is important to note that population size might be highly influenced by social structure and behaviour in addition to population density (Ritchie et al. 2012). Monitoring mortality rates of prey populations may be useful in detecting the relative impacts of the apex predators, if the causes of other sources of mortality on the prey are known.

Salomon et al. (2010) note: "Shifts in alternative states can be long lasting and difficult to reverse because the factors driving recovery of a system back to its original state need to be substantially stronger than those causing the initial shift. Although theoretical guidelines exist, detecting these shifts in real ecosystems is not a trivial task. In many cases, what remains unclear is the threshold density of predator reduction that will induce a trophic cascade and associated state shift and what feedback mechanisms cause ecosystem shifts to be irreversible." Table 3.5.1.1 presents a summary extracted from Salomon et al. (2010) that outlines factors that can influence the likelihood of a fisheries-induced trophic cascade to occur in marine ecosystems if apex predators are reduced/removed from the ecosystem. These attributes may be useful to consider when scoping/assessing the likelihood that a trophic cascade could occur in marine ecosystems being evaluated. The trophic indicators reviewed by Cury et al. (2005) and discussed under the Highly-Connected Species support function could also be useful here.

In the context of the new Act it is also necessary to document the impact apex predators have on CRA fishery species by considering:

i) Whether the abundance or availability of an apex predator limits survival or growth of one or more life history stages (i.e. is a limiting factor), or limits reproduction in adults of CRA fishery species;

ii) What would the population-level impacts of altering the abundance or availability of an apex predator species be on the CRA fishery species?

Given the high mobility of apex predators there is potential for impact events to occur outside the distribution of the CRA fishery species but within the same ecosystem, potentially causing impacts to the CRA fishery species.

Table 3.5.1.1. Factors that alter the occurrence and magnitude of fishery-induced trophic cascades by either dissipating or amplifying the transmission of indirect fishing effects throughout marine food webs (Extracted from Table 1 of Salomon et al. 2010).

Context-dependent Factors	Effect	Explanation
Species diversity and trophic complexity	Dissipate	High species diversity facilitates the replacement of overfished species.
Top-down control	Amplify	Marine ecosystems under strong top-down control are less resilient to the exploitation of top predators and more susceptible to fishing induced trophic cascades.
Regional oceanography	Dissipate or amplify	Factor mediates rates of recruitment, primary production, growth and maturation, predation, and herbivory.
Recruitment limitation and variability	Dissipate	Low or sporadic recruitment and intermediate trophic levels can slow the recovery rate of prey following predator depletion.
Local physical disturbances	Dissipate	Physical disturbances can decouple the trophic links between predators, herbivores and primary producers.
Multi-trophic-level fisheries	Dissipate	Prey release that may have occurred following the exploitation of predators is offset by the harvest of prey.
Disease	Dissipate	A dramatic depletion in predators may cause prey to exceed their host density threshold for epidemics.
Predator-avoidance behaviour of prey	Amplify	Because most organisms behave in ways that moderate their exposure to predation risk, often at the cost of reduced food intake, predator depletion can also lead to increased prey foraging effort or efficiency.
Habitat complexity and spatial settings	Dissipate	Predator effectiveness can be dampened by prey-avoidance behaviour and the availability of safe hiding spots.

Fisheries reference points can be used to assess the status of top predators. However, since top predators tend to be long-lived, population changes owing to recruitment failure may not be reflected in the adult populations for many years (Rouse et al. 1997).

3.5.2 Operational Implications

Apex predators in freshwater systems are very often among the preferred species in CRA fisheries, and may already be considered under the Act, as "part of" CRA fisheries. Hence the sustainability of impacts of a w/u/a on an apex predator should already be part of the decision process at each scale.

Existing knowledge should be sufficient to identify apex predators for most aquatic ecosystems of Canada. Information on the actual abundance of specific apex predators in specific areas will not be available. However, because apex predators are important for maintaining top-down control on ecosystem structure impacts on apex predators should be considered given the potential for non-linear (amplified) impacts on the broader ecosystem or risk moving ecosystems into alternate states that may be undesirable for CRA fishery species.

3.6 INDIRECT SUPPORT FUNCTIONS - HIGHLY-CONNECTED SPECIES

The strength and number of species interactions in a food web influence the impact of changes in species abundance. The simple predator/prey relationships discussed in Section 3.1 above, operate within a much more complex network structure, where many additional species and their direct and indirect effects can play important roles for the productivity and stability of local CRA fishery species as well as the stability of the broader community.

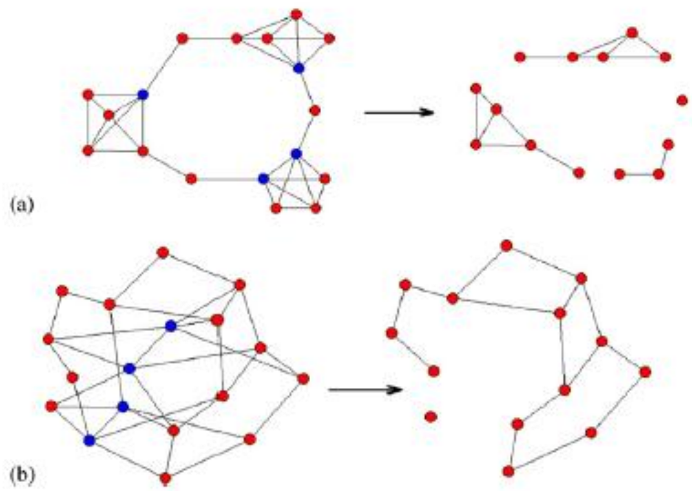


Figure 3.6.1. Schematic illustration of node removal in two networks with identical connectance and node degree distribution (DD). (a) Network formed by three main clusters connected by few links. The four most connected nodes coincide with the bottlenecks (blue), thus their removal collapses the web to several small isolated clusters (right network). (b) Random rewiring of network shown in A, which improves the expansion properties keeping constant the connectance and DD. The removal of the same four hubs does not produce the collapse of the web, which remains as a main cluster containing 72% of original nodes. (extracted from Figure 1 of Estrada 2007).

Food webs have many trophic links between species some of which are strong but most of which are weak and a small number of species show a much higher number of both strong and weak links to other species thus acting as important hubs in the food web network (McCann et al. 1998, Solé and Montoya 2001, Dunne et al. 2002, Koen-Alonso 2009). Food web structure is relatively more robust to the random elimination of species than to the directed removal of the highly connected ones (Solé and Montoya 2001, Dunne et al. 2002). Therefore, highly connected species are inordinately important to preserve the overall ecosystem structure and function on which the productivity of CRA fishery species is based. These highly connected species can play multiple roles in the food web (target of CRA fisheries, key prey, keystone species, prey of key prey of CRA fishery species), but many of these roles are explicitly addressed in other sections of this document.

Food webs with the same level of connectance and average number of links per node (degree distribution) can have fundamentally different architectures and vulnerability to collapse (Figure 3.6.1; Estrada 2007). This aspect of structure has been captured by another metric known as the "expansion properties" of the system. A system with good expansion has no bottlenecks (often wasp-waist species) and is less vulnerable to collapse (Figure 3.6.1).

Food chain length is an important characteristic of ecological communities as it influences community composition and function and the concentration of contaminants in higher trophic levels such as apex predators which may be CRA fishery species (Post et al. 2000). There is a positive relationship between lake size and food-chain length (Figures 3.6.2, 3.6.3) unrelated to productivity. This generally equates to lower diversity in smaller lakes where there are fewer species to take on key functional roles in the foodweb. There is sufficient evidence to conclude that the supporting role of highly-connected species in freshwater systems may be important to the ongoing productivity of CRA fishery species especially in small lakes.

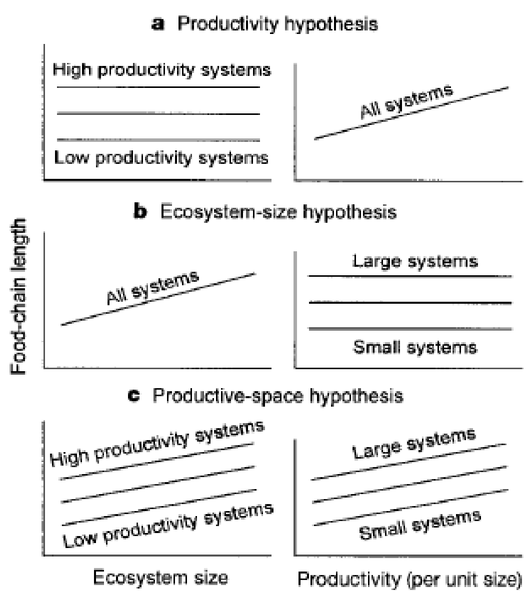


Figure 3.6.2. Hypothesized relationships between food-chain length and ecosystem size, and between food-chain length and productivity. a, for the productivity hypothesis; b, for the ecosystem-size hypothesis; c, for the productive-space hypothesis (extracted from Figure 1 of Post et al. 2000).

In marine food webs every species in the system has weak connections to multiple predators and even more prey and prey-of-prey. Hence the likelihood of compensation somewhere in the ecosystem is much higher than if species had few strong linkages. Theoretical studies have shown that if the system is characterized by only a few strong interactions embedded in a matrix of weak ones, these weak links enhance the stability of the entire system (McCann et al. 1998) and this kind of architecture (i.e. of strong and weak energy channels connected through nodal species) has been identified in real marine ecosystems (Rooney et al. 2006). In marine food webs diversity enhances ecosystem stability; therefore in speciose systems with mostly weak species interactions, prey of prey are less likely to be key contributors to the productivity of CRA fishery species but in less diverse systems, individual species taken on a larger proportion of support functions for CRA fishery species.

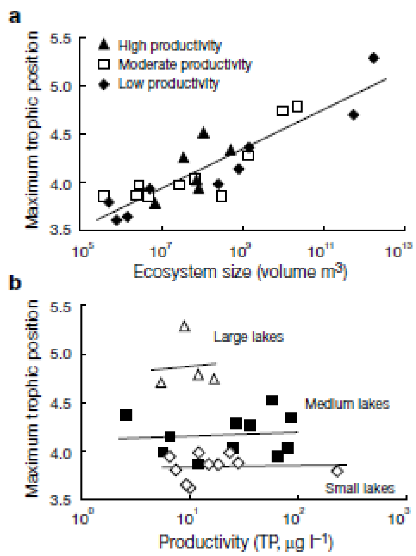


Figure 3.6.3 Relationships between maximum trophic position and ecosystem size or productivity. a, Ecosystem size for low (2-11µg l^{-1} total phosphorus (TP)), moderate (11-30 µg l^{-1} TP) and high productivity takes (30-250 µg l^{-1} TP). b, Productivity for small (3 10^5 to 3 X 10^7 m^3), medium (3 X 10^7 to 3 X 10^9 m^3) and large lakes (3 X 10^9 to 2 X 10^{12} m^3). Maximum trophic position is the trophic position of the species with the highest average trophic position in each of the lake food webs. The data are from 25 lakes in northeastern North America. (extracted from Figure 2 of Post et al. 2000).

(a) Important food chain considerations when food web models are not available

In most cases food web models will not be available for the system under study and so the connectance of species can only be speculated on. In such cases an extension of the food chain leading to the CRA fishery species should be considered. If the prey of prey of CRA fishery species are themselves key prey of key prey, following the definitions outlined in Section 3.1 above, then they should also be considered as supporting species.

Because aquatic foodwebs are size structured (Kerr and Dickie 2001, Neubert et al. 2000), the longer food chain length found in larger lakes equates to a larger range of body sizes at the upper end. This highly intuitive concept leads us to a proxy that could be useful for broad risk assessment in some evaluations: the more naturally truncated the size range of large fish in a lake, the shorter its food chain and less diverse it is likely to be and the greater the probability that key support functions for CRA fishery species fall to fewer species, i.e., there is less redundancy in the system. For example, this suggests that there may be more risk that a proportionally equivalent perturbation caused by a w/u/a could disrupt an important support function in a small lake versus a large lake.

Cury et al. (2005) assessed 46 marine trophic indicators used to characterize single food web components, changes in the functioning or structure of the whole ecosystem, and their ability to capture different types of trophic controls (bottom-up; top-down; mixed).

Six of those proved useful in detecting ecosystem level patterns: catch or biomass ratios, primary production required to support catch, production or consumption ratios and predation mortality, trophic level of the catch, fishing-in-balance, and mixed trophic impact. Where the necessary data exists, such indicators could be used as alternatives to food web models to detect changes in the trophodynamics of marine ecosystems. Xu et al. (2001) present a suite of structure and function indicators suitable for assessing changes in the trophodynamics of lakes.

3.6.1 Assessing Impacts on Highly-connected Species

It is highly unlikely that food web structure will be known at the time of most scoping exercises under the new Act. Collecting the necessary data and undertaking the analyses necessary to determine whether this support function is operative in influencing CRA fisheries species will be an expensive but possible procedure. It is also important to consider:

i) If the system that will be perturbed has properties that increase its likelihood of having a highly-connected species and low redundancy (e.g. a small lake).

ii) Whether the abundance or availability of the highly-connected species limits survival or growth of one or more life history stages (i.e. is a limiting factor), or limits reproduction in adults of CRA fishery species;

iii) What would the population-level impacts of altering the abundance or availability of the highly-connected species be on the CRA fishery species.

The discussion of how to determine threshold levels for highly-connected species, once identified, will be similar to those discussed for key prey in Section 3.1.1.3 in a single species context.

3.6.2 Operational Implications

If species in the food chain linking to CRA fishery species are highly-connected and meet considerations ii) and iii), then healthy populations of the highly-connected species must be maintained for the benefit of the CRA fishery species. We anticipate that very few highly connected species in key support roles will ever meet any objective standard of likely impact on the productivity of marine CRA fisheries. Such support functions if they occur are more likely to be found in small freshwater lakes and ponds.

3.7 INDIRECT SUPPORT FUNCTIONS - ENVIRONMENT- MODIFYING SPECIES

All organisms directly or indirectly modulate the physical environment in which they live. However some do so in ways that create resources for CRA fishery species and their prey. The activities of these organisms provide abiotic habitats that would not otherwise be available, often by means of disturbance to the physical environment. Because of the structural changes that are created, such organisms have been referred to as ‘ecosystem engineers’ (Jones et al. 1994). In the original definition, two types of ecosystem engineer were recognized: *allogenic engineers* that modify the environment by mechanically changing materials from one form to another and *autogenic engineers* that modify the environment by modifying themselves. Such organisms can be considered keystone species (Section 3.3) if their effects are large and disproportionate to their abundance, and may include structure-providing species (autogenic) if they modify bottom currents through their height off the bottom. In most cases the provision of habitat or other resources is an incidental by-product of the engineer's activity. Trophic interactions in the form of provision or consumption of tissue are excluded from this definition and autogenic engineers are only considered here if they modify bottom currents and physically or chemically alter the abiotic habitat.

Examples of environment-modifying species include excavating organisms such as crabs, bottom-feeding fish, rays, walruses, Gray whales, tile fish and salmon which dig nests; bioturbating (see glossary) or burrowing organisms such as clams and worms which oxygenate the sediments; filter feeders such as mussels which alter water clarity and sediment chemistry; and aquatic plants which alter flow, light quality and physical structure /cover and absorb nutrients / pollutants; amongst others. Ecosystem engineers have the potential to affect most aspects of the dynamics of aquatic ecosystems, but they are not important in all ecosystems. Thus, a major challenge in understanding the roles of ecosystem engineers is to discern the context dependency of their effects.

For example, numerous species of freshwater fish (smallmouth bass) or anadromous fishes (e.g., salmonids) dig or construct nests in which they lay their eggs, producing patches of disturbed substrate. Pacific salmon (*Oncorhynchus* spp.) dig large nests or redds. Depending on the species and size of salmon, a female salmon digs a pit up to 0.4 m deep and ranging from 1 to 17 m^2. Coho salmon redds average 2.8 m^2 in size. This nest digging has a variety of impacts on benthic habitats and communities. It displaces fine sediments, subsequently coarsening sediment. For example, a 5-year study of large sediments in a British Columbia stream found that sockeye salmon nest digging moved more sediment, and buried marked sediments deeper, than many flood events (Gottesfeld et al. 2004). In addition, bioturbation from salmon dislodges fine particulate matter into the stream's water column, driving a pulse in the concentration of suspended particulate matter. Concentrations of suspended particulate matter in stream water during salmon spawning are at least four times higher than before spawning. While salmon nest digging is a substantial disturbance to spawning areas, their bioturbation may actually decrease the susceptibility of streams to erosion from floods. Specifically, by sorting sediments into size classes, salmon nest digging may increase critical shear stress. Any CRA fishery species which depends on such environments will also be dependent on the species which creates them.

3.7.1 Assessing Impacts on Environment-Modifying Species

Badano et al. (2006) present a framework and methodology for assessing the impact of environment-modifying species on three general features of community organization: (1) species richness and composition, (2) stability of richness over time, and (3) dominance patterns of species assemblages. Their general approach involves comparing species assemblages in engineered and unmodified patches and can be used to determine whether or not the engineer has an important ecosystem effect through its activities. This method can be used to determine whether a mechanism exists, and its importance, which might affect the productivity of a CRA fishery species (but does not directly evaluate such a linkage).

The ICES-JRC Task Force 6 report (Rice et al. 2010) describes an axis of degradation for 'ecosystem engineers' as "the degree to which the functions served by the engineer's characteristic of the ecosystem are lost as the bioengineers are killed or the structures they create are damaged. The nature of the damage may vary considerably from permanent ecological damage e.g. the physical destruction of biogenic deep-water coral reefs, to recovery within days e.g. the reconstruction of burrows by benthic annelids. Sensitivity of various types of bioengineers to human perturbations varies greatly and is defined in relation to the degree and duration of damage caused by a specified external factor. Gradients of degradation will thus vary depending on the frequency and severity of the specific disturbance i.e. if the pressure is permanent, re-occurring or sporadic and the resilience of the particular bio-engineer(s) to the pressure(s) on it."

3.7.2 Operational Implications

For implementation of the Fishery Protection Provisions of the FA, it is very unlikely that the concept of environment-modifying species would lead to any additional areas being relevant for consideration, other than those already warranting consideration due to the presence of fish that are part of CRA fisheries. However, if fish that are part of CRA fisheries depend on the habitat modifications caused by environment-modifying species for some life history function(s), the impact on environment-modifying species become part of the factors considered in self-assessments and decisions by field officers. These self-assessments and management decisions would benefit from lists being available of what environment-modifying species are likely to be tightly linked to fish that are part of CRA fisheries, and what sorts of w/u/a would have detrimental impacts on the environment-modifying species. The former can be developed from existing knowledge for at least some important species in CRA fisheries at regional scales, and the latter developed nationally from previous work on Pathways of Effects (Clarke et al. 2008). (Pathways of Effects models, which describe the cause and effects between activities and their impacts on selected endpoints, can be built at different scales to meet various needs and also to assess cumulative effects to support the decision making process.)

3.8 IMPORTANT ECOSYSTEM FUNCTIONS RELATED TO FISH HABITAT

Fish habitat of CRA fishery species is protected under the new legislation. Although 'support' functions are restricted to those ecological functions preformed by 'fish', we highlight two important aspects of fish habitat that have demonstrated links to the productivity of CRA fishery species. These are plant-based biogenic habitats and primary production.

3.8.1 Plant-based Biogenic Habitats

Salt marshes are a good example of a biogenic habitat that can directly support CRA fishery species. They are grass-dominated habitats that extend from the low intertidal zone to the upper limits of the highest high tides. Salt marshes are among the most biologically productive ecosystems in the world and they help support rich coastal and estuarine food webs. Salt marshes provide critical resting and feeding grounds for migratory birds and serve as nurseries for some juvenile fish, shellfish, crabs, and shrimp because the physical structure of the grasses offers hiding places from predators. Eels are also known to inhabit brackish waters such as salt marshes (Gray and Andrews 1971). The roots and stems of marsh plants improve water clarity by slowing water flow and trapping waterborne sediments, which block sunlight penetration, clog filter-feeding animals and fish gills, and may contain toxins or heavy metals. In addition, the grasses absorb excess nutrients that enter groundwater and surface water from fertilizers and sewage discharge. This reduces the risk of eutrophication in estuaries and nearby coastal waters. Below the salt marshes eelgrass beds form a critical fish habitat. Eelgrass also provides vital services to improve water quality by filtering suspended sediment and excess nutrients. The ecological importance of eelgrass beds along the Atlantic coast became clear after an outbreak of a disease in the 1930s caused by a slime mold. Up to 90 percent of eelgrass in the region was killed and the die-off led to massive erosion and dramatic changes in water quality. Scallops, American brant, and other animals that relied on eelgrass beds for food and shelter suffered extensive mortality (Thayer et al. 1984). A recent review by ICES has shown that saltmarshes and eelgrass beds are used by commercial fish species primarily for nursery functions (ICES 2012).

In freshwater, wetlands and submerged aquatic plants along the shores of lakes, rivers, streams and ponds provide habitat for fish to spawn, feed and hide from predators or for ambush (e.g., northern pike). Aquatic plants also help to maintain water quality by stabilizing sediments, affecting flows, uptaking contaminants and affecting dissolved oxygen and suspended sediments/ turbidity. Functional linkages between wetlands separated by non-wetland areas (wetland complexes) can be very important to ecosystem functioning and include wildlife usage (e.g., bird forage areas), and surface water and groundwater connections. There are also upland wetland community types (e.g., meadow marsh - grasses) that are important for shallow marsh spawners like pike and warmwater fishes.

Freshwater aquatic plants are classifed into three groups (Figure 3.8.1.1): floating (water lilies), emergent (reeds and rushes) and submergent (many species of aquatic plants). The role of submergent aquatic vegetation (SAV) in structuring lake ecosystems, and in particular fish communities, has been the source of much research (e.g., Valley et al. 2004). Many fish, such as largemouth bass and northern pike, depend on SAV for food and shelter as well as spawning and nursery habitats where young may attach to foliage. Non-game fishes such as darters, minnows, and killifishes also depend primarily on nearshore emergent and submergent vegetation. These non-game fishes have been used to support significant bait fisheries in some areas and by definition those bait fishes are important prey of CRA fishery species. Productivity rates are generally higher in these vegetated areas and impacts to SAV could therefore cascade up the food web to mid- and higher-level CRA fishery species. Depending on the relative availability of aquatic vegetation area and the species assemblage requirements, aquatic vegetation may be of significant value to fish that are part of CRA fisheries or to fish that support such fish, and should continue to recevie consideration when evaluating potential impacts of proposed w/u/a. Such consideration will often be aided by setting thresholds and targets for conservation, protection and recovery when sufficient information exists on appropriate spatial scales (**Table 3.8.1.1.1)**. In some cases SAV may be resilient to some types of perturbation. In fact some level of natural disturbance, especially water level flucutation, is necessary to maintain healthy and diverse wetland assemblages (Figure 3.8.1.1; Mortsch et al 2006).

Also the techniques are well established for restoration of wetlands provided the conditions and seed supply are not too degraded. Rarely is it less expensive to reconstruct a wetland than it is to conserve it in its original functional state and level. There is much debate about whether the conditions and functional role of newly constructed wetlands are adequate and what the time lags are for full recovery.

Connectivity and size especially in comparison to the historical extent of a wetland are important to maintain. In some cases, wetlands at a particular location may have qualities that may not be reproducible.

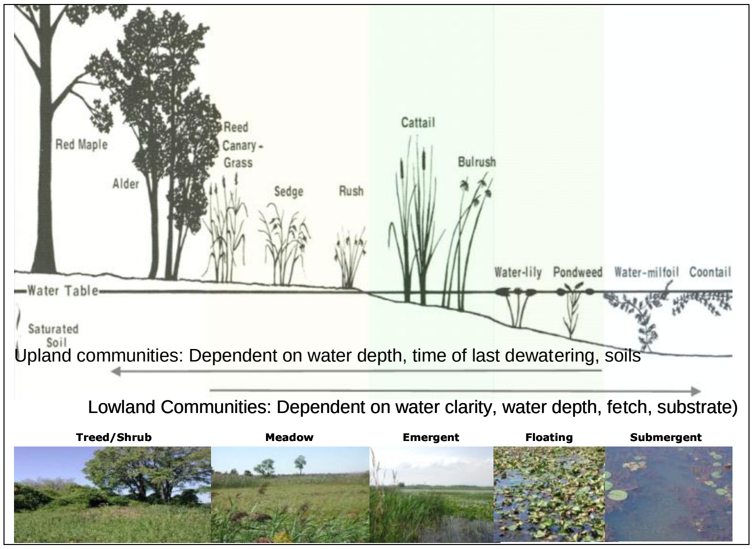


Figure 3.8.1.1: Schematic of wetland vegetation communities associated with different flooding regimes and other variables, such as soils/substrate and fetch (modified from Mortsch et al. 2006).

3.8.1.1 Assessing Impacts of Plant-Based Biogenic Habitats

In coastal and freshwater areas land-based and aquatic stressors include shoreline modification, infilling, water level manipulation, sedimentation, contamination, navigational dredging, contaminants, invasive species and eutrophication arising from excessive nutrient run-off, which can all alter biogenic habitats. As an example, plant structure-providing species can be influenced by w/u/a that modify the nutrient regime. For example, Duarte (1995) and Valiela et al. (1997) found that as N levels increase in shallow estuarine systems, seagrass beds are replaced by faster-growing macroalgae and eventually phytoplankton, although an assessment of more than 30 shallow systems with varying depths, water residence times and N input rates showed that this pattern does not always hold (Nixon et al. 2001). Similarly a host of urbanization impacts has affected Great Lakes coastal vegetation including eutrophication (Whillans 1990).

In some cases, precise thresholds for identifying when serious harm to structure-providing species affects fish production can be developed. In order to define thresholds for the abundance and spatial distribution of structure-providing species, we need to know how abundance and distribution influence the productivity of the CRA fishery species. For example, Valley et al. (2004) reviewing the role of SAV in Minnesota lakes reported that generally, conditions for game fish deteriorate when the percentage of a basin that is covered with SAV falls below 10% or exceeds 60%. This range does not consider basin morphometry (i.e., shallow *vs*. deep, sheltered *vs*. open) which ultimately controls how much vegetation naturally grows within a lake.

Table 3.8.1.1.1. Examples of Biogenic Habitats which May Provide Support for CRA Fishery Species Ranked by their Relative Ability to Recover from Perturbations. (M=marine; FW=freshwater; E=marine estuarine).

Structure-providing Species Categories	**Characteristics of Structure-providing Species**	**Examples**	**Relative Recovery Potential**	**Comments**
Short-lived Aquatic Vegetation	Annual reproduction with high clonal capacity; rapid growth rates; annual or short-lived perennial life span.	Coon's Tail/Hornwort (*Ceratophyllum demersum*); Pickerel Weed (*Pontederia* spp) FW; Sea lettuce (*Ulva* spp.) M; Sea feathers (*Chordaria flagelliformis*) M	High	Likely not of concern if capable of rapid recovery (i.e. local seed banks; adjacent SAV for recolonization).
Saltmarshes and Freshwater marshes	Perennial species with good regeneration capacity through stoloniferous roots; and sometimes seeds; widespread to regional species (depending on zones).	Cordgrass (*Spartina* spp.) E; Glassworts (*Salicornia* spp.) E; Bulrush (e.g. *Scirpus* spp.), Cattail (*Typha* spp.), Wild Rice (*Zizania* spp.) FW.	High to Moderate to Low depending upon scale of impact (local to regional)	Recovery can take 2–10 years or even longer depending on the nature and degree of the disturbance and the relative maturity of the marsh involved; Habitat can be subject to non-linear effects due to erosion; Good success with restoration projects; May be under stress from climatic changes (sea / water level changes) and human impacts (eutrophication, sedimentation, invasives, infilling / dyking / draining).
Submerged seagrass Meadows	Perennial species with good regeneration capacity through stoloniferous roots; sexual reproduction episodic and of variable success.	Eelgrass (*Zostera* spp.)	Moderate to Low	The potential for seagrass colonization is a function of both rhizome elongation, which determines patch growth, and reproductive effort, which sets the potential for formation of new patches. Clonal propagation is considered to be the most important process for the maintenance of seagrass meadows. Recruitment from sexual reproduction depends on the flowering probability and survival of the seeds, which are highly variable between species. Gaps in the order of 1 m^2 can be closed within one or a few years through horizontal rhizome extension (i.e. clonal growth), whereas gaps tens of meter across require a decade or longer to be recolonised through seed dispersal.

3.8.2 Primary Producers

Primary production is a key ecological function in all aquatic ecosystems, and one that can be influenced by human activities causing changes to nutrient inputs or light penetration and its quality (e.g., increasing suspended sediment load). Primary production can affect the productivity of CRA fishery species directly (if the CRA fishery species is a primary consumer) or indirectly through trophodynamics in some ecosystems (those with bottom-up control). When considering the support function of primary producers on CRA fishery species here, we limit our discussion to phytoplankton communities. Other primary producers such as sea grasses, marsh grasses, sea weeds etc. may be important supporting species and habitats for CRA fishery species through their provision of structure (Section 3.2) or as key forage (Section 3.1).

Globally, algae are responsible for almost 50% of the photosynthetic carbon fixed every year. The flow of energy from pelagic primary production through the food web will ultimately determine the productivity of CRA fishery species. “Bottom-up” control of productivity in higher trophic levels has been demonstrated in freshwater (Nixon 1988), estuarine (Houde and Rutherford 1993) and marine environments (Iverson 1990). Friedland et al. (2012) demonstrated that chlorophyll concentration, particle-export ratio, and the ratio of secondary to primary production were positively associated with fishery yields in a sample of 52 large marine ecosystems. Particle export flux and mesozooplankton productivity were also significantly related to yield on a global basis. There are likely baseline production differences between ecosystems and habitats but these are discussed in the Ongoing Productivity companion paper (Randall et al 2012). Further, some CRA fishery species will be directly linked to primary production. For example, bivalve molluscs are filter feeders and their growth is directly affected by planktonic food availability.

Elements of phytoplankton communities that are important to consider include the species composition, which influences the size composition and hence availability to grazing zooplankton and ichthyoplankton; abundance/biomass; and the duration and timing of blooms (periods of enhanced production). There are indications that some of the variation in year-class abundance in marine fish populations results from processes and events during planktonic stages of the early life cycle (Houde 1987, Pepin and Myers 1991). Timing of reproduction has often been implicated as a factor contributing to the success or failure of year-classes (Cushing 1975).

3.8.2.1 Assessing Impacts on Primary Producers

We acknowledge that there is an enormous literature on primary production that has not been thoroughly reviewed during the preparation of this report. We have not discussed the *important processes* that involve primary producers, such as benthic-pelagic coupling or microbial loops. Microbial food webs can comprise a significant fraction of the total community biomass in arctic rivers and lakes (Wrona et al. 2005) and w/u/a have potential to affect these functions especially when they involve long-term chronic disturbances. Here, we focus on the factors impacting primary production which in turn have the potential to affect the productivity of CRA fishery species, and have been shown to do so in the literature. The magnitude, composition, spatial extent and duration of a phytoplankton bloom depend on a variety of conditions, such as light availability, nutrients, temperature, and stratification of the water column. We discuss each of these factors and the impact human activities may have on them.

Chlorophyll is a key and widely used diagnostic marker of phytoplankton. The development of rapid spectrophotometric and fluorometric methods in the 1960s made it possible to map phytoplankton distribution in large bodies of water such as lakes and oceans. Optical properties of chlorophyll are important for remote sensing from space. In the 1980s, the coastal zone colour scanner (CZCS) made it possible to establish the key features of phytoplankton distribution throughout the world oceans, revealing the true extent of features such as coastal upwellings or equatorial enrichment. More recent sensors such as SeaWifs, MODIS or MERIS can provide chlorophyll maps for oceans and lakes over a specific region in near real time, allowing scientists to monitor phytoplankton dynamics and perturbations.

Primary production is dependent on temperature, light quality, light quantity and availability of nutrients (carbon, nitrogen, phosphorus, silica, trace metals, such as iron and a few vitamins). Consequently, human activities that alter these factors can affect primary production. In many cases pulse disturbances are likely to recover rapidly and have little lasting impact on the ecosystem function, however, the timing of the disturbance can be critical and if occurring during bloom periods could cause severe lagged disruptions to CRA fisheries through recruitment failure. However, it is important to consider cumulative effects which may initially cause increased primary production but ultimately lead to eutrophication with detrimental consequences for CRA fishery species. We draw attention to the work of Devlin et al. (2007) in developing nutrient thresholds for the EU Water Framework Directive (see below).

Temperature varies daily and on a seasonal basis as well as with latitude. The responses of marine and freshwater algae to temperature change have been summarized by DeNicola (1996) and include a variety of effects that have been studied at the cellular, population and community level. Individual responses are highly dependent upon the variability in the physicochemical environment and spatio-temporal pattern in species distribution. Physiological responses to temperature include changes in concentrations of photosynthetic and respiratory enzymes, changes in cell quota and nutrient uptake, as well as alterations in fatty acids and proteins. Individual populations have been shown to exhibit minimum, maximum and optimal temperatures for growth that contribute to species composition and diversity and eventually lead to seasonal succession. Temperature appears to be more important for selecting species than for controlling biomass. However, community structure usually recovers rapidly (< 1 yr) when temperature stress is discontinued.

Absorption and refraction by water, and dissolved and suspended matter determine the quantity and the spectral quality of light at a given depth. Light decreases exponentially with depth at a rate that depends on the particle content of the water. The euphotic zone is defined as the depth reached by 1% of the surface irradiance. In general, this depth is considered to be the lower limit beyond which phytoplankton cannot photosynthesize. However, some picoplankton appear to photosynthesize at lower levels down to 0.1% of the surface irradiance. Light quality is also an important factor as different species of algae have photosynthetic pigments adapted to harvest different wavelengths of light. As light decreases with depth, its spectral range also narrows in the blue region, as this wavelength is the least attenuated. Light attenuation occurs naturally as selected wavelengths are absorbed by the algae but human activities that increase the particulate matter content of the water will affect both light quality and light quantity and therefore primary production. The mixing depth of the producers and the photic depth of the system also determine light availability. Light conditions predominantly limit primary production in well mixed estuaries where the photic depth is determined by turbidity. Activities such as tidal barrages or other wet renewable energy schemes, which cause changes to the tidal regime in estuaries can cause changes to primary production.

In algae, carbon, nitrogen and phosphorus follow more or less closely a stochiometric relationship of 106:16:1 (by atoms), known as the Redfield ratio. When the N:P ratio of the seawater departs from the 16:1 value, then either N or P becomes limiting. Many human activities have a significant impact on the nitrogen cycle with potential for permanent alteration of the habitat. Introduction of nitrogen-based fertilizers for example can dramatically increase the amount of biologically available nitrogen in an ecosystem. Because nitrogen availability often limits the primary productivity of many ecosystems, large changes in the availability of nitrogen can lead to severe alterations of the nitrogen cycle. In nearshore marine systems, increases in nitrogen can often lead to anoxia (no oxygen) or hypoxia (low oxygen), altered biodiversity, changes in food-web structure, and general habitat degradation. These effects can also be seen in arctic freshwater ecosystems (Hansson 1992). One common consequence of increased nitrogen is an increase in harmful algal blooms (Howarth 2008). Toxic blooms of certain types of dinoflagellates have been associated with high fish and shellfish mortality in some areas and frequently force seasonal closures of many shellfish beds. Even without such economically costly effects, the addition of nitrogen can lead to changes in biodiversity and species composition that may lead to changes in overall ecosystem function.

Increases in nitrogen in freshwater ecosystems can lead to increased acidification. Increased eutrophication must be regarded in light of cumulative impacts leading to chronic alteration to primary production. Devlin et al. (2007) have developed guidance on setting thresholds based on empirical evidence for nutrients taking account of the biological response to nutrient enrichment evident in different types of water. This work was done to support nutrient thresholds and reference levels for ecological assessments under the EU Water Framework Directive. Their nutrient classification tool is based on the integration of three indices of: nutrient concentrations, primary production and dissolved oxygen levels. Nutrient concentrations indicate the level of enrichment, primary production indicates the accelerated growth of marine plants in response to the nutrient concentrations, and measurements of dissolved oxygen can indicate if the increased production has impacted the biology of species in the waterbody. Such thresholds may be informative in determining operational definitions of serious harm in the context of the Fisheries Act. In this context, however, the EU Water Framework Directive had a specific provision setting the management objective as maintenance or restoration of pristine water quality, and the quantitative benchmarks were set for that standard. Consequently, although the general approaches for setting management targets may transfer to Canadian requirements under the FA, the numerical values for management benchmarks would have to be recalculated for consistency with the provisions of the FA. Any such targets/benchmarks would also have to take account of a variety of other pieces of federal, provincial, and municipal policies and regulations regarding water quality standards. It may be appropriate to first investigate if water quality meeting these other standards would also be of high enough quality for the ecosystem to not degrade the productivity of fish that are part of CRA fisheries. If so, this aspect of fishery protection may already (or best) be delivered by these other policies, regulations, and standards. Where nutrients are limiting, such as in Arctic freshwater ecosystems (ponds, streams, lakes etc.) the input of nutrients has been shown to increase primary and fish production (cf. Wrona et al. 2005 and references therein).

Industrial agriculture, with its reliance on phosphorus-rich fertilizers, is the primary source of much of the excess nutrients responsible for fouling lakes. Development around lakes and ponds can cause a significant increase of phosphorus loading, which could result in the degradation of not only the surface water but the groundwater as well. Lakes and reservoirs respond more severely to excessive phosphorus than do fast moving waters, and do so at lower concentrations of phosphorus. Critical phosphorus concentrations which lead to eutrophication have not been defined because of site specific variables. The Vollenweider loading function can be used to assess overall aquatic ecosystem functioning in lakes and estuaries (Jones and Lee 1988). The model uses a statistical modeling framework to describe the utilization of phosphorus load in the production of planktonic algal chlorophyll, primary productivity, water clarity as controlled by primary production, hypolimnetic oxygen depletion rate, and overall fish yield. The basis for these empirical models is the functioning of several hundred ecosystems including lakes in Canada and the United States; they serve as a reasonable description of a "norm" and range of expected ("normal") conditions against which perturbations can be assessed. For example, based on the physical constraints that control water volume, the hydraulic residence time in the lake, and mean lake depth, combined with phosphorus loading, the Vollenweider model predicts the existing in-lake phosphorus concentration. Vollenweider found that when the annual phosphorus load to a lake is plotted as a function of the quotient of the mean depth and hydraulic residence time, lakes which were eutrophic tended to cluster in one area and oligotrophic lakes in another. These functions allow for prediction of the effects of nutrient loadings on these important ecosystem attributes.

It is necessary to document the impact primary producers have on CRA fishery species in order to determine their support function and potential thresholds by considering:

i) Whether the abundance, size structure, species composition, duration and/or timing of blooms of primary producers limits survival or growth of one or more life history stages (i.e. is a limiting factor) or limits reproduction in adults of CRA fishery species;

ii) What would the population-level impacts of altering the abundance, size structure, species composition or availability of primary producer communities be on the CRA fishery species?

The trophic indicators for marine ecosystems reviewed by Cury et al. (2005) and discussed under the Highly-Connected Species support function in Section 3.6 above could also be useful here, in particular the indicator: Primary Production needed to Support Catch (Pauly and Christensen 1995, Carr 2002). Xu et al. (2001) outline a number of ecosystem indicators for monitoring the structure and function of lake ecosystems including phytoplankton cell size and biomass (structure) and algal carbon assimilation ratio (function) which could be used to monitor local impacts of w/u/a.

3.8.3 Operational Implications for Plant-Based Biogenic Habitats and Phytoplankon

Any activity which results in a major input of organics and the resulting nutrients into freshwater, estuarine, coastal and marine ecosystems could affect primary producers through organic enrichment and changes to light quantity and quality. Point sources may have little impact if of short duration and isolated from other contributing factors (consideration of cumulative effects). Large scale chronic impacts such as contributions from agricultural fertilizers and livestock are predominant sources of non-point source run-off that result in water quality issues. Because of the pervasiveness of these threats it will be increasingly important to address i) and ii) so that links to CRA fishery species can be established, and to assess cumulative impacts.

In oligotrophic systems water quality changes from a w/u/a may have significant impacts on primary production. It is important to recognize that the w/u/a may introduce water quality changes in one location that have downstream or otherwise remote effects mediated through transport through the water column or ground water. For example, nutrients added to a tributary of a river could cause changes in primary producers at the river mouth.

4. OTHER ISSUES TO CONSIDER

In assessing the support species of CRA fishery species there are some additional generic considerations that should be taken into account. In this section we have attempted to outline some of the issues that could be important in this context. They are often difficult to operationally define but they should still be considered even if only in a subjective manner. It is important to note that each decision to carry out a w/u/a provides an opportunity to monitor the outcomes of the projects, update knowledge of the impacts of w/u/a on CRA fishery species and supporting species, and inform future management decisions in an adaptive way, noting, of course, taking advantage of such opportunities has resource and operational implications.

4.1 CUMULATIVE IMPACTS

Cumulative impacts can be defined as the aggregate impact which results from the incremental impact of an action when added to the consequences of other past, present, and reasonably foreseeable future actions and their interactions. We have discussed the role of cumulative impacts with respect to habitat degradation and fragmentation (Section 3.2) resulting in reduced habitat quality and supply. In order to assess cumulative impacts on a habitat it will be valuable to know what is the minimum size and spatial arrangement of the habitats that will support the CRA fishery species. Such thresholds are not well understood, even for well-studied ecosystems. Further, our view of cumulative impacts is often seen in light of present day habitats which can already be degraded. Cumulative impacts should also be considered when evaluating impacts on the abundance of prey species or species with other important functional roles in the ecosystem. The Canadian Environmental Assessment Agency (http://www.ceaa.gc.ca/default.asp?lang=En&n=9742C481-1&offset=5&toc=show) has provided a list of references on addressing cumulative environmental effects and assessment methods.

4.2 SPATIAL AND TEMPORAL SCALES IN RELATION TO SUPPORT FUNCTIONS

Species distributions range from those with very localized and patchy distributions to the highly migratory fish and marine mammals which use entire oceans and sometimes rivers too (salmon, eels). These varying spatial scales must be considered, particularly when we consider the interactions between the physical environment and biological processes, including variations in the climate system with the effects of CRA fisheries and their combined effects on biota.

The ICES-JRC Task 6 report (Rice et al. 2010) recognized that a "wide range of human activities causing pressures that may degrade the status of the sea floor operate at different but always patchy spatial scales" and that "The patchiness of the human activities causing the pressures also means that the scales of initial impacts of those activities are usually also local. Initial impacts of fishing are where the gear is actually deployed; aquaculture where the facilities are sited; industries and municipalities where the facilities or town are located; river-based depositions start in the river plumes; shipping, cables and pipelines in corridors."

Geographical boundaries of ecosystems are not static and should be considered dynamic or fuzzy. They depend on the scale considered and can never encompass all the relevant processes. Boundaries may also be variable as the ecosystem's extent and location change seasonally or from year to year under changing climatic conditions (Garcia et al. 2003).

Intermediate scales which reflect biogeographic zones (DFO 2009) can link variations in the physical and chemical environment with biological productivity, the status of various marine populations, and the wide spectrum of human interactions contributing to observed changes. This is similar to how the management of freshwater is apportioned into zones which are hydraulically (e.g. watershed) or biologically based (e.g. fish stocks or metapopulations). Although these scales often have high biological meaning, they are also often much larger than the scales of w/u/as and their direct impacts.

Assessing at a scale which is much larger than the impact scale may result in either the impact not being observed or that causes of observed changes cannot be identified. Context for the impact and whether it warrants management action is needed at both local and relevant larger geographic scales (e.g., within the boundaries of impact, within the scale at which an average-sized fish subpopulation functions, or within an area that is hydrologically discrete, or within the context of the entire range of the specific fishery or species range). Likely more than one scale and resolution needs to be considered to encompass the range of impacts and assess the full range of consequences of a w/u/a.

Table 4.2.1 summarizes expectations for the anticipated spatial scales at which w/u/a affecting the species performing the support functions listed will usually be manifested, that is, ask:

1) Whether impacts will appear as a local phenomenon; and
2) Does local degradation affect the big picture in a proportionate or disproportionate way?

Table 4.2.1. Anticipated Effects of Work/Undertakings/Activities for each Support Function in Relation to Spatial Scale and Response Functions.

Supporting Function	Anticipated Spatial Scale of Impacts	Vulnerability to Degradation	Qualities of the Axis of Degradation	Potential Degradation Thresholds	Common Deleterious Actions	References
Key Prey Species	Generally degradation will be at the same scale as the impact unless source/sink population affected, or unless a critical life history stage or process is affected (e.g. spawning aggregation site).	Requires assessment on a case by case basis as it will be species-dependent.	Response dependent on alternative prey options.	Fisheries reference points	Fisheries, toxic chemicals and pollutants; Any w/u/a affecting essential habitat supply (depends on trophic level of species)	
Structure Providing Species	Most likely has degradation effects at the same or similar scale as the impact unless the structure provider is a keystone species.	Often high if subject to mechanical disturbances due to morphology of dominant species; some may be able to tolerate short term impacts; should be assessed on a case by case basis. Invasive plants could colonize disturbed areas and therefore impact may be more long-lasting or permanent.	See Note 1 below	See Note 2 below	Any w/u/a affecting biogenic structure or life cycle of species comprising biogenic habitat (see Note 3 below)	
Keystone Species	Degradation at apparently local scale can be widespread in a generation through population connectivity of prey.	Often high because they are not necessarily very abundant but environmental tolerances can vary.	Non-linear response due to cascading effects; may have alternate stable equilibrium states.	Fisheries or habitat reference points depending on the keystone effect.	Fisheries, hunting, toxic chemicals and pollutants that disproportionately affect the keystone. Any w/u/a affecting essential habitat	Gregr et al. 2008, Power et al. 1995, Paine 1996

Supporting Function	Anticipated Spatial Scale of Impacts	Vulnerability to Degradation	Qualities of the Axis of Degradation	Potential Degradation Thresholds	Common Deleterious Actions	References
Wasp-Waist Species	Local populations may be depleted and affect ecosystem functioning in a disproportionate way depending on species.	Relatively resistant to most common destructive activities except for overfishing when wasp-waist species is a CRA fishery species.	Non-linear response due to cascading effects.	Fisheries reference points	Fisheries; Any w/u/a affecting essential habitat supply.	Bakun 2006, Shannon et al. 2009
Highly-Connected Species	Local to ecosystem scale impacts depending on recoverability and size of impact and alternate prey.	Requires assessment on a case by case basis as it will be species-dependent; relatively resistant to degradation due to reproductive capacity / ubiquity.	Non-linear response due to cascading effects.	Fisheries or other reference points depending on taxa.	Fisheries; Any w/u/a affecting essential habitat supply (depends on trophic level of species)	
Apex Predators	May have degradation effects at larger scale than the impact due to high mobility and relative rarity of apex predators.	Highly vulnerable as species generally have a long life expectancy and produce few offspring with a higher reproductive investment not conducive to rapid stock rebuilding. Tolerances may also be low.	Non-linear response due to cascading effects.	Fisheries reference points	Fisheries especially recreational fisheries, water flow and dam construction - change in thermal regime, vegetation removal, gravel extraction	Ritchie and Johnson 2009

Supporting Function	Anticipated Spatial Scale of Impacts	Vulnerability to Degradation	Qualities of the Axis of Degradation	Potential Degradation Thresholds	Common Deleterious Actions	References
Environment Modifiers	May have far-field degradation effects for nest builders if the impact occurs outside of spawning/nursery area.	Requires assessment on a case by case basis as it will be species- and taxon-dependent. Invasive plants could colonize disturbed areas and therefore impact may be more long-lasting or permanent.	May be rapid response of some species due to strong habitat modifications created by the EM species.	Fisheries reference points for some; integrity indices; habitat supply.	Destructive fishing practices, (see above)	
Primary Producers	May have far-field degradation effects in open systems but spp relatively ubiquitous.	Rapid response to changes in nutrient and light regimes if those factors are limiting in impacted area but also rapid recovery depending on type. (match-mismatch applies)	Nutrient uptake follows Michaelis-Menton kinetics with rapid uptake then saturation.	Nutrient thresholds and reference levels for ecological assessments under the EU Water Framework Directive; planktonic indices.	Agricultural run-off, sewage, pollution, turbidity, tidal energy schemes.	Devlin et al. 2007

Note 1: Recovery may be highly protracted as recruitment may be linked to the modified habitat created by the presence of the adults and/or species may be long lived (coral, sponge) with episodic recruitment. Non-linear impact on species requiring structure depending on loss or degradation.

Note 2: For stoloniferous plants- the patch size which could be repopulated through root systems (asexual) vs. distances dependent on sexual reproduction for recolonization; For attached fauna- distance to nearest neighbour relative to dispersal distances to avoid local extinction vortex; For attached plants-depth/area of photic zone.

Note 3: Any w/u/a affecting biogenic structure or life cycle of species comprising biogenic habitat. (e.g. aquatic pollution, aggregate extraction, dredging, trawling, infilling, sedimentation, mechanical removal).

4.3 NON-NATIVE SPECIES

In this document we have considered examples for support roles filled by native species only. Non-native species fulfill some of these roles already and the implications of a w/u/a affecting support functions provided by such species must be considered. Further, some invading or nuisance species have the potential to cause greater ecosystem change than do some of the support species identified, and those have also not been considered. Making decisions on whether to maintain or eradicate non-native species in relation to w/u/a is outside of the scope of this paper.

5. THE FUNCTIONAL DEFINITION OF 'SUPPORT' IN OTHER JURISDICTIONS

The EU member States are currently struggling with similar but not identical challenges as they implement the Marine Strategy Framework Directive (MSFD). This fairly new legislation is an overarching policy framework for management of all human activities in marine environments. In addition to many provisions about various industries and socio-economic goals, States are required to achieve "Good Environmental Status (GES) in their waters by 2020. Moreover interpretation of GES is guided by ten "Descriptors" of what constitutes GES. These Descriptors are a complex and sometimes overlapping mix of pressure, state, and process properties of ecosystems and their uses, some of which are directly relevant to "supports" in the amended Fisheries Act; particularly the Descriptors Biodiversity, Food Webs, and Seafloor Integrity.

Achievement of GES is the purview of member States, but overall guidance documents for each Descriptor, and for aggregate assessments of GES, were prepared by joint working groups of ICES and JRC. All are available on the ICES website (http://www.ices.dk/projects/projects.asp#MSFD). These guidance documents include extensive literature reviews of the concepts, what characterizes "axes of degradation" of each Descriptor and its subcomponents, the ecological characteristics of the boundary on the axes that differentiate 'good" from "not good" ES, and practical issues associated with implementation (selection of indicators, dealing with scale, etc). Material in these guidance documents could be helpful in the next phase of building the implementation approach for "supports", particularly with regard to what actions are appropriate when different strengths of evidence for linkages or impacts is available, and for guiding consistent judgment calls on decisions along a gradient of impact.

The other potentially relevant piece of EU legislation is the Water Framework Directive (WFD). The WFD is over two decades old, and is highly prescriptive, generally requiring pristine water quality be maintained except where impacts are explicitly condoned. Work for implementation of the MSFD has revealed challenging differences between the standards set in the two Directives, and there are a number of stresses in agencies trying to implement both. Given the much greater similarity of the objectives of the MSFD and the amended Fisheries Act of Canada, lessons for operational guidance are more likely to come from experience with MSFD than the WFD.

In the US the provisions regarding essential fish habitat (EFH) in the Magnuson-Steven Act are the closest parallel to the "supports" function in the amended Fisheries Act of Canada (http://www.nmfs.noaa.gov/msa2005). The EFH provisions proved very challenging to implement, with initially large differences in approach and interpretation across the Fisheries Management regions. Substantial litigation accompanied by substantial national working group efforts have increased the coherence of interpretation of EFH in the US. Science experts in the US have suggested that the role of the Courts in shaping the interpretation of EFH in the US was so strong that their approaches may limit relevance to the Canadian context. Many of the tools developed for quantifying habitat quantity and quality could be extremely valuable at the implementation stage, and it is the “scoping guidance” on how the policies should be interpreted that may not generalize from one country’s legislation to another’s.

The legal bases for aquatic habitat protection in Australia and New Zealand are both substantially different than in Canada, with each country having some overarching “sustainability act” guiding all resource management. However, extensive risk based tools for prioritizing and guiding activities for protection of aquatic habitats have been developed by these jurisdictions, and these tools may prove useful in the Canadian context.

6. GLOSSARY OF TERMS

Apex predators: Apex predators occupy the top trophic position in a community; these are often large-bodied and specialized hunters that have no predators of their own within their ecosystems. They can have a controlling influence on the structure of lower trophic levels, referred to as “top-down” control (e.g., large sharks).

Biogenic habitat: Habitat created by a living organism (i.e. sea pen fields, sponge reefs, deep sea coral, etc.).

Bioturbation/Bioturbators: Species which through burrowing or manipulating sediments affect the flux of oxygen, nutrients, sulphides and other chemicals from the sediments on which other organisms depend and as such they are essential for the structure and function of sediment and near bottom communities.

Cascading effects: The effects from a direct action on a species or habitat that become apparent in different components of the ecosystem at later times and often in areas beyond the direct impact of the action. (defined as the indirect effects of exploiting marine predators on the abundance, biomass, or productivity of species, or species assemblages, two or more trophic links below the exploited predator).

Environment-modifiers: Organisms that directly or indirectly modulate the physical environment in which they live in ways that create resources for CRA fishery species and their prey. The activities of these organisms provide abiotic habitats that would not otherwise be available, often by means of disturbance to the physical environment (e.g., walrus).

Generation-time: The average age of spawners in the population. This should be determined over multiple generations as generation time is likely to be affected by previous exploitation and size selective mortality processes.

Highly-Connected Species: A species with a high proportion of links to other species in a food web compared with the average number of links between species. (e.g., krill).

Key Prey Species: The suite of species that are essential food items. Key prey can be distinguished from general prey items in that the relative abundance, diversity, availability and/or nutritional value of key prey impacts the productivity of a CRA fishery species.

Keystone species: Those which have an impact on ecosystem structure and functioning disproportionate to their biomass or abundance in the ecosystem (e.g., sea otter).

The criteria for a keystone species are that the species exerts top-down influence on lower trophic levels and prevents species at lower trophic levels from monopolizing critical resources, such as space or key producer food sources. They maintain community diversity by preying selectively on competitively superior prey taxa, thereby preventing the exclusion of relatively weak competitors.

Primary-producers: In this context: Photosynthetic algae living in the water column (phytoplankton).

Globally, algae are responsible for almost 50% of the photosynthetic carbon fixed every year. The flow of energy from pelagic primary production through the food web will ultimately determine the productivity of CRA fishery species. “Bottom-up” control of productivity in higher trophic levels has been demonstrated in freshwater, estuarine and marine environments.

Structure-providing Species: In this context, these are biogenic habitats created by animals (‘fish’). This may be the organism itself, such as a bed of mussels or sponges, or arise from an organism’s skeletons, such as the mounds created by dead corals or sponges.

These biogenic substrates provide three-dimensional habitats for a large variety of species. The link between the CRA fishery species and the biogenic habitat can range from essential to facultative or it may only be a preferred location, and the importance of habitat will vary seasonally and across life-history stages.

Wasp-waist Species: Species usually occupying an intermediate trophic level and expected to expected to play a critical role in regulating the transfer of energy from primary and secondary producers to the higher trophic level species in the ecosystem. (e.g., capelin).

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