

NUTRIENT YIELDS FROM NORTHWEST ATLANTIC FISHERIES:  
ANALYSIS, INDICATORS, AND OPTIMIZATION

by

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## Abstract

There is a fundamental challenge to the dual objectives of improving ocean health and ending hunger and malnutrition. Seafood derived from fisheries plays a critical role in providing a variety of nutrients to people who are at high risk of malnutrition. However, catches of overfished stocks must be reduced, at least in the near term, to allow stock biomass to rebuild. Reducing catches, in turn, may result in reduced availability of nutrients for people. This suggests that fisheries decisions may have unintended consequences for human nutrition. Currently, nutrition information does not inform fisheries decision-making processes. One challenge to the integration of nutrition and fisheries policy is that fisheries yields are quantified, analyzed, and broadly conceived in terms of the weight of the catch, without regard for the nutrient content of the catch. In this dissertation, I explore this issue from multiple perspectives. In Chapter 2, I analyze the yields of a suite of nutrients obtained from fisheries landings in the Northwest Atlantic Fishery Organization (NAFO) region over the period 1950-2014. Results demonstrate that trends and patterns in nutrient yields can differ, sometimes substantially, from those associated with catch weights. Some species may appear to be minor from a catch weight perspective but play outsized roles in supplying specific nutrients. Notably, recent yields of multiple nutrients have been disproportionately reliant on one species (Atlantic herring, *Clupea harengus*). In Chapter 3, I evaluate the nutrient yield consequences of the use of Atlantic herring as bait in the Maine fishery for American lobster (*Homarus americanus*). Results indicate that the lobster fishery likely consumes more nutrients through its use of herring than it produces through lobster landings. In Chapter 4, I present a theoretical approach to optimizing fisheries' nutrient yields relative to landings weights. This approach maximizes the difference between total nutrient yields and total fishery landings. In Chapter 5, I apply this optimization approach to NAFO fisheries. The results of Chapter 5 indicate that recent NAFO nutrient yields could be maintained at relatively high levels even if total landings were to be reduced.

## **Lay Summary**

Fisheries make critical contributions to human nutrition, particularly for populations at risk of nutrient deficiencies. Fisheries also face a highly uncertain future due to climate change and the effects of historic and ongoing overexploitation. Recent years have seen a number of calls for fisheries policy to take nutrition into consideration. However, a fundamental gap precludes this integration: fisheries yields are analyzed in terms of catch weights, not nutrient yields. In this dissertation, I develop methods and indicators for the analysis of nutrient yields in fisheries, put forward an approach for optimizing nutrient yields, and apply these measures to the fisheries of the Northwest Atlantic Fisheries Organization (NAFO) region. Results show that trends and patterns in nutrient yields often differ substantially from those associated with catch weights, illustrating the need for dedicated analysis of nutrient yields. Optimization scenarios indicate that catches may often be reduced with minimal effects on nutrient yields.

## **Preface**

All of this dissertation's research chapters have been prepared for submission as manuscripts. John David Driscoll was lead investigator for all chapters, responsible for concept formation, data collection and analysis, and manuscript composition, with Dr. Kai Chan contributing as a co-author to all of these steps.

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## List of Abbreviations

ALA	Alpha-linolenic acid
$B_{80}$	Catch biomass required to produce the first 80% of a nutrient's total yield
$B_{MSY}$	Biomass at maximum sustainable yield
DHA	Docosahexaenoic acid
$E$	Simpson's evenness
EPA	Eicosapentaenoic acid
LC-PUFA	Long chain polyunsaturated fatty acids (LC-PUFA)
MSY	Maximum sustainable yield
MTL	Mean trophic level
NAFO	Northwest Atlantic Fisheries Organization
RDA	Recommended Daily Amount

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## Chapter 1. Introduction

Of all human enterprises, few if any are as spatially extensive, resource intensive, and environmentally consequential as the systems that exist to provide food for humans. Cropland and pastures cover an estimated 38% of the earth's ice-free land (Foley et al., 2011), and fishing fleets operate over more than half of the world's ocean surface (Watson et al., 2013). Through agriculture, humans have altered or replaced ecosystems, upset vast geochemical cycles, and appropriated a significant proportion of the world's available fresh water (Foley et al., 2005). The impacts of fisheries differ in type but have been similarly transformative. Overall, land use change (primarily driven by agriculture) and direct exploitation (primarily through fisheries) represent the two primary drivers of anthropogenic impact on terrestrial and marine ecosystems, respectively (Diaz et al., 2019). The sustainability of the global human population clearly requires sustainable food systems.

Fisheries are an integral part of these food systems. The nutritional contributions of seafood to diets in developing and/or low-protein countries have important consequences for human health, as nutrients derived from seafood can be particularly important for people facing food insecurity (Béné & Heck, 2005). Per capita seafood consumption is greater in developed countries than in developing countries, but seafood consumption has more than doubled in developing and/or low income food-deficit countries since the early 1960s (FAO, 2016). In the coming decades, demand for seafood is projected to increase substantially (Tilman & Clark, 2014). Fisheries will bear the brunt of much of this demand: while the share of seafood produced by aquaculture has consistently grown in recent decades, data from the United Nations Food and Agriculture Organization (FAO) indicate that fisheries still supplied approximately 56% of seafood production as of 2014 (FAO, 2016).

However, the future of food from fisheries faces many challenges, including an apparent long-term decline in global fisheries catches and catch per unit effort

(Pauly & Zeller, 2016a) and a generally increasing proportion of stocks that are overexploited (FAO, 2018). While the status of stocks appears to be improving in those regions that have the most rigorous fishing management, concerns persist for areas with weaker oversight (Hilborn et al., 2020). Looking to the future, food production from fisheries is likely to be increasingly challenged by projected spatial shifts in catch potential due to climate change (Cheung et al., 2010), as well as restrictions on catches required by marine conservation efforts (Rice & Garcia, 2011).

International objectives for ending hunger and malnutrition (DSDG, 2019) and sustainably managing marine fisheries (DSDG, 2019; CBD, 2019) thus face a basic challenge. Fisheries make key contributions to human diets, particularly in regions where people may be at risk of food insecurity and/or nutrient deficiencies (Golden et al., 2016a). However, for many fish stocks and ecosystems, reductions in fisheries exploitation are required to rebuild overfished stocks and protect key ecosystem components (Worm & Branch, 2012; Teh & Sumaila, 2020). While such efforts can rebuild stocks, they are associated with reduced catches, at least in the near term (Hilborn et al., 2020). Reduced catches, in turn, may increase the risk of nutrient deficiencies in dependent human populations (Golden et al., 2016a). Thus, efforts to reduce exploitation, while allowing stocks to rebuild and long-term catches to stabilize, may have consequences for human health if they result in sustained periods of relatively low catches.

Awareness of this challenge is rising. As fisheries are increasingly being understood as part of the larger food system, recent research has highlighted the need for fisheries policy that is informed by food security and nutrition concerns. However, a key gap currently impedes this integration: there are no established methods or indicators to support the analysis or monitoring of the status of nutrient yields from fisheries. This gap is the focus of my dissertation. In this introductory chapter, I review the context for this research: the role of fisheries in providing nutrition to

humans, the status of global fisheries, and evidence for the necessity of nutrition-informed fisheries science and policy.

### **1.1. Fisheries and human nutrition**

The four components of food security are availability, access, utilization, and stability (FAO, 2009). The availability element refers to not only the amount of food that is available, but also its type and quality (Loring & Gerlach, 2015). Access comprises economic elements (e.g., affordability), policy and management elements (e.g., government trade agreements, hunting and fishing regulations), and considerations of physical and logistical access to supplies of food (Loring & Gerlach, 2015). The third element, utilization, refers to a person's ability to meet all of their food-related needs from the foods that they can access (Loring & Gerlach, 2015). These three elements are hierarchal, as availability allows but does not guarantee access, and access allows but does not guarantee utilization (Barrett, 2010). The fourth component, stability, essentially refers to the persistence of the first three components over time and across variability in political, economic, and environmental conditions (Cawthorn & Hoffman, 2015). By focusing on the yield of protein and a suite of micronutrients, this dissertation addresses aspects of the "availability" dimension (i.e., supply), with consequences for the "utilization" dimension (specifically, micronutrient profiles of diets). It should be noted that fisheries make other contributions to food security as well (e.g., enhancing access through providing employment; HLPE, 2014).

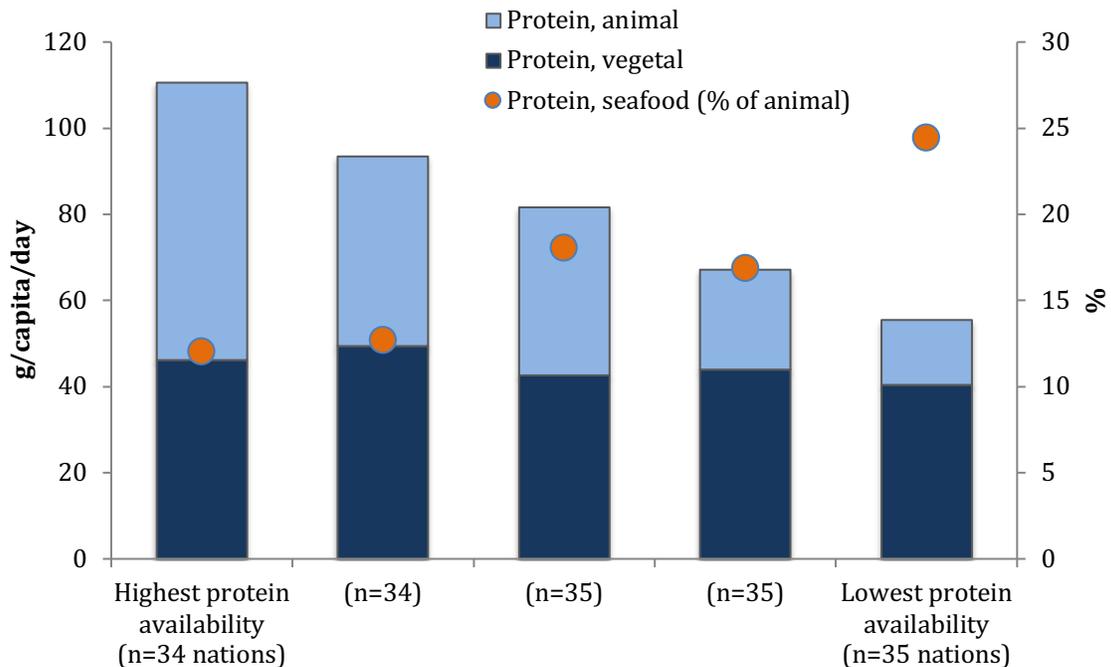
The concept of food security takes into consideration nutrition as well as dietary energy (FAO, 2009). However, efforts to evaluate and address food security have tended to focus on availability of dietary energy (Cawthorn & Hoffman, 2015). This is a shortcoming, as diets that are heavily dependent on carbohydrate-rich staples such as rice, wheat, and maize may appear sufficient from a dietary energy standpoint while failing to supply adequate protein and/or micronutrients (Béné & Heck, 2005). A diet that provides sufficient energy but insufficient amounts of protein can lead to protein malnutrition (i.e., kwashiorkor). Imbalanced diets can also lead to

a number of micronutrient deficiencies, in a phenomenon known as “hidden hunger” (Muthayya et al., 2013). For these reasons, the United Nations Food and Agriculture Organization (FAO)’s suite of food security indicators includes several that are explicitly focused on protein and micronutrients. Protein-related indicators account for two of the five “availability”-related food security indicators (“average protein supply” and “average supply of protein of animal origin”), and the prevalence of micronutrient disorders (e.g. anemia, vitamin A deficiency, and iodine deficiency) account for four of the eight “utilization” indicators (FAO/IFAD, 2015).

#### 1.1.1. Macronutrients

Globally, seafood from fisheries and aquaculture supplied approximately 17% of available animal protein in 2017 (FAOSTAT, 2020). However, the contribution made by seafood is much higher for some regions. In 2017, for the 35 nations with the lowest availability of animal protein and total protein, seafood from fisheries and aquaculture provided nearly 25% of available animal protein (Figure 1.1). In the same year, for the 47 nations designated by the United Nations as “Least Developed Countries”, seafood provided over 29% of available animal protein (FAOSTAT, 2020).

Furthermore, the data in Figure 1.1 likely under-represent seafood’s actual contribution to protein availability. The activities of small-scale fisheries, shore-side harvesters (who are often women; Harper et al., 2013), and other subsistence efforts are under-reported for many nations, and thus may not make their way into official fisheries data (Pauly & Zeller, 2016a). These missing fisheries may be of the sort that are closely linked to the activities of people who are at risk of food insecurity: for example, McGarry and Shackleton (2009) reported that mussels were the most common wild food, of any sort, collected by at-risk children living in a coastal region of South Africa.



**Figure 1.1.** Mean daily per-capita availability of plant and animal-derived protein across 173 nations in 2017. Nation groups organized from highest protein availability to lowest availability. Contributions of seafood to total animal protein shown on secondary axis. Data from (FAOSTAT, 2020).

### 1.1.2. Micronutrients

Micronutrients refer to the array of vitamins and minerals that the human body requires for proper nutrition. Deficiencies in micronutrients are estimated to affect up to 2 billion people worldwide, and contribute to a number of negative health consequences, especially in children and child-bearing women (Muthayya et al., 2013). Relative to chronic hunger (i.e., insufficient intake or biological use of calories and/or macronutrients), the burden of hidden hunger on human health may have declined comparatively less in recent decades (Gödecke et al., 2018).

At the global scale, the most prevalent micronutrient challenge is iron deficiency and associated anemia (Skalicky et al., 2006). Iron is required for the synthesis of hemoglobin, and iron deficiency is the primary cause of anemia (Caulfield et al.,

2006). The global prevalence of anemia has essentially remained unchanged since 2012 (FAO, IFAD, UNICEF, WFP and WHO, 2019). Globally, iron deficiency anemia is estimated to affect 18.1% of children (0-60 months) (Black et al., 2013), and 32.8% of women of child-bearing age (FAO, IFAD, UNICEF, WFP and WHO, 2019). While the prevalence of anemia in women of child-bearing age is particularly high in African and Asian countries, iron deficiency and associated anemia may be found at high prevalence in high-risk populations in developed countries, including the United States (Skalicky et al., 2006). Anemia in pregnant women increases the risk of maternal and perinatal mortality (Black et al., 2013). Iron is found in a variety of animal and plant-derived foods, but the type that is found in animal-derived foods is heme iron, which is more easily absorbed during digestion (Golden et al., 2016b).

Zinc is required for a host of basic processes in the human body, with even mild deficiency resulting in measurable impacts on a variety of processes (Prasad, 2013). An estimated 17% of the world's population may be at risk of zinc deficiency, based on dietary patterns (Black et al., 2013). Zinc deficiency is primarily caused by a lack of sufficient intake, rather than losses (Caulfield et al., 2006), and is a characteristic of diets that rely heavily on protein from cereals (Prasad, 2013). Animal tissue is the best source of zinc, and components of plant-based diets (fiber and phytates) can inhibit its absorption in the body (Caulfield et al., 2006).

Iodine deficiency can lead to a number of health impacts, including multiple fetal and childhood developmental problems such as hearing impairments, poor growth, and cognitive impairment (Caulfield et al., 2006). Iodine deficiency is considered to be the leading cause of preventable brain damage (De Benoist et al., 2008). Globally, nearly 31% of the world's population was estimated to have insufficient dietary intake of iodine as of 2007, with Europe (52.0%), the eastern Mediterranean (47.2%), and Africa (41.5%) having the highest prevalence among regions (De Benoist et al., 2008). Soils are naturally iodine deficient in many parts of the world, especially mountainous areas, and human populations in such regions will inevitably be deficient in their iodine intake unless exogenous sources are introduced

(Zimmerman, 2009). Seawater, on the other hand, contains a relatively high concentration of iodide ions. Thus, while most foods contain little iodine, foods derived from marine ecosystems (e.g. marine fish and marine plants) can contribute substantial amounts to human diets (Zimmerman, 2009).

Poor dietary intake, and in particular low consumption of animal-derived foods, is the primary cause of vitamin A deficiency (WHO, 2009) and a major cause of vitamin B<sub>12</sub> deficiency (Stabler & Allen, 2004). Vitamin A deficiency is associated with several health effects, particularly in pregnant women and children. It is a primary driver of preventable childhood blindness, and by weakening resistance to infection, increases infection-related mortality (WHO, 2009). As of 2009, an estimated 33% of the world's preschool-aged children and 15% of pregnant women were deficient in vitamin A, with southeast Asia and Africa demonstrating the highest prevalence of deficiency in both groups (WHO, 2009). Vitamin B<sub>12</sub> deficiency is associated with a variety of potentially severe health effects, which may have potentially irreversible consequences when they occur in children (WHO, 2008).

Marine organisms are the primary dietary sources for the long chain polyunsaturated fatty acids (LC-PUFA) eicosapentaenoic acid (EPA) and docosahexaenoic acid (DHA), which together with alpha-linolenic acid (ALA) are collectively known as the omega-3 fatty acids. Blasbalg and colleagues evaluated trends in omega-3 consumption from different food sources in the USA over the 1909-1999 period, and found that fish were consistently the most significant source of EPA and DHA (Blasbalg et al., 2011). A high ratio of omega-6 to omega-3 fatty acids in diets is thought to be correlated with a host of negative health consequences. Loring and colleagues (Loring et al., 2010) compared the health benefits of omega-3 consumption in a subset of Alaskan fish species with risks posed by bioaccumulated methylmercury, and found that the benefits of salmon and trout, (which have the highest concentrations of omega-3s) consistently outweighed risks.

The importance of seafood to micronutrient nutrition is particularly high for people who otherwise consume few animal-derived foods. Relative to plant-sourced foods, animal-derived foods tend to have higher amounts of and/or more bioavailable micronutrients, including vitamin B<sub>12</sub>, riboflavin, iron, and zinc (Murphy & Allen, 2003). Some of these micronutrients work synergistically in the body: vitamin A, riboflavin, and iron are all needed for the synthesis of hemoglobin, for example, and thus a targeted subsidization of only one of these micronutrients may not address anemia (Murphy & Allen, 2003). Thus, even small amounts of animal-derived foods may provide a suite of important micronutrients that are often lacking in people who primarily eat a vegetarian diet (Murphy & Allen, 2003). For this reason, the micronutrients supplied by fish (e.g., calcium, iodine, iron, vitamins A and B, and zinc) may be extremely important for proper nutrition where diets are dominated by starchy staples (Béné & Heck, 2005).

## **1.2. Global fisheries: status and projections**

While humanity's cumulative need for nutrition from fisheries has never been greater, many of the world's marine fish stocks, and the ecosystems that support them, require rebuilding and increased protection from human exploitation.

Historical over-exploitation of large vertebrates and structure-forming invertebrates in coastal waters put into motion ecological effects that have cascaded through space and time, ultimately weakening the resilience of many ecosystems to modern anthropogenic impacts (Jackson et al., 2001). The consequences of historical degradation of coastal ecosystems continue to be propagated forward, as lost habitat and degradation of water quality may limit species recovery (Lotze et al., 2006).

In the latter half of the 20<sup>th</sup> century, global catches increased as fisheries expanded into previously unfished regions (Swartz et al., 2010), significantly increased their effort in fished areas (Watson et al., 2013), and exploited deeper-water stocks (Morato et al., 2006). Beginning in the middle of the 1990s, however, total catches at

the global scale began to decline (Pauly & Zeller, 2016a), despite continued high effort and spatial coverage. Catches from newly-exploited stocks, found in deeper waters (Morato et al., 2006) and further offshore (Swartz et al., 2010), failed to make up for declines in established fisheries (Swartz et al., 2010). Two consecutive decades of reported global landings that remained generally flat in the face of increasing effort implied that broad limits to fisheries' exploitation had been reached (Worm & Branch, 2012). Heightened concern led to improved management in some parts of the world, and over the past decade, many fish stocks in areas with the most intensive fisheries management regimes have shown evidence of rebuilding (Hilborn et al., 2020). However, those stocks in regions with less effective management remain at high risk of overexploitation (Hilborn et al., 2020).

In the immediate future, global catches are more likely to decline than to show any sustained increase. The primary drivers for catch declines will be the reduced productivity of stocks as a result of overexploitation, and measures taken to restrict exploitation to allow stock rebuilding. In regions where effective management has recently halted overexploitation and allowed stocks to rebuild, catches have been substantially reduced (Hilborn et al., 2020). The adoption of ecosystem-based management policies will likely result in steeper reductions in allowable catch than would otherwise occur under traditional single-stock maximum sustainable yield (MSY)-focused management, in order to limit total removals from ecosystems (Patrick & Link, 2015), maintain sufficient availability of forage species in food webs (Smith et al., 2011), protect specific trophic linkages (e.g., the case of the sea lion conservation measures in the Eastern Bering Sea and Aleutian Islands groundfish fishery; NPFMC, 2017), conserve age and size structure in long-lived species (Berkeley et al., 2004), and/or protect at-risk stocks caught in fisheries for more abundant stocks (Vert-pre et al., 2013).

Projections indicate that climate change may shift catch potential offshore and towards the poles (Cheung et al., 2010). This projection indicates that catch potential may decline in areas where it is perhaps of greatest importance for both food

security and employment (i.e. nearshore waters in tropical and sub-tropical regions), and drive it instead to regions that are more easily accessed by large-scale, fuel- and capital-intensive fisheries (Sumaila et al., 2019). Climate change may also have consequences for the political palatability of marine conservation efforts, as the uncertainty associated with climate-related impacts may ultimately increase fishing interests' discount rate, leading them to prioritize immediate benefits over those in a less-certain future (Sumaila et al., 2011).

Going forward, some scientists see a general divergence between current and projected states of fisheries in different parts of the world (Hilborn et al., 2020). Catch restrictions in well-managed fisheries may allow overfished stocks to recover, but in other parts of the world, the lack of controls on exploitation implies the potential for continued decline (Hilborn et al., 2020). It has also been suggested that catch restrictions in the developed world may lead to a transfer of fishing capacity into less-managed waters (Worm & Branch, 2012). In either case (low biomass and tight restrictions in well-managed regions, or declining catches and lack of control in regions with weaker management), increases in the global catch are not to be expected in the near term.

### **1.3. Nutrition-informed fisheries policy and analysis**

The challenge is clear: while fisheries yields can play a key role in addressing micronutrient deficiencies (Hicks et al., 2019), declines in fisheries catches may put millions at risk of developing new deficiencies (Golden et al., 2016a). Increasing awareness of the importance of seafood to food security, especially in rural and/or developing regions, will likely require fisheries managers to explicitly plan for food security-related objectives (McClanahan et al., 2015). This may take on particular importance in those regions where fisheries management has been relatively lax to date, as improvements to management in those regions may lead to reduced catches, with consequences for nutrient yields.

Despite the rising visibility of this issue, nutrition information is not yet being incorporated into fisheries science and management approaches. Traditionally, the focus of fisheries science and management has been devoted almost entirely to questions of stock status and acceptable exploitation rates, with forays into questions of ecosystem status and change as deemed necessary. The lack of nutrient production-related methods and objectives in fisheries science and management may allow for blind spots in our understanding of fisheries, and these blind spots may in turn create the potential for fisheries decisions to have significant unforeseen consequences for the health of dependent human populations.

I contend that nutrition-informed fisheries science and management should, at a minimum, accomplish the following points of emphasis:

- A. Understand the scale of nutrient yields from different fisheries and taxa, both in absolute terms and in terms of human health contributions;
- B. Understand the ways in which temporal and/or ecological trends and patterns in nutrient yields may differ from those for landings,
- C. Identify the key taxa underpinning yields of the various nutrients;
- D. Monitor changes in points A, B, and C, and
- E. Understand the nutrition needs of dependent human populations, with an emphasis on the potential for fisheries' products to address specific micronutrient deficiencies.

The research presented in this dissertation is meant to be a first step toward addressing points of emphasis A-D. Furthermore, I also develop methods by which fisheries catches may be analyzed to identify options for maximizing nutrient yield relative to catch biomass. Such efforts are one potential approach to balancing conservation and nutrition concerns (Hicks et al., 2019).

## 1.4. Chapter summaries

In all, I developed four research chapters, organized around three simple research questions (corresponding points of emphasis for nutrition-informed fisheries, and chapter numbers, are indicated in parentheses):

1. **How might we analyze and monitor fisheries' nutrient yields?**

(Points A-D; chapter 2)

2. **What do we learn about fisheries, and their nutrient yields, when we do so?**

(Points A-C; chapters 2 and 3)

3. **How might fisheries' nutrient yields be optimized?**

(Additional point of emphasis for balancing conservation and nutrient yield objectives; chapters 4 and 5)

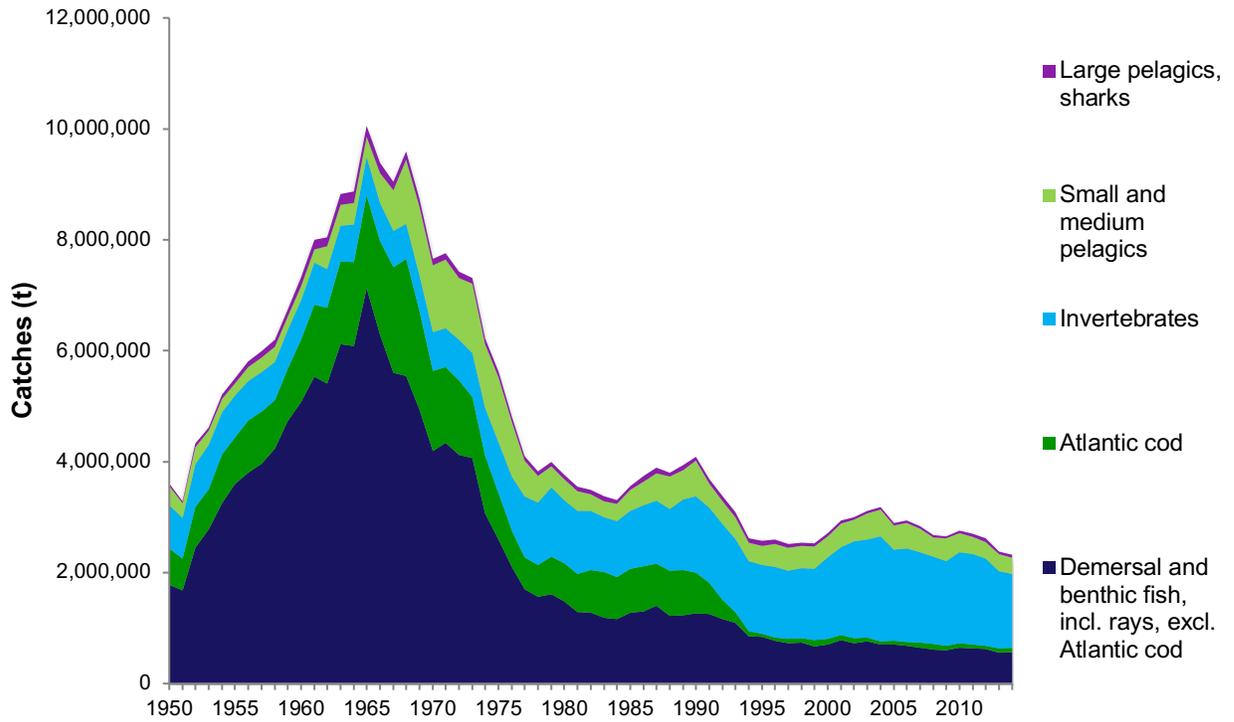
I chose to focus on the fisheries of the Northwest Atlantic Fisheries Organization (NAFO) region, as this region is a relatively data-rich area with a long and well-studied history of fisheries exploitation. The NAFO region has undergone a dramatic shift in ecosystem structure and fisheries landings over the past century. Previously, landings were dominated by groundfish, particularly Atlantic cod (*Gadus morhua*), but the fishery-driven collapse of groundfish stocks, and the trophic cascades that followed, shifted the trophic system (and fishery) from groundfish dominance to dominance by shellfish and small pelagics (Frank et al., 2005). Beginning in the mid-1960s, landings of groundfish declined sharply and landings of invertebrates increased substantially (Figure 1.2). I hypothesize that nutrient yields have changed more than landings, primarily due to the substantial differences in edible food yield rates across the relevant taxa.

In Chapter 2, I pursue research questions 1 and 2 by analyzing the yields of a suite of nutrients obtained from the NAFO region for the years 1950-2014. In that chapter, I develop and present several novel approaches, methods, and indicators. These include the first use of Simpson's evenness ( $E$ ) as an indicator of nutrient yield distribution across taxa, an indicator that I call " $B_{80}$ " as an index of nutrient

distribution across biomass, and nutrient-specific mean trophic levels and trophic distribution patterns as a means of identifying trends and patterns in the ecological origins of nutrients.

In Chapter 3, I focus on research question 2, by modeling the consequences that use of Atlantic herring (*Clupea harengus*) bait has for the net nutrient yields of the Maine fishery for American lobster (*Homarus americanus*). Results indicate that this lobster fishery's bait use likely renders it a net consumer of most assessed nutrients. To my knowledge, this is the first study to conclude that a fishery may be a net consumer, rather than producer, of nutrients. As such, this fishery has apparent parallels to terrestrial and aquaculture animal production systems that have come under scrutiny in recent years for their relatively inefficient use of resources.

Research question 3 is addressed in Chapters 4 and 5. I develop a theoretical approach to optimizing nutrient yields relative to catch biomass in Chapter 4, and apply it to NAFO fisheries in Chapter 5. The approach developed and applied in these two chapters is meant to maximize the difference between nutrient yields and catch biomass, and does so by identifying the set of taxa that are "optimal" and "sub-optimal" for a given nutrient, based on their contributions to the nutrient's yields vs. their contributions to catch biomass. The results of this optimization approach indicate that in multispecies contexts, nutrient yields may be largely maintained even in the face of substantial reductions to total catches. It is my hope that these two chapters may provide ideas that will be useful in efforts to balance human nutrition concerns with conservation objectives.



**Figure 1.2.** Reconstructed catch data (landings + discards) for the Northwest Atlantic (FAO Area 21), 1950-2014. Data from (Pauly & Zeller, 2016b).

## **Chapter 2. Analyzing nutrient yields in fisheries: the Northwest Atlantic, 1950-2014**

### **2.1. Introduction**

International goals for ending both hunger and malnutrition (United Nations Sustainable Development Goal 2; DSDG, 2019) and overfishing (Aichi Biodiversity Target 6; CBD, 2019) (United Nations Sustainable Development Goal 14; DSDG, 2019) are fundamentally interrelated. Micronutrient deficiencies are estimated to affect up to 2 billion people worldwide (Muthayya et al., 2013), and seafood plays a particularly important role in providing these nutrients to at-risk populations (Béné & Heck, 2005; Thilsted et al., 2016). However, approximately 33% of assessed fish stocks are over-exploited and nearly 60% are fully exploited and cannot withstand additional fishing pressure (FAO, 2018). To end overfishing and rebuild overfished stocks, potentially lengthy periods of reduced catches are required (Worm et al., 2009), but declines in fisheries catches may put millions at risk of micronutrient deficiencies (Golden et al., 2016a). Increasing awareness of this interrelatedness of fisheries decisions and human nutrition suggest that fisheries policy will be informed by nutrition considerations (Thilsted et al., 2016). However, while there has been recent progress made in understanding the nutrient contents of fisheries (Hicks et al., 2019), there are no methods or indicators specifically designed to support the evaluation and management of fisheries for nutrient-related outcomes.

Here, we present a suite of analyses and indicators by which trends and patterns in fisheries' nutrient yields may be assessed, monitored, and broadly understood. We illustrate the use of these methods by applying them to evaluate the yields of protein, iron, zinc, vitamin A, vitamin B<sub>12</sub>, and the omega-3 fatty acids eicosapentaenoic acid (EPA) and docosahexaenoic acid (DHA) obtained from fisheries landings in the Northwest Atlantic Fisheries Organization (NAFO) region from 1950 through 2014. Here, we report on nutrient yields from landings (i.e., retained catch) only.

## 2.2. Methods

This study consisted of two broad components: a) estimating the yields of the assessed nutrients from annual NAFO fisheries landings, and then b) developing methods and indicators to assess potentially informative aspects of the resulting nutrient yield estimates.

The first step was to estimate how much of each selected nutrient was yielded by landed NAFO catches in each year. In this step, landings of major species were first converted to estimates of edible biomass, and then to estimates of the amount of each nutrient embodied in the edible biomass. Catch data were obtained from the Sea Around Us database (SAU RFMO 10 v45) (Pauly & Zeller, 2016b). Taxa were initially selected for inclusion based on their contributions to the first 95% of total catches (i.e., retained catches + discarded catches) over the 1950-2014 period. This initial selection resulted in the inclusion of 44 species or species groups. Of these, three generic species groups were of insufficient taxonomic resolution to allow for reliable estimates of edible biomass or nutrient yield; these three were excluded from the analysis. Atlantic menhaden (*Brevoortia tyrannus*) were also excluded due to their functional inedibility to humans (see Appendix A for selection criteria and taxon-specific information). One taxon, goosefishes (Lophiidae) was added, as it had become an increasingly important component of catches in the latter decades of the assessed period, despite not contributing to the first 95% of total catches across the entire period. Finally, catches of one taxon (American lobster, *Homarus americanus*), were differentiated by region (USA, Canada, and other nations), as lobster landings in these regions differ in molt stage and thus edible biomass yield. The result of these exclusions and inclusions was that 43 taxonomic groups were included in the analysis; see Appendix A.1 for details regarding the selection of taxa, and Appendix B for the list of all taxa included in the study.

Landings weights were then converted to estimates of edible biomass, via edible biomass conversion rates. Edible biomass estimates were then converted to estimates of the nutrients yielded by that edible biomass, via nutrient conversion

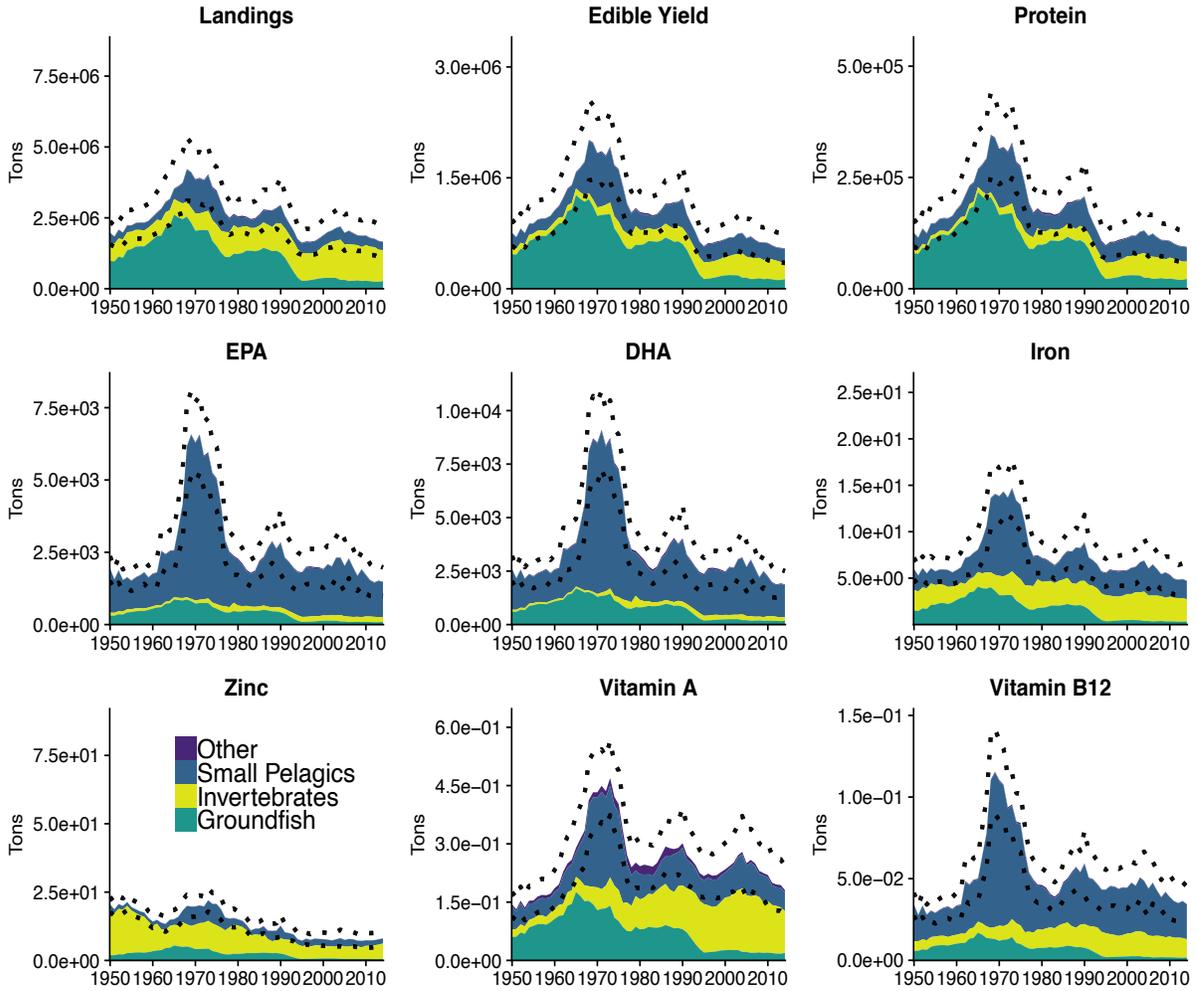
rates. Wherever possible, taxon-specific conversion rates were used; however, data gaps sometimes required that estimates be derived from similar taxa. Taxon-specific details for conversion rates and gap-filling procedures are presented in Appendix A.2. Uncertainty estimates were derived as described in Appendix A.3. All taxon-specific conversion rates are shown in Appendix B.

A number of subsequent analyses were performed to provide specific insights. Nutrient yields were contextualized by quantifying the number of people who could, in theory, be supported by those yields for one year; this was done by applying Recommended Daily Amount (RDA) values, or similar estimates, to the estimates of nutrient yields (see Appendix A.4 and Appendix C). To evaluate nutrient yield distributions across taxa, biomass, and ecosystem components, three indicators were developed and evaluated: Simpson's evenness ( $E$ ), nutrient distribution across trophic levels, and a novel indicator ( $B_{80}$ ), respectively. The methods used to arrive at these indicators are described in Appendices A.5, A.6, and A.7, respectively.

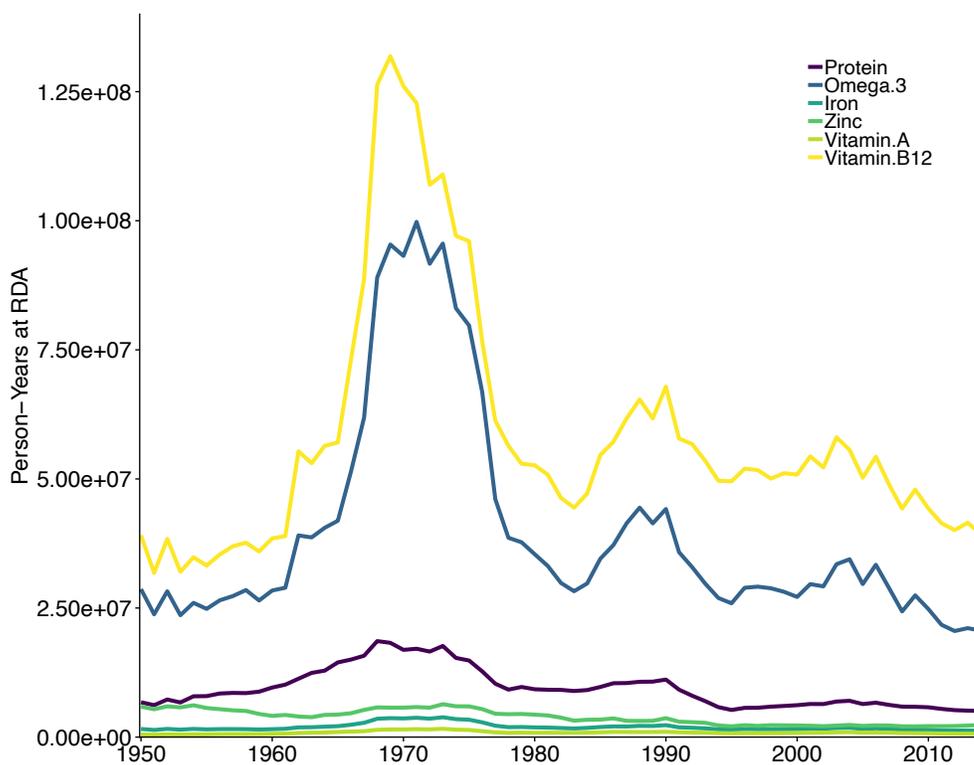
## **2.3. Results**

### **2.3.1. Nutrient yields, trends, and key taxa**

As the first step, yields of nutrients were quantified and then contextualized in terms of human dietary needs. In absolute amounts, NAFO fisheries' recent annual yields of the assessed nutrients ranged over six orders of magnitude, from tens of kilograms of vitamin B<sub>12</sub> to tens of thousands of tons of protein (Figure 2.1). When expressed relative to human dietary needs, this range contracts to two orders of magnitude: NAFO fisheries' recent annual nutrient yields could theoretically support between  $7 \times 10^5$  (vitamin A) to  $4 \times 10^7$  (vitamin B<sub>12</sub>) adult humans for one year at recommended daily consumption rates (Figure 2.2). Thus, of the assessed nutrients, NAFO fisheries' most significant yields relative to human dietary needs were vitamin B<sub>12</sub> and the omega-3 fatty acids. Protein, the nutrient that is often cited when nutrient outputs of fisheries are discussed, has in recent years been produced in quantities sufficient to support approximately  $5.5 \times 10^6$  people for one year (Figure 2.2).

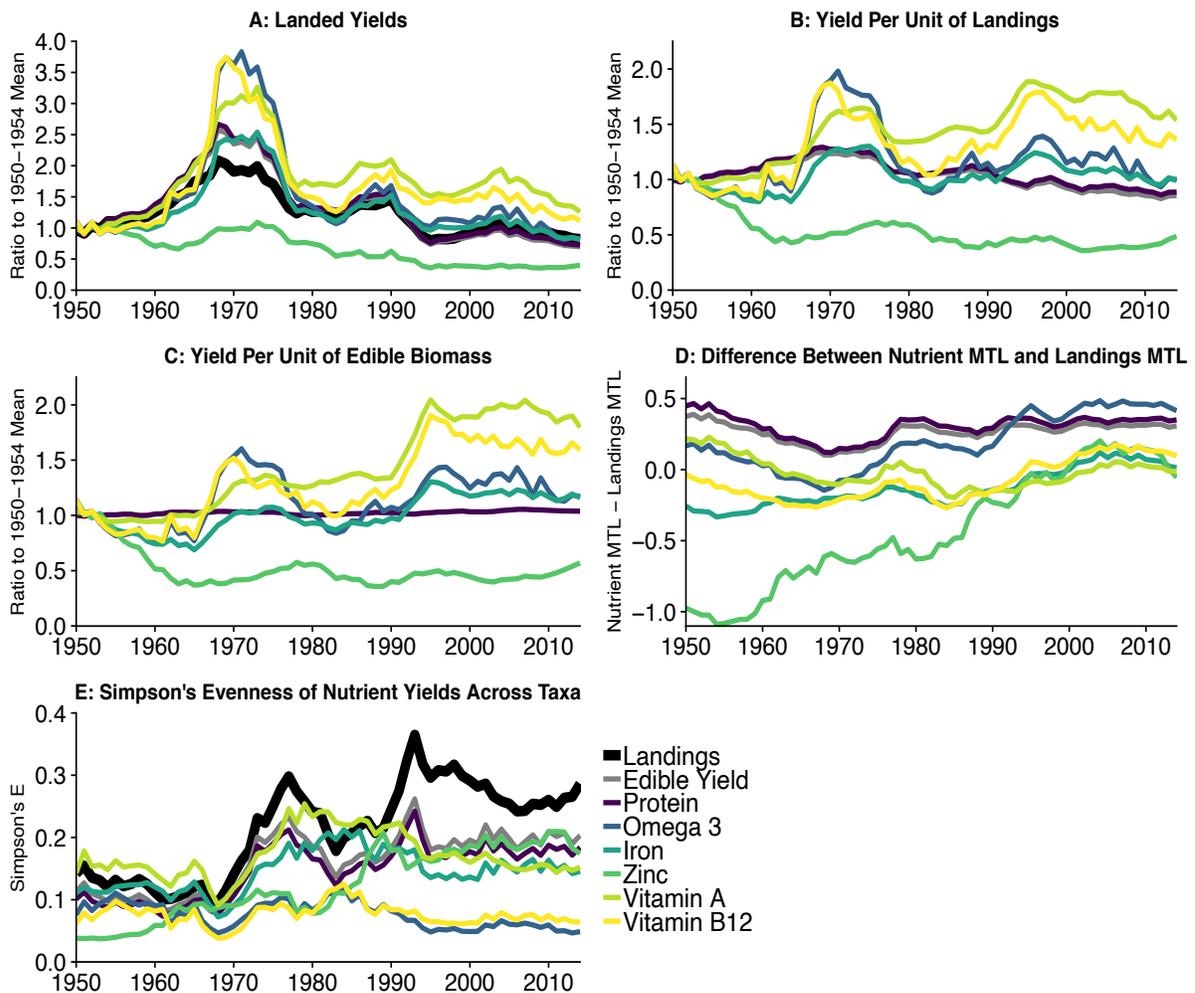


**Figure 2.1.** Trends in NAFO landings weights and nutrient yields from major taxonomic groups, 1950-2014. To facilitate ease of comparison of relative change in yields across nutrients, all vertical axes are scaled to a maximum value of 4.5 times the value in the first year (1950). Dotted lines indicate upper and lower uncertainty estimates for the totals (approximate confidence level = 95%; see Appendix A).



**Figure 2.2.** Landed yields of selected nutrients expressed as number of person-years that yields could support at relevant RDAs or, for omega-3s, suggested intake levels. EPA and DHA are combined, as per (FAO/WHO, 2008).

Trends in nutrient yields were compared to trends in landings weights to identify patterns and areas of substantial divergence. Yields of most nutrients increased more, and subsequently declined further, than total landings (Figure 2.1, Figure 2.3A). The maximum single-year landings value was approximately 2.1 times the 1950-1954 mean annual landings; in comparison, the maximum single-year yields of all nutrients other than zinc were 2.5-3.8 times their respective 1950-1954 means. Declines from these maximum values were greater for most nutrients than for landings: while the mean 2010-2014 landings was approximately 42% of the maximum single-year landings, mean 2010-2014 yields of all nutrients other than vitamin A were comparatively more reduced (22% - 34%; Figure 2.3A).



**Figure 2.3.** Trends in NAFO nutrient yields, 1950 – 2014.

(A) Changes in landings, edible biomass, and nutrient yields, relative to 1950-1954 mean.

(B) Nutrient yields per unit of landings, relative to 1950-1954 mean.

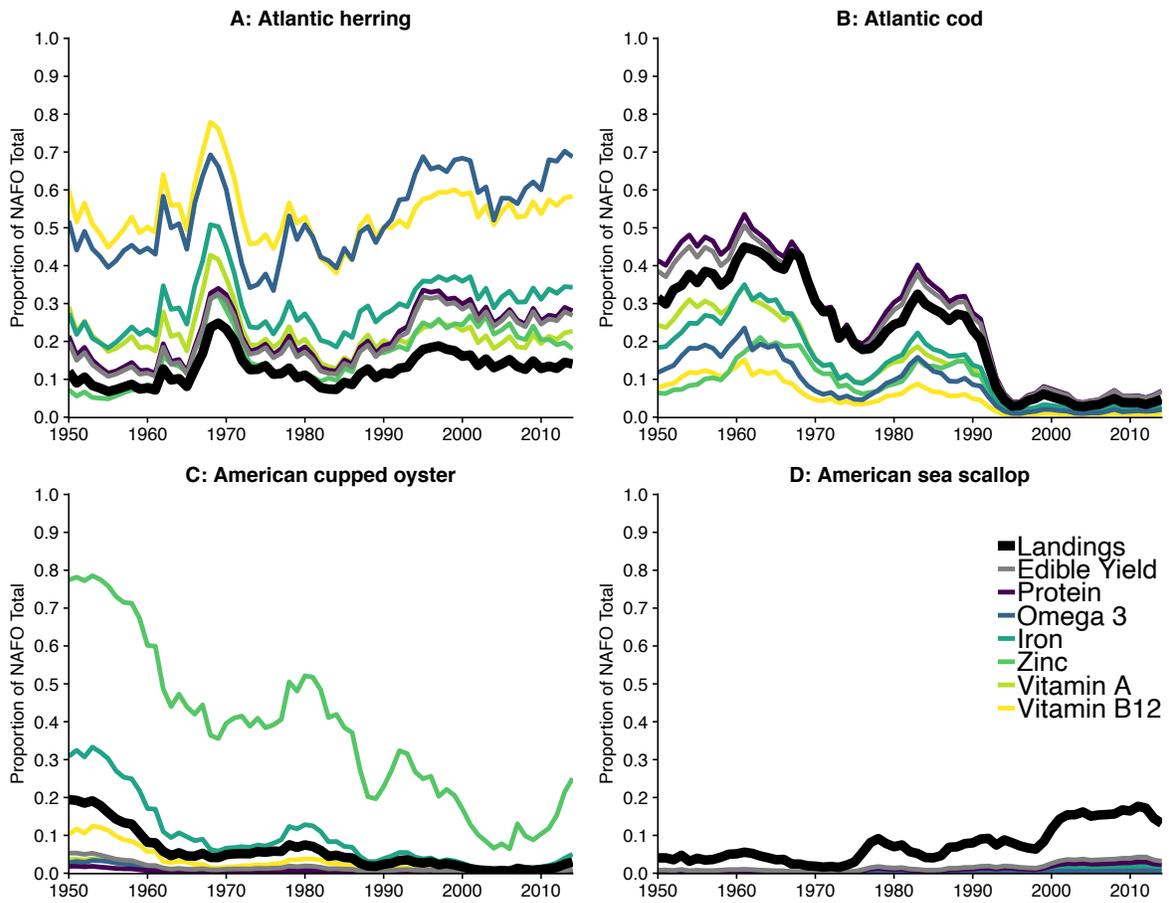
(C) Nutrient yields per unit of edible biomass, relative to 1950-1954 mean.

(D) Differences between nutrient mean trophic levels and landings mean trophic level. Positive values indicate that a nutrient is mainly derived from species relatively higher in food webs.

(E) Evenness (Simpson's  $E$ ) of the distribution of nutrient yields across taxa. Higher values represent more even contributions across species. Minimum possible  $E$  ( $N=43$ ) is 0.023.

Nutrient yields per unit of landings fluctuated substantially for multiple nutrients (Figure 2.3B). Notably, the ratio of edible biomass to landings exhibited a gradual yet consistent decline since the 1970s. Figure 2.3C shows nutrient yields corrected for this trend in edible biomass; by the end of the assessed period, each unit of edible biomass produced by NAFO fisheries yielded approximately 1.9 times as much vitamin A as at the start of the study period, and more vitamin B<sub>12</sub> (approximately 1.6x), omega-3 fatty acids (1.2x), and iron (1.15x). Conversely, yields of zinc per unit of edible biomass declined sharply over the first decade in the assessed period, to approximately ½ of its mean 1950-1954 value, and remained relatively steady at this level.

While landings were first dominated by groundfish and then by invertebrates, small pelagic fish consistently made outsized contributions to NAFO nutrient yields (Figure 2.1). The relatively high contributions of small pelagics to NAFO nutrient yields were driven by Atlantic herring (*Clupea harengus*) (Figure 2.4A). Atlantic herring constituted approximately 14% of average annual NAFO landings during the years 2010-2014, but contributed 20% - 70% of average annual yields of the various nutrients. The relative importance of Atlantic herring to NAFO nutrient yields is underscored by a comparison to Atlantic cod (*Gadus morhua*) (Figure 2.4B). Prior to the collapse of the cod fisheries in the 1990s, cod landings were annually responsible for 20% to nearly 45% of NAFO's landed biomass, and were routinely two to five times greater than Atlantic herring landings. Even so, herring supplied a much higher proportion of NAFO's total omega-3 and vitamin B<sub>12</sub> yields, and broadly comparable amounts of, vitamin A, iron, and zinc; the contributions of pre-collapse cod were consistently greater than herring for only edible biomass and protein. Since the cod collapse, herring contributed far more to NAFO's yields of all assessed nutrients (Figures 2.4A and 2.4B).



**Figure 2.4.** Contributions of four notable species to NAFO-wide landings (solid black line), edible biomass, and nutrient yields: (A) Atlantic herring, (B) Atlantic cod, (C) American cupped oyster, and (D) Atlantic sea scallop. The vertical axis represents the proportion of NAFO's total.

Species that appear to be relatively minor from a landings weight perspective may make substantial contributions to yields of specific nutrients, as illustrated by American cupped oyster (*Crassostrea virginica*) and zinc (Figure 2.4C). In the first years of the assessed period, oysters accounted for approximately 20% of NAFO landings, but contributed over 75% of NAFO's zinc yields. In subsequent years, declining landings of cupped oysters led to reduced NAFO-wide zinc yields (Figure 2.1), but despite low landings, the species continued to contribute importantly to zinc yields (Figure 2.4C). In contrast, some species contribute comparatively less to nutrient yields than to landings weights: while landings of American sea scallop (*Placopecten magellanicus*) rose rapidly after 2000 and annually accounted for 13% to nearly 18% of total NAFO landings in the years 2010-2014, they did not contribute more than 3.2% of any assessed nutrient during those years (Figure 2.4D).

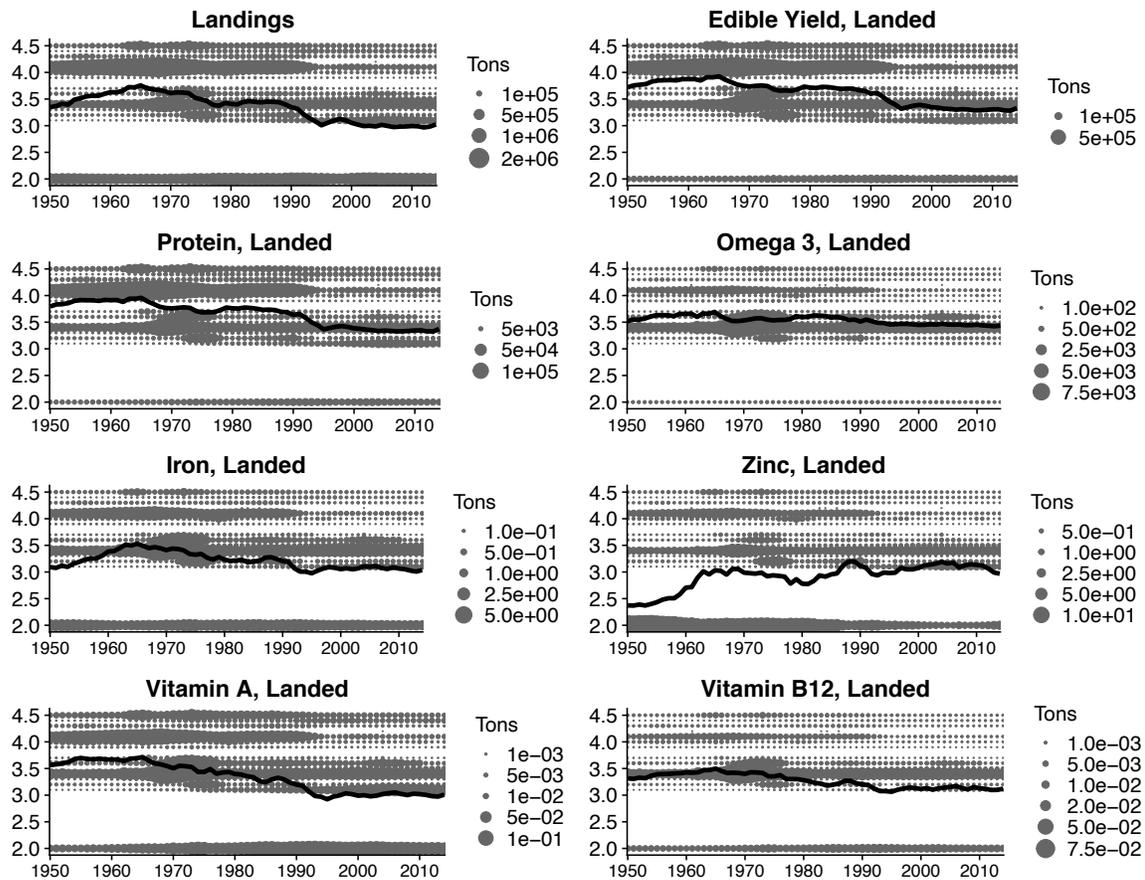
### 2.3.2. Nutrient yield indicators

In order to understand how the marine ecosystem underpins yields of different nutrients, and to monitor changes in these patterns over time, we assessed nutrient distributions across trophic levels (Figure 2.5) and calculated nutrient-specific mean trophic levels (Figure 2.3D). Nutrient yields were differentially obtained across trophic levels. Relative to landings, both protein and the omega-3 fatty acids relied to a greater degree on high trophic level finfish. Zinc demonstrated a unique pattern, with a strong reliance on low trophic level invertebrates in the earliest years, giving way to a broader distribution across trophic levels even as total zinc yields declined (Figure 2.1).

High reliance on a small number of taxa to supply a nutrient may indicate a lack of system-wide resilience (Elmqvist et al., 2003), and thus may represent a scenario that threatens human nutrition. Here, we use Simpson's evenness ( $E$ ) (Morris et al., 2014)(Appendix A.5) as an indicator of the distribution of nutrient yields across exploited taxa over time, with a higher  $E$  value representing a greater degree of evenness across taxa (Figure 2.3E). NAFO landings generally became more evenly distributed across the assessed taxa over time, but nutrient yields remained

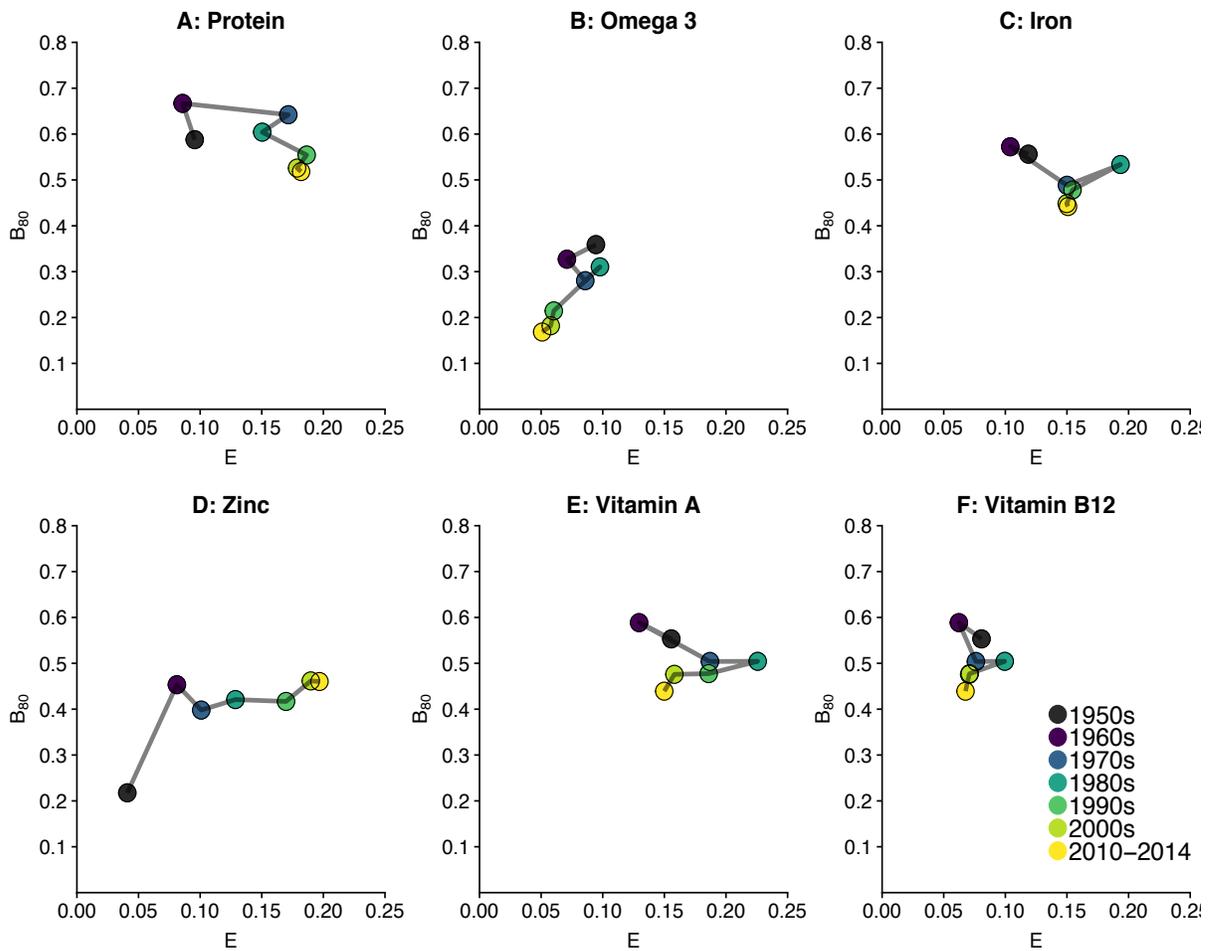
comparatively more concentrated in fewer taxa. Notably, yields of the omega-3 fatty acids and vitamin B<sub>12</sub> remained particularly concentrated in few taxa, with recent  $E$  values (0.046-0.075) that approach the minimum (0.023 for N=43). Similarly low  $E$  values were observed for zinc yields at the start of the study period due to zinc's reliance on oysters (Figure 2.4C); the subsequent decline of zinc that accompanied reduced oyster landings serves as an illustration of both the consequences of high reliance on one taxon, and the potential utility of the  $E$  indicator to identify situations of low resilience.

Balancing conservation and nutrition concerns may be possible by catch reductions that are strategically planned to minimize effects on nutrient yields (Hicks et al., 2019). The potential for such optimization may be highest when the yields of a nutrient are highly concentrated in smaller amounts of total landings biomass. Here, we develop an index,  $B_{50}$ , to indicate the degree to which nutrients are distributed across landings biomass. This index is the ratio of the biomass required to produce the first 50% of the nutrient's yields (assuming yields are sequentially obtained from the highest-yielding taxa to the lowest) to total landings biomass (Appendix A.7). The index thus declines as the nutrient yields become increasingly concentrated in less biomass.



**Figure 2.5.** Distribution of nutrient yields across trophic levels, and mean trophic level (black line).

Viewing  $B_{80}$  in conjunction with  $E$  conveys key summary information about the status of nutrient yields (Methods) (Figure 2.6). For example, yields of the omega-3 fatty acids were consistently concentrated in few taxa (low  $E$ ) and were primarily derived from a small proportion of total landings biomass (low  $B_{80}$ ). A different pattern is evident for protein, which has become more evenly distributed across taxa, but more concentrated in biomass. Zinc yields have become more broadly distribution across both taxa and biomass, but yields of iron, vitamin A, and vitamin  $B_{12}$  have become increasingly concentrated in taxa and biomass in recent decades.



**Figure 2.6.** Nutrient evenness across taxa and concentration in biomass, 2010-2014 means. The x axis shows evenness of nutrient yields across taxa (Simpson's  $E$ ), the y axis shows the proportion of total NAFO landings biomass required to produce the first 80% of the nutrient's yields ( $B_{80}$ ). Simpson's  $E$  increases as nutrient yield becomes more evenly distributed across taxa, while  $B_{80}$  increases as the first 80% of nutrient yields become more evenly distributed across biomass.

For the most recent years in the study (2010-2014), the  $B_{80}$  index shows that NAFO landings could be substantially reduced while minimizing effects on nutrient yields. For iron, zinc, vitamin A, and vitamin B<sub>12</sub>, mean 2010-2014 landings could be reduced by more than 50% while maintaining mean annual yields at approximately 80% of totals. The difference was even greater for the omega-3 fatty acids, as total landings could be reduced by more than 80% while maintaining approximately 80% of yields. Protein showed the least potential for strategic landings reductions, but even then, protein yields could be maintained at 80% with landings reductions of nearly 50%.

There are two key caveats to these potential landings reductions. First, they are nutrient-specific: the taxa for which landings would be maintained or reduced vary from nutrient to nutrient, and thus a scenario that strategically reduces landings while minimizing effects on one nutrient may not lead to a desirable outcome for other nutrients. Secondly, the act of reducing landings of multiple low-yielding species to focus on a subset of high-yielding species may have implications for the evenness of nutrient distribution across taxa, and thus the ongoing resilience of nutrient provision.

## **2.4. Discussion**

The results of this study carry several notable implications for our understanding of global fisheries. First, trends in landings weights may substantially underrepresent changes in nutrient yields, and thus trends in landings weights cannot be assumed to represent trends in nutrient yields. Second, nutrient yields may be highly dependent upon small subsets of fished taxa, and may remain so even if landings become more evenly distributed across taxa. Third, species that appear to be minor from a landings perspective may make outsized contributions to yields of specific nutrients. Shifting species composition of landings can thus result in substantial fluctuations in total nutrient yields, yields per unit of landings, and yields per unit of edible biomass. Thus, an ecosystem regime shift could have greater impacts on nutrient yields than on landings weights alone.

The example of zinc yields and American cupped oysters suggests that a high degree of reliance on one taxon for nutrient yields may indicate a system-wide lack of resilience, with declines in that taxon leading to reductions in nutrient availability. Currently, NAFO's yields of multiple nutrients may be at risk of declining due to recent, sharp downturns in Atlantic herring recruitment, spawning biomass, and catches (NEFSC, 2018a). Such scenarios may represent particularly high-risk situations for human health, and thus the identification of a disproportionate reliance on one taxon may be a key benefit of nutrient-specific analysis of fisheries yields.

The central role that Atlantic herring play in NAFO-wide nutrient yields highlights the global issue of using such small pelagic "forage fish" for purposes other than direct human consumption. Globally, landings of small pelagic species, including Atlantic herring, are often used to produce fishmeal and fish oil (Shepherd & Jackson, 2013). Most of the biomass used for fishmeal and fish oil is composed of food-grade fish (Cashion et al., 2017), and small pelagic species may contain higher concentrations of certain nutrients (e.g., omega-3 fatty acids; Hicks et al., 2019). The use of such species for uses other than direct human consumption may thus represent a waste of nutrients.

Preference for consuming some parts of the animal but not others could have a substantial influence on the nutrients yielded by that animal. For example, the relatively low contributions of sea scallops to total NAFO nutrient yields, and the relatively high contributions of oysters to zinc yields, may be due to the parts of the animal that are generally consumed. This study analyzed yields only from published estimates of what is termed "edible flesh" and did not account for consumption of other parts of the animal (e.g., roe, eyes, skin, bones, etc.), nor foods derived from these products (e.g., fish stews).

Finally, heterogeneity in nutrient concentrations and edible biomass yields across taxa presents an opportunity for selectively reducing catches to facilitate stock

rebuilding and conservation while minimizing effects on region-wide nutrient yields, with careful consideration of the ongoing resilience of nutrient yields. Considering the importance of this issue for both human health and biodiversity conservation, there is a need for advanced optimization procedures capable of incorporating information and objectives across multiple nutrients of concern for dependent populations, as well as improved information for edible biomass yield and nutrient content rates.

## **Chapter 3. Nutrient yield consequences of the use of Atlantic herring (*Clupea harengus*) as bait in the Maine fishery for American lobster (*Homarus americanus*)**

### **3.1. Introduction**

The global food system is one of the primary drivers of large-scale environmental change (Rockstrom et al., 2009). Spatially, our efforts to obtain food dominate both land and sea: nearly 40% of the earth's ice-free land is estimated to be devoted to crops and pastures (Foley et al., 2011), and fisheries' operations cover more than half of the oceans' surface waters (Watson et al., 2013). On land, agriculture drives the alteration of ecosystems, affects geochemical cycles, and appropriates substantial amounts of available fresh water (Foley et al., 2005). In the oceans, fisheries are associated with a host of environmental impacts, including habitat impacts and the incidental catch of non-targeted species (Chuenpagdee et al., 2003), reductions in the abundance of large species (Christensen et al., 2003; Worm et al., 2009), and overfishing of targeted stocks (Worm & Branch, 2012). Thus, at the global scale, land use change (primarily driven by agriculture) and direct exploitation of marine species have been identified as the primary drivers of negative impacts on terrestrial and marine ecosystems, respectively (Diaz et al., 2019).

Against this backdrop, the continued use of agricultural land and resources to produce animal products has come under criticism due to its apparent inefficiency. At its most basic, the issue is one of thermodynamics: from a dietary energy perspective, growing plants to feed to animals, and then eating the animals, is less efficient than simply eating the plants ourselves. While methods and estimates vary, the conclusion that can be drawn from multiple studies on the subject is that the use of crops to feed animals results in the loss of vast amounts of dietary energy that could, in theory, be used to support billions of people (Foley et al., 2011; Cassidy et al., 2013; West et al., 2014).

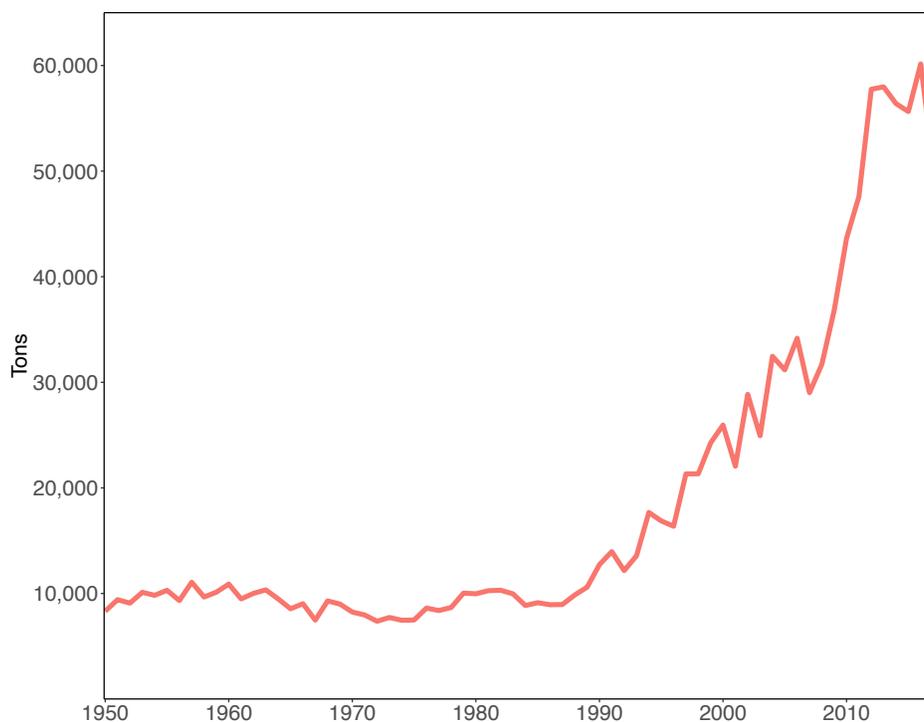
An analogous criticism has been levied at those aquaculture systems that use fishmeal and fish oil derived from wild-caught fish. Globally, aquaculture is the primary consumer of fishmeal and oil (Tacon & Metian, 2015). Fishmeal and fish oil are primarily derived from small pelagic fish (Shepherd & Jackson, 2013). Small fish can contain high concentrations of important nutrients (Roos et al., 2007; Hicks et al., 2019), however, and much of the biomass used for fishmeal and fish oil is composed of food-grade fish (Cashion et al., 2017). The use of potentially edible wild-caught species to provide fishmeal and fish oil for aquaculture operations has thus been a source of criticism of this fast-growing sector (Deutsch et al., 2007).

To date, criticisms of inefficiency in fisheries have focused on clear-cut examples of waste (e.g., the highly visible issue of incidental catch and discarding of marine species), rather than inefficient levels of edible inputs. To a degree, this is understandable: while inputs of feed to animal agriculture and much aquaculture are obvious requirements of those systems, fisheries appear to require no intentional inputs of anything that could be edible to humans. However, there is a potentially analogous situation that occurs in some fisheries: the use of edible species as bait in fisheries that target high-value species.

Some fisheries use substantial amounts of bait that could otherwise be consumed by humans, and therefore consume nutrients in the process of producing nutrients. In the state of Maine, the trap fishery for American lobster (*Homarus americanus*; hereafter referred to as “lobster”) use substantial amounts of Atlantic herring (*Clupea harengus*; hereafter referred to as “herring”) for bait. In 2006, for example, the Maine lobster fishery used approximately 3 tons of bait for every ton of lobster that it produced, with the vast majority of bait being composed of herring (Driscoll et al., 2015). Given the importance of herring to the yields of many nutrients across the entire northwest Atlantic region, demonstrated in Chapter 2, the Maine lobster fishery’s consumption of herring bait may represent a scenario in which a fishery consumes substantial amounts of nutrients that could otherwise be available to humans.

### 3.1.1. The Maine lobster fishery

The Maine lobster fishery dominates USA landings of American lobster. From 2008-2013, landings from the state of Maine accounted for approximately 82.4% of the USA's lobster landings (ASMFC, 2015). After a long period of relative stability, annual landings in the Maine lobster fishery have increased substantially over the past three decades, increasing from approximately 10,000 tons/year in the late 1980s to over 50,000 tons by 2012 (Figure 3.1). This is one of the United States' most lucrative fisheries: from 2005-2017, landings for this fishery accounted for approximately 1.1% of the USA's domestic landings by biomass, but were worth 7.4% of USA domestic fisheries' landed value (NOAA Fisheries, 2019).



**Figure 3.1.** Maine lobster landings, 1950-2017. Data source: (Maine DMR, 2018)

### 3.1.2. Maine lobster and Atlantic herring: a complex relationship

The Maine lobster fishery and the New England Atlantic herring fisheries are deeply intertwined with one another, as the lobster fishery depends heavily on the herring

fishery to supply most of its bait. This has not always been the case, however. In the 1930s, the advent of the fishery for redfish (*Sebastes* spp.), a slow-growing groundfish, provided the lobster fishery with a preferred bait in the form of redfish “racks”, the skeletons and associated matter that remained after processing of the catch (Dow, 1961). The introduction of redfish rack bait coincided with a rapid expansion of fishing effort and landings in the USA lobster fishery (Dow, 1961). Racks remained the primary lobster bait for several decades; Acheson (1975) notes that the Maine lobstermen of that era used fish racks for bait. All told, the racks of redfish and other fish, along with some use of suckers and menhaden, were the primary bait used in the Maine lobster fishery until approximately 1980 (Acheson, pers. comm.). Declining redfish landings in the 1980s encouraged the lobster fishery to shift toward using Atlantic herring for bait (NEFMC, 1999b). Atlantic herring thus became the primary source of bait for this fishery in the early 1980s and has remained the most popular bait ever since (Acheson, pers. comm.). The scale of herring bait inputs to New England’s coastal waters via the modern lobster fishery is such that it may provide a nutrition subsidy that indirectly supports lobster growth (Grabowski et al., 2010).

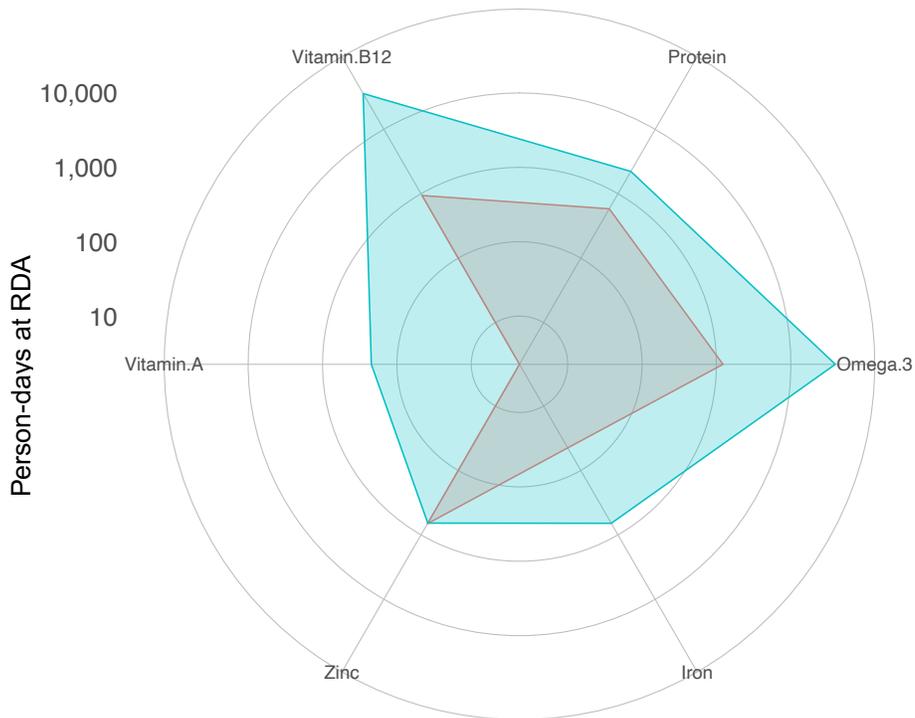
### 3.1.3. Nutrient profiles of American lobster and Atlantic herring

American lobster and Atlantic herring differ in both the amount of edible biomass that they yield per unit of landings, and in the nutrition profiles of that edible biomass.

The Maine lobster fishery primarily catches lobsters in the later summer and early fall months, when the catch is primarily composed of recently-molted “soft-shell” lobster (Holland, 2011). Soft-shell lobsters yield less edible biomass per unit of landings than do “hard-shell” lobsters, generally caught in colder months, which have had time to grow to fill out their shells. Generally, soft-shell lobsters are said to yield approximately 12.5%-18%, while hard-shell lobsters yield approximately 20-28%. Dow and colleagues, for example, estimated that soft- and hard-shell lobsters yielded 12.5% and 25%, respectively (Dow et al., 1975). In comparison to lobster, Atlantic herring have a much higher edible biomass yield rate, as approximately 63%

of the Atlantic herring's biomass is edible flesh (FAO, 1989).

As a result of its higher edible biomass yield rate and generally higher nutrient content rates, each ton of herring can in theory provide higher yields of most assessed nutrients, compared to a ton of lobster. Figure 3.2 compares the nutrients yielded by one ton of landings of both herring and lobster, with the nutrients characterized in terms of the number of person-days at the nutrient-specific recommended daily amount (RDA) or equivalent reference intake (see Appendix C for RDA estimates). The person-days that one ton of herring landings could support, in theory, are substantially greater for five of the six nutrient categories (note that EPA and DHA are combined as omega-3 fatty acids, and furthermore note the log scale, and thus the twin implications that small differences on the axes indicate large differences in yield and that this effect increases with distance from the center). In the case of omega-3s and vitamin B<sub>12</sub>, one ton of Atlantic herring catch provides enough nutrients to supply more than one order of magnitude more humans at the relevant daily amount than does lobster (Figure 3.2). The one nutrient that Atlantic herring does not yield at a greater rate is zinc, for which American lobster is the slightly more nutritious species. Thus, per unit of biomass, Atlantic herring appear to have substantially higher yields of several nutrients when compared to American lobster.



**Figure 3.2.** Number of person-days at relevant RDA or equivalent measure that could be supplied from nutrients yielded from one ton of landings. Note that this figure uses a log scale, and thus each concentric ring represents a change of one order of magnitude.

#### 3.1.4. Research questions

The nutrient profiles of herring and lobster suggest that the use of the former as bait may reduce the net nutrient yields associated with the latter. Thus, in this chapter, I evaluate the consequences of bait use for the net nutrient yields from the Maine lobster fishery. More specifically, this chapter explores two research questions:

1. What are the consequences for this fishery's net yield of nutrients if its consumption of nutrients via bait biomass is taken into consideration?
2. How does the net yield of nutrients differ across nutrients?

### 3.2. Methods

There are three components to this study:

1. A model of the amount of herring used as bait in the lobster fishery,

2. Estimates of the gross nutrient yields embodied in lobster landings and herring bait, and
3. Estimates of the net nutrient yields from the lobster fishery, found by subtracting the nutrients embodied in the herring bait biomass from the nutrients embodied in the lobster biomass.

### 3.2.1. Model of herring bait use in the Maine lobster fishery

#### 3.2.1.1. *Published estimates*

There is no single time series of data for the Maine lobster fishery's bait use. A review of available literature returns only four recent single-year estimates for Atlantic herring use as bait in the Maine lobster fishery (Table 3.1). The sources for the estimates for 1981, 1995, and 1996 cite primary sources that are no longer available, and did not include any information regarding underlying methods or assumptions. It should be noted that for the 1995 and 1996 estimates, only those herring that were landed in Maine were considered. Since the Maine lobster fishery uses herring landed in other New England states, these are likely underestimates of total herring use in the lobster fishery in those years; indeed, the reference for these estimates (NEFMC, 1999a) states that "most observers" would have estimated that 50,000-70,000 tons of herring were used as bait in those years. The 2006 estimate was obtained from survey responses of Maine lobster license holders (n=83; Driscoll et al., 2015).

In addition to these four estimates of herring use in the Maine lobster fishery, several other estimates are available for total bait use of all types and/or for bait use across lobster fisheries in New England (Table 3.2).

**Table 3.1.** Estimates of Atlantic herring use as bait in the Maine lobster fishery.

<b>A</b>	<b>B</b>	<b>C</b>	<b>D</b>	<b>E</b>
<b>Year</b>	<b>Atlantic herring used as bait (t) in Maine lobster fishery</b>	<b>USA Atlantic herring landings<sup>1</sup> (t)</b>	<b>% of USA Atlantic herring landings used as bait in Maine lobster fishery</b>	<b>Source for bait amount estimate</b>
1981	1,460	65,092	2.2	(NEFMC, 1999b)
1995	24,000**	68,953	34.8	(NEFMC, 1999a)
1996	31,755*	88,703	35.8	(NEFMC, 1999a)
2006	88,395	120,833	73.2	(Driscoll et al., 2015)

1 = (NOAA Office of Science and Technology, 2019)

2 = (Maine DMR, 2018)

\* = Herring landings from Maine only

+ = Estimate derived from the 1996 estimate, which according to the source was approximately 8,000 tons more than the 1995 value

**Table 3.2.** Estimates of bait use (all types) across New England/Northeast USA lobster fisheries

<b>Year</b>	<b>Bait use estimate (t) and bait type(s)</b>	<b>Regional scope for bait use estimate</b>	<b>USA Atlantic herring landings<sup>1</sup> (t)</b>	<b>% USA herring landings used in lobster fishery for that region</b>	<b>Source for bait amount estimate</b>
<b>1975</b>	18,000 – 22,000 (all types)	Maine			(Dow et al., 1975)
<b>1996</b>	50,000 (herring)	New England	88,702	56.4	(Saila et al., 2002)
<b>1999</b>	65,352 (herring)	Northeast	79,420	82.3	NEFMC, 1999c, p. 8)
<b>2002</b>	116,000 (all types)	Gulf of Maine			(Saila et al., 2002)
<b>2006</b>	101,604 (all types)	Maine			(Driscoll et al., 2015)
<b>Post-2007</b>		Gulf of Maine		Up to 80%	(Lehuta et al., 2014)
<b>2010</b>	70,000 – 75,000 (herring)	New England	65,730	106.5 – 114.1	(Grabowski et al., 2010)

1 = (NOAA Office of Science and Technology, 2019)

2 = (Maine DMR, 2018)

This available information for the use of Atlantic herring in the USA lobster fisheries, while scant, presents the bare outlines of a consistent picture. The use of herring in the Maine lobster fishery was minimal as late as the early 1980s, but increased substantially thereafter. Thus, the USA lobster fisheries became an increasingly dominant user of the USA's Atlantic herring landings over this time. At the start of the 1980s, the Maine lobster fishery used approximately 2% of USA Atlantic herring landings (Table 3.1). By the mid-1990s, it was using at least 35%, and by the mid-2000s, it was using over 70% (Table 3.1). At broader geographic scales, the USA lobster fishery progressed from using approximately 50% of USA Atlantic herring landings in the mid-1990s to potentially using up to 80%, or more, in recent years (Table 3.2).

#### *3.2.1.2. Model description*

While the information presented in Tables 3.1 and 3.2 in the previous section suggests a multi-decade trend of increasing herring use in the Maine lobster fishery, it does not provide year-to-year information for this trend. Here, I use a simple model that provides a range of estimates for the annual amounts of herring used as bait in the Maine lobster fishery from 1995 through 2017.

The basis for this model is the annual landings of Atlantic herring in the United States. While some herring may be imported from Canada for use as bait, it is assumed for the purposes of this model that the amount of Atlantic herring used as bait in the Maine lobster fishery in any given year is generally constrained by the total catch of Atlantic herring in the United States for that year. Annual herring landings thus represent a pool from which the Maine lobster fishery withdraws an amount each year. Proceeding from this assumption, the model estimates annual Atlantic herring bait use as a proportion of annual Atlantic herring landings.

For the years 1995-2006, the proportion of annual herring landings used as bait was informed by the published single-year estimates of herring use in the Maine lobster fishery for 1995, 1996, and 2006 (Table 3.1). These three estimates indicate that the

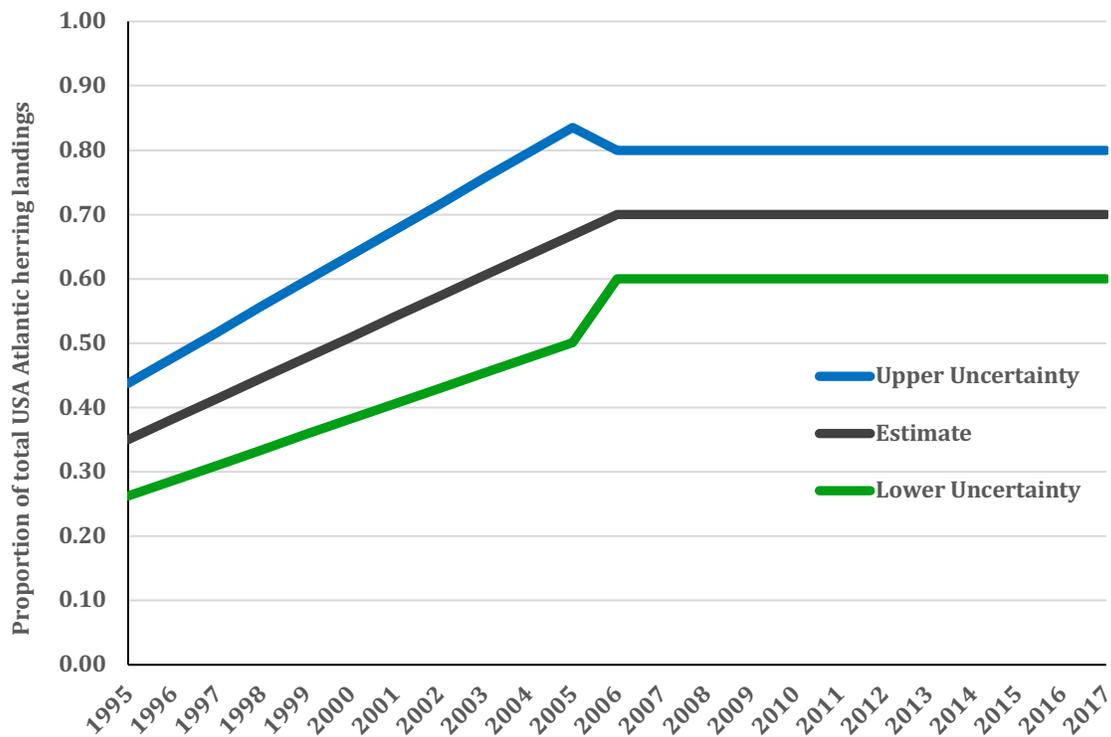
Maine lobster fishery used approximately 35% of total USA Atlantic herring landings in 1995 and 1996 (a value that is likely a conservative estimate, for the reason noted previously), and used slightly over 70% by 2006 (Table 3.1). For the other years between 1995-2006, the model estimates the lobster fishery's use of herring by linearly interpolating between 35% of Atlantic herring landings in 1995 and 70% in 2006 (Figure 3.3).

Due to the absence of specific estimates for any years after 2006, the model maintains an estimate of the 70% of annual USA Atlantic herring landings for the years 2007-2017. This estimate for the 2006-2017 period were derived from the literature. Driscoll and colleagues (2015), using surveys of Maine lobster license holders (n=83), reported that Maine lobster fishers in the 2006 fishery used Atlantic herring at rates that would have equaled approximately 73% of the 2006 USA Atlantic herring landings (Table 3.1). Grabowski and colleagues (2010) estimated that "about 70%" of Atlantic herring were used as bait in New England lobster fisheries (Table 2); the methods by which they arrived at this estimate are not known (Grabowski et al., 2010). Thus, 70% of Atlantic herring landings is considered the "best guess" for the average annual herring use in the Maine lobster fishery since 2006 (Figure 3.3).

However, this may be an underestimate for this period overall. It is notable that the 2006 estimate (in which the Maine lobster fishery was estimated to use approximately 73% of that year's Atlantic herring landings) was also the year of the highest herring landings in recent decades; thus, the proportion of herring destined for use as lobster bait may be higher in years when herring landings were lower. Indeed, Lehuta and colleagues (2015) stated that the Gulf of Maine lobster fishery may have consumed up to 80% of annual herring landings in the years after 2007; methods were not apparent for this estimate (Lehuta et al., 2014).

Thus, 80% of annual USA Atlantic herring landings was used as the upper uncertainty boundary for the years 2006-2017. Of the few sources found in the

literature, none suggests that the Maine lobster fishery used less than 70% of total herring in the years after 2006, and there is no indication that herring has been replaced by other baits at a large scale. For this reason, 60% of total herring landings was assumed to represent an adequately cautious “low” estimate for those years. For the preceding years (1995-2005), upper and lower uncertainty boundaries were arbitrarily set at +/- 25% of the model’s estimate of herring bait biomass (Figure 3.3). See Section 3.2.3, “Uncertainty” below for more information on uncertainty.



**Figure 3.3.** Modeled proportions of total USA Atlantic herring landings used as bait in the Maine lobster fishery, 1995-2017.

### 3.2.2. Estimating nutrient yields

For both herring and lobster, yields of seven nutrients were estimated: protein, the omega-3 fatty acids eicosapentaenoic acid (EPA) and docosahexaenoic acid (DHA), iron, zinc, and vitamins A and B<sub>12</sub>. Nutrient yields associated with lobster landings and herring bait estimates were determined by a two-step process. First, edible biomass yield ratios were applied to convert landings weights to estimates of edible

biomass. Nutrient content rates were then applied to these estimates of edible biomass, resulting in estimates of edible nutrient yields.

Maine lobster landings data were obtained from Maine's Department of Marine Resources (Maine DMR, 2018). Atlantic herring landings data were obtained from the National Marine Fisheries Service (NOAA Fisheries, 2020). The values used for edible biomass yield and nutrient content rates, and their sources, are shown in Appendix B. Additional considerations for lobster are as follows.

#### *3.2.2.1 Estimating gross nutrient yields from lobster landings*

Estimating the nutrient yields of the Maine lobster fishery requires consideration of the fishery's catches of soft-shell vs. hard-shell lobsters, as the proportion of body weight that is edible can vary substantially between these two molt stages. There are no published estimates of the proportion of Maine lobster yields that have been composed of soft-shell vs. hard-shell lobsters over time, but, as mentioned previously, the Maine lobster fishery's primary season occurs when lobsters are in their soft-shell stage (Holland, 2011).

There are a variety of estimates of the edible biomass yield of lobsters in their soft- and hard-shell states. As mentioned previously, soft and hard-shelled lobsters are generally said to have edible biomass yields of 12.5%-18% and 20-28%, respectively: Dow and colleagues estimated that soft- and hard-shell lobsters yielded 12.5% and 25%, respectively (Dow et al., 1975).

Thus, for the purposes of this study, an assumption was made that Maine lobster fishery's landings are composed of 70% soft-shell and 30% hard-shell lobsters, and the edible biomass yield rates for soft- and hard-shell lobsters were assumed to be 15% and 25%, respectively. Combining these two estimates, the edible biomass yield rate for the Maine lobster fishery is estimated to be 18%.

### 3.2.3. Uncertainty

There is no empirical source from which uncertainty can be estimated for the model's estimates of annual herring bait. Thus, in lieu of traditional confidence intervals, lower and upper bounds for the modeled herring bait biomass were established as follows: for the years 1995-2005, the bounds were arbitrarily set at +/- 25% of the model's estimate of herring bait biomass. For 2006-2017, as described in Section 3.2.1.2, the lower and upper bounds were set at 60% and 80% of total USA Atlantic herring landings, respectively.

The resulting lower and upper bait biomass estimates were converted to nutrient yield estimates by applying edible biomass yield ratios and nutrient content estimates, as described in Chapter 2.

At this stage, there were estimates of the nutrient yields associated with the lower and upper bait biomass bounds, i.e., the lower nutrient bound (LNB) and upper nutrient bound (UNB) for each nutrient *V*. The next and final step at this point was to determine the uncertainty in these lower and upper nutrient yield estimates. Ideally, this would involve the combination of standard errors of the mean for both edible biomass yield estimates and for nutrient content estimates, and subsequent calculation of traditional 95% confidence intervals. However, standard errors of the mean were not available for herring or lobster edible biomass yield ratios, or for estimates of vitamin A content. Thus, in lieu of standard errors of the mean, "proxy standard errors of the mean" (PSEM) were developed for estimates of lobster and herring edible biomass yield ratio, and vitamin A content. For edible biomass yield rates, this consisted of applying an uncertainty rates of 20% and 30% to the herring and lobster edible biomass yield rate estimates, respectively, after consideration of the quality and agreement of available edible biomass yield estimates; for vitamin A, the PSEM was the standard error of the mean vitamin A content across 43 taxa from the northwest Atlantic (see Chapter 2 and Appendix A for more detail on these PSEM methods). For the remaining nutrient content estimates, standard errors of the mean (SEM) were available from the literature, or were estimated from other

taxa (see Chapter 2). Edible biomass yield and nutrient content SEM and PSEM values are shown in Appendix D.

Using these SEM and PSEM values, proxy 95% confidence intervals for the LNB or UNB of each nutrient  $V$  were obtained as shown in Equation 1:

$$\text{Proxy 95\% Confidence Interval}_{LNB_V} = 1.96 * LNB_V \left( \sqrt{\left(\frac{\sigma_{E_{ED}}}{E_{ED}}\right)^2 + \left(\frac{\sigma_{E_V}}{E_V}\right)^2} \right) \quad \text{Equation 1}$$

Where  $E_{ED}$  is the edible biomass yield estimate,  $E_V$  is the estimate of the concentration of nutrient  $V$  in the edible biomass, and  $\sigma$  is the SEM or PSEM. (Note that Equation 1 is for the  $LNB_V$ , but can apply to the  $UNB_V$  as well).

The final step was to apply these proxy 95% confidence intervals to each nutrient  $V$ 's LNB and UNB. This was done as shown in Equations 2 and 3, respectively:

$$\text{Uncertainty}_{LNB_V} = LNB_V + \text{Proxy 95\% Confidence Interval}_{LNB_V} \quad \text{Equation 2}$$

$$\text{Uncertainty}_{UNB_V} = UNB_V - \text{Proxy 95\% Confidence Interval}_{UNB_V} \quad \text{Equation 3}$$

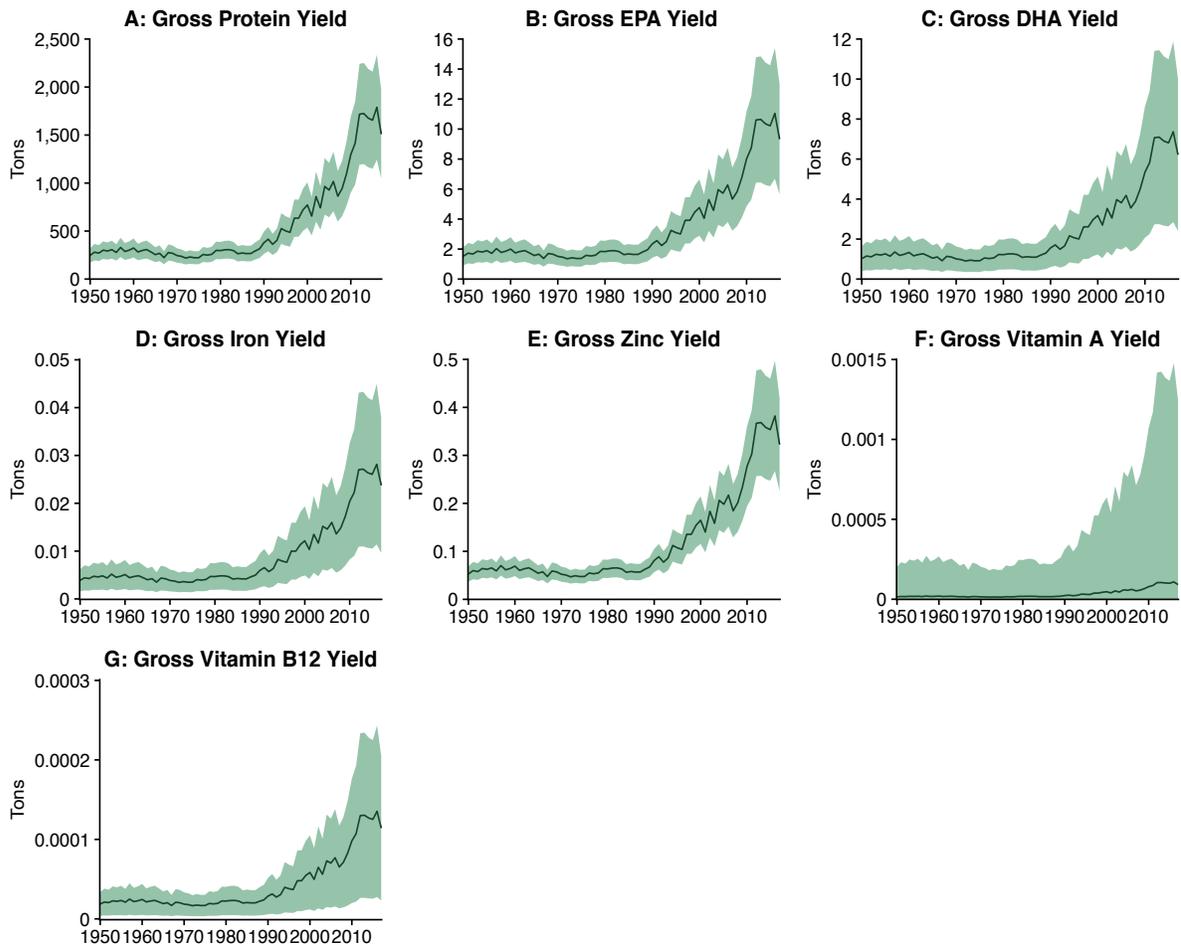
Note that because the net nutrient yields of all nutrients except zinc were negative, the proxy 95% confidence intervals were *added* to the LNB and *subtracted* from the UNB. This is because the “lower” and “upper” refer to the lower and upper boundaries estimated for bait biomass, and thus the resulting nutrient yields, being negative numbers, would be closer and further from zero for lower and upper bait biomass estimates, respectively.

### 3.3. Results

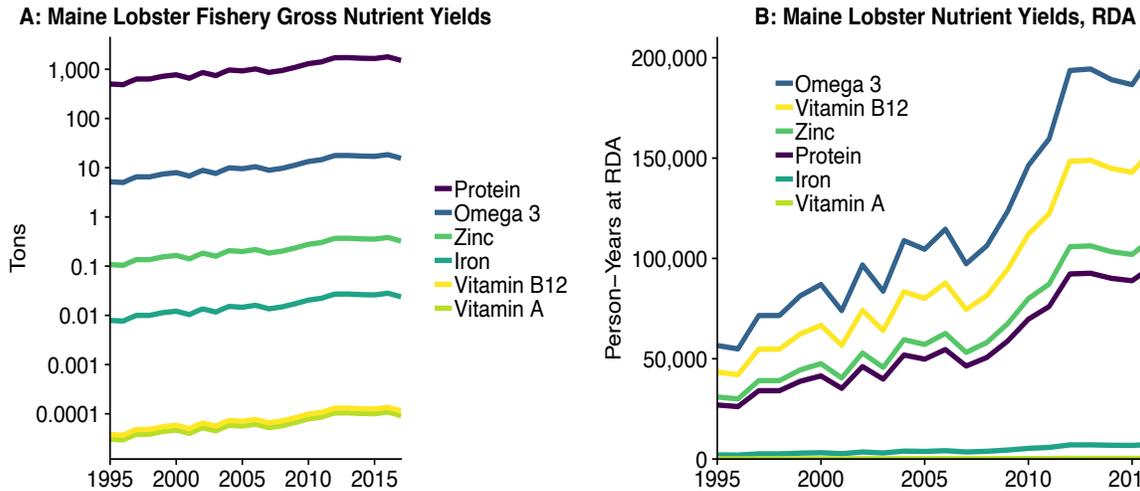
#### 3.3.1. Lobster gross nutrient yields

From 1950 to 2017, gross nutrient yields from the Maine lobster fishery's landings increased by over six times (Figure 3.4), which predictably followed the landings

trend (Figure 3.1). In absolute amounts, the gross nutrient yields at the end of the assessed period spanned eight orders of magnitude, from tens of grams of vitamin B<sub>12</sub> to thousands of tons of protein (Figure 3.5a). When expressed in terms of the number of humans who could, in theory, be supported for one year at the relevant RDA or equivalent amount, the gross yields of the omega-3 fatty acids EPA and DHA (combined, as omega-3 fatty acids) and vitamin B<sub>12</sub> were the most substantial, with recent gross yields providing nutrients sufficient to support 130,000 to 200,000 human-years (Figure 3.5b). At the other end of the spectrum, gross yields of vitamin A, while highly uncertain, were likely the lowest relative to human dietary needs, with recent annual yields theoretically supporting several hundred person-years (Figure 3.5b).



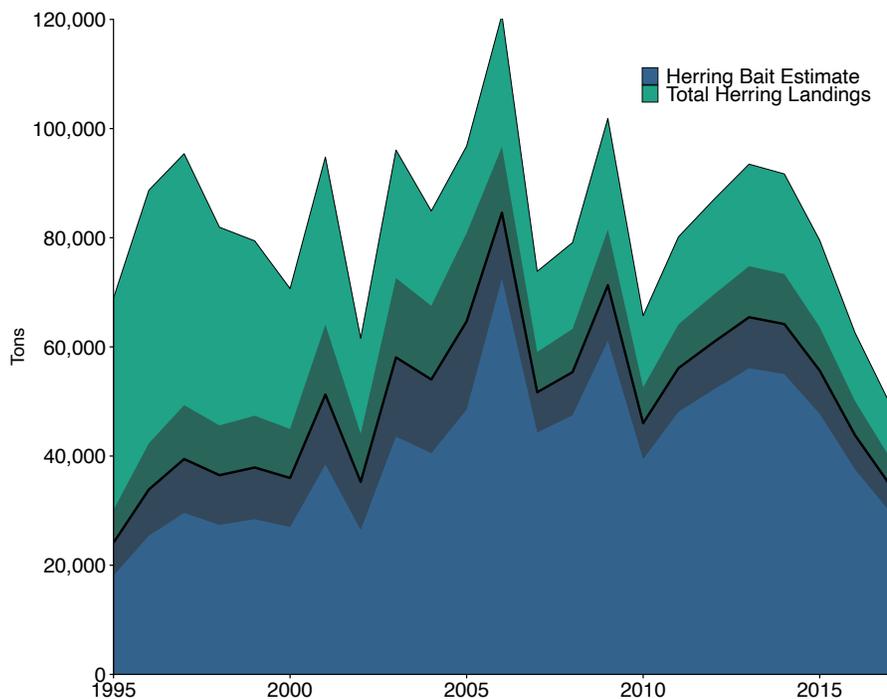
**Figure 3.4.** Gross yields of nutrients from Maine lobster landings, 1950-2014.



**Figure 3.5.** Maine lobster fishery gross nutrient yields, 1950-2014, expressed in absolute amounts on a log scale (Figure 3.5A) and in terms of the relevant RDA or equivalent measure (Figure 3.5B).

### 3.3.2. Modeled Atlantic herring bait estimates

Modeled Atlantic herring bait use in the Maine lobster fishery grew from approximately 35,000 tons/year in 1995-2000 (average low and high bounds of the estimates = 22,513-46,767 t) to nearly 58,500 tons/year for 2001-2014 (48,172-68,813 t) (Figure 3.6). For 2015-2017, declining landings of Atlantic herring resulted in modeled herring bait use declining, to an average of approximately 44,700 t/year (38,319-51,092) (Figure 3.6).



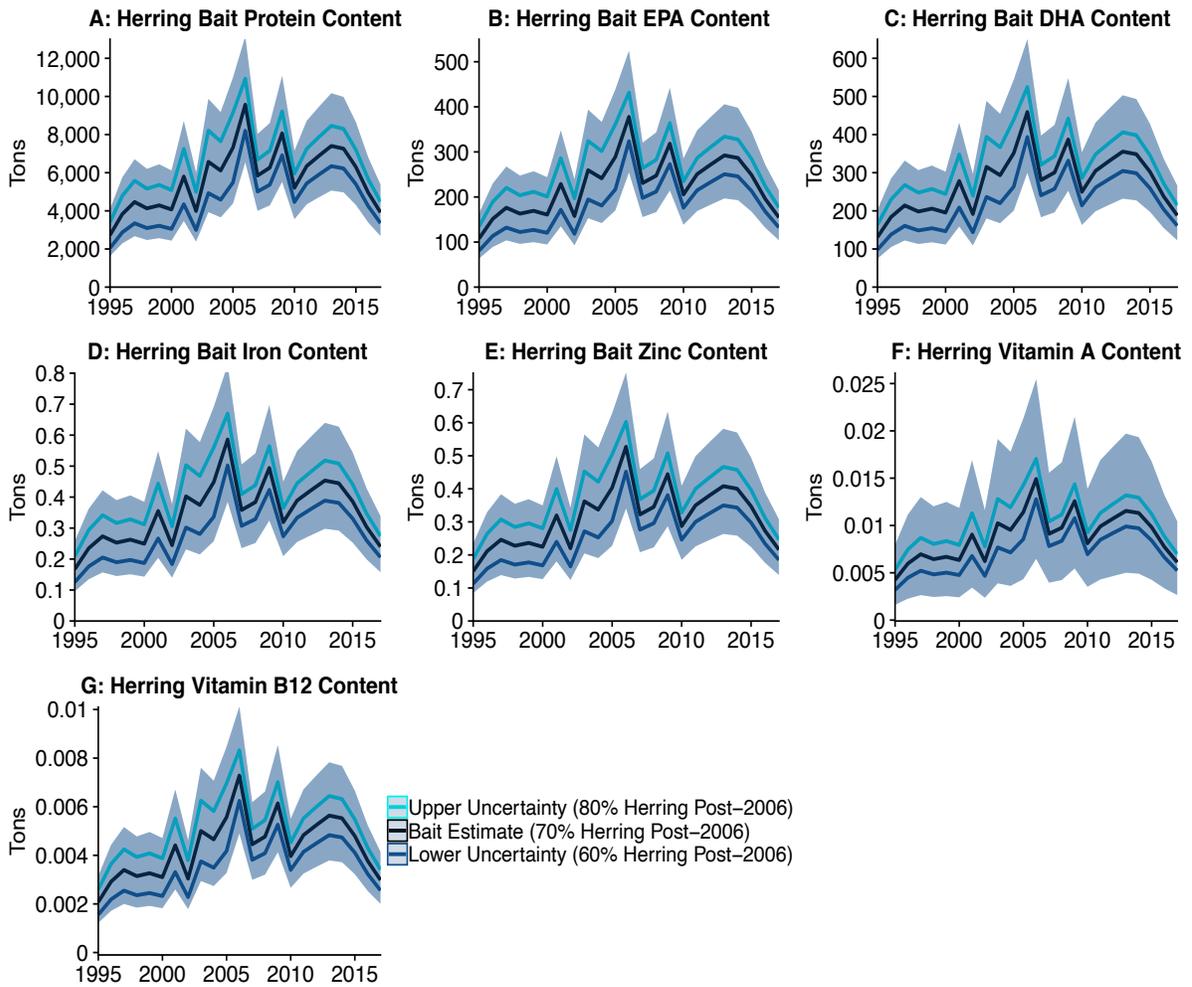
**Figure 3.6.** Modeled estimates of Atlantic herring bait use in the Maine lobster fishery, 1995-2017. The bait estimate is indicated by the solid black line. Shaded areas represent lower and upper uncertainty boundaries.

### 3.3.3. Net nutrient yields of the Maine lobster fishery

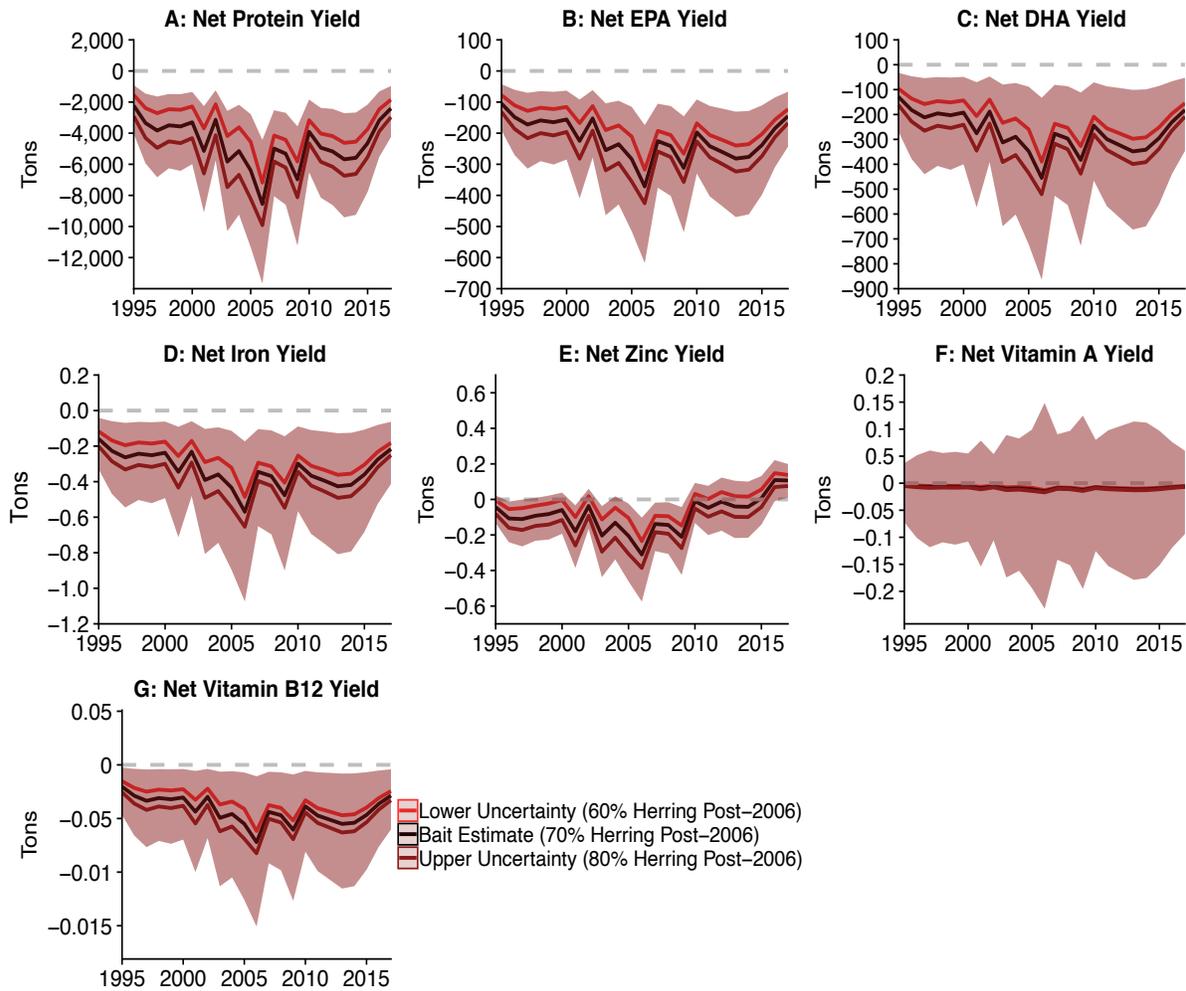
The nutrients embodied in the modeled herring bait are shown in Figure 3.7. When the nutrients embodied in the modeled herring bait inputs (Figure 3.7) are subtracted from the Maine lobster fishery’s gross nutrient yields (Figure 3.4), the results are estimates of net nutrient yields from the lobster fishery (Figure 3.8).

The estimates of net nutrient yields from the lobster fishery are characterized by substantial uncertainty: note that the uncertainty bands for the net yields in Figure 3.8 are considerably broader than the bands for the estimates of nutrients in herring bait (Figure 3.7), for example. This is the result of uncertainty in estimates of herring bait use, estimates of hard shell vs. soft-shell landings in the lobster fishery, estimates of edible biomass yield in hard vs. soft-shell lobster and in herring, and estimates of the yields of the specific nutrients from lobster and herring biomass.

Despite the substantial uncertainty, it is likely that annual net nutrient yields from the lobster fishery were negative – i.e., the lobster fishery was a net consumer of nutrients – for protein, EPA, DHA, iron, and vitamin B<sub>12</sub>, for the assessed period of 1995-2017 (Figure 3.8). For these five nutrients, yields have consistently been net negative at all three herring bait estimate levels, and even the broad uncertainty bands have not crossed into net positive yields (Figure 3.8). In contrast, the lobster fishery has likely been a net producer of zinc in recent years (Figure 3.8E). High uncertainty precludes reliable interpretation of the fishery's net yield of vitamin A (Figure 3.8F).



**Figure 3.7.** Nutrient content of modeled estimates of Atlantic herring used as bait in the Maine lobster fishery, 1995-2017.



**Figure 3.8.** Net nutrient yields from the Maine lobster fishery, 1995-2017. These figures represent the net result of subtracting the estimated nutrient content of herring bait (Figure 3.7) from the estimated nutrient content of lobster landings (Figure 3.4). The dashed line represents net neutral nutrient yield.

### 3.4. Discussion and Conclusion

The results of this analysis suggest that for many years the Maine lobster fishery has been a net consumer, rather than producer, of multiple nutrients. This conclusion can be drawn despite substantial uncertainty regarding herring bait inputs, the proportion of landings composed of soft vs. hard-shelled lobster, edible biomass yield rates, and nutrient yields per unit of edible biomass. Of the assessed nutrients, only zinc, and possibly vitamin A, have been produced in net positive amounts by

the Maine lobster fishery in recent years. The scale of net nutrient consumption by the Maine lobster fishery varies across the assessed nutrients. While there is substantial uncertainty, the results indicate that annual herring bait use in recent years may have embodied approximately 2.5-5 times more edible protein than annual lobster landings, and much more omega-3 fatty acids (approximately 17-50x), iron (10-20x), and vitamin B<sub>12</sub> (26-50x).

In Atlantic herring, this study addresses the Maine lobster fishery's primary bait. However, there are other fish species that are also used on a whole-fish basis (i.e., not as "racks" or other post-processing wastes). Thus, this study likely underestimates the actual scale of whole-fish bait use in the lobster fishery. Driscoll et al. (2015) reported that approximately 5% of the bait used in the 2006 fishery was menhaden (*Brevoortia tyrannus*). Since that time, reductions in allowable catch of Atlantic herring have likely increased the use of menhaden and other bait sources. As of 2019, the Maine Department of Marine Resources had approved over two dozen species for use as lobster and/or crab bait, including some from distant regions (e.g., the US and Canadian west coasts, Alaska, South America, Australia; Maine DMR, 2019). As these and other sources of whole-fish biomass are not included in this study, the real whole-fish bait inputs in the Maine lobster fishery are likely greater than those modeled here, further supporting the conclusion that the Maine lobster fishery has been a net consumer of multiple nutrients.

Challenging as they may be for the industry, current events could, in theory, lead the Maine lobster fishery toward improvements in nutrient yields. Substantial reductions in Atlantic herring catch limits, enacted to protect the declining herring stock, have presented a significant challenge to the lobster fishery in recent years. As a result, many in the industry are considering, or have already moved to embrace, alternatives to herring (Waterman, 2019). Some of those alternatives are simply other fish species in which the whole fish is used for bait, however. The use of such alternative whole-fish baits would obviously not be expected to result in wholesale improvements in the lobster fishery's net nutrient yields.

The use of the oil-rich Atlantic menhaden, in particular, presents an interesting question. Due to their boniness and taste, menhaden are widely considered to be functionally inedible for humans. They are currently fished entirely for reduction to fishmeal and fish oil or for use as bait (ASMFC, 2019), and after Atlantic herring, they were the second-most preferred bait in the Maine lobster fishery as of 2006 (Driscoll et al., 2015). The question, then, is this: how does the use of a functionally inedible species as bait affect the lobster fishery's net nutrient yields? Clearly, it is an input, and likely a nutritious one, due to the species' high oil content. Unlike the Atlantic herring, however, the consequences of using menhaden as bait cannot be compared to a hypothetical in which the menhaden's edible biomass is directly consumed, because as far as human diets are concerned, there is essentially no edible biomass associated with this species. To understand the nutrient yield consequences of menhaden bait use in the lobster fishery, then, the inputs would likely be characterized in terms of the nutrition content of the fishmeal and fish oil that could otherwise be derived from the menhaden inputs, but even this brings challenges, as humans do not directly consume fishmeal.

The environmental consequences of this scale of bait use extend beyond the consumption of nutrients. Atlantic herring are a key prey species for multiple predators in the Gulf of Maine/Georges Bank region, with predator consumption of herring substantially exceeding fisheries landings at the end of the 20<sup>th</sup> century (Overholtz et al., 2008). The lobster fishery is thus in indirect competition, to some degree, with the marine predators of herring – a situation that is particularly notable in light of recent downturns in herring abundance (NEFSC, 2018b). Furthermore, like all industrial processes, the process of fishing, storing, and transporting Atlantic herring for use as bait results in a suite of chemical emissions, including greenhouse gases. While the emissions associated with the herring fisheries are relatively low, compared to other fisheries, the scale of herring bait use in the lobster fishery is such that herring bait is a substantial contributor to the lobster fishery's overall suite of chemical emissions (Driscoll et al., 2015).

To the author's knowledge, this is the first example of a fishery potentially consuming more human-available nutrients than it produces. In this regard, the Maine lobster fishery may run counter to what has, to date, been an unspoken assumption about fisheries: that they are inherently net nutrient producers, because their nutrients are obtained "for free" from marine ecosystems rather than from systems that are obviously and intentionally based on artificial nutrient inputs (e.g., agriculture and much aquaculture production). The case of Maine lobster, however, challenges this assumption: not only is the growth of lobsters potentially subsidized by herring inputs (Grabowski et al., 2010), but now, through the research presented in this chapter, it appears that the very act of catching lobsters requires so much bait as to make the fishery a net consumer of multiple nutrients.

The Maine lobster fishery is clearly among those food production systems that require lower-valued inputs to obtain higher-value products, alongside much animal agriculture and some aquaculture systems. Rather than cropland and crops, in this case, it is a highly nutritious and ecologically critical species that is being exploited, largely to serve as an input to a system that produces a higher-value food product at a likely net nutritional loss. There is also an additional nuance: in this case, the lower-value input has higher concentrations of many nutrients than the higher-value output. The parallels to the larger debate about shifting diets from animal-derived foods to plant-derived foods are therefore not only clear, but emphasized: from a nutritional perspective, the use of herring as bait for lobster is particularly inefficient.

Furthermore, there are potential justice implications, as herring are a much more affordable source of nutrients than are lobsters. In 2014, the average landed value of USA lobster (\$7.57/kg) was more than 20 times greater than the average landed value of USA herring (\$0.32/kg) (Pauly & Zeller, 2016b). As a delicacy, the nutrients embodied in lobster are likely destined for consumption by populations that are not at risk of nutrient deficiencies. Herring, however, may be an affordable option for people facing food insecurity due to economic concerns.

This example may or may not be an outlier among fisheries; additional research on other bait-using fisheries is necessary to understand the degree to which the Maine example is unique or illustrative of a larger trend in bait-using fisheries. To this end, there is a clear need for a significant improvement in the information available for bait use in fisheries. That there was such little information available for bait use in a fishery as well-studied and relatively data-rich as the Maine lobster fishery is somewhat surprising, but it appears indicative of the state of information availability for bait use in fisheries in general. To the author's knowledge, there are no regional or global estimates of bait use in fisheries. Information, where it exists, is often so vague as to be functionally useless, offering little detail on the species composition or amounts of bait used. Considering the attention that is paid to a variety of other fishery-related topics, the dearth of reliable information for bait use is notable and clearly requires remediation.

## **Chapter 4. A theoretical approach to optimizing nutrient yields in fisheries**

### **4.1. Introduction**

The United Nations Food and Agriculture Organization (FAO) estimates that approximately 33.1% of assessed fish stocks were overfished in 2015 (FAO, 2018). To meet international objectives for sustainable development (Sustainable Development Goal target 14.4; DSDG, 2019) and biodiversity conservation (Aichi target 6; CBD, 2019), the exploitation of these stocks must be reduced to sustainable levels. Where management has been effective at rebuilding stocks in recent years, catches have declined (Hilborn et al., 2020). Fisheries provide key nutrients to millions of people, and declines in fisheries catches may put millions at risk of nutrient deficiencies (Golden et al., 2016a). Therefore, an emerging challenge for fisheries science and management is to develop the capacity to make decisions necessary for effective conservation and management, while being informed of potential consequences for the availability of key nutrients for dependent populations.

In some cases, the nutrition requirements of dependent human populations may be met with a fraction of total landings biomass (Hicks et al., 2019). In the analysis of nutrient yields from the Northwest Atlantic Fisheries Organization (NAFO) region (Chapter 2), the first 80% of most nutrients' yields in recent years were obtained from approximately 50% of the landings. This implies that catch reductions, when required for conservation concerns, may be strategically planned to minimize effects on nutrient yields.

In this chapter, I approach this issue from the standpoint of mathematical optimization: how might fisheries' yields of a given nutrient, or a suite of nutrients, be maximized relative to its removals of biomass from marine ecosystems? This question parallels the concept of maximum economic yield (MEY), in which the

difference between a fishery's revenues and costs is maximized (Hoshino et al., 2018). Gordon (1954) defined the "optimum degree of utilization of any particular fishing ground" as the state in which net economic yield ("the difference between total cost...and total receipts...") is maximized. Optimizing fisheries for nutrient yields is essentially the same concept, seeking to maximize the difference between fisheries' nutrient yield "receipts" and their exploited biomass "costs".

In this chapter, I develop an approach for the determination of mathematically optimal scenarios for nutrient yields from fisheries. This approach estimates optimal nutrient yield and the associated optimal landings (i.e., retained catch), identifies the set of taxa that will result in the optimal yield for a given nutrient, and provides options for optimizing for one nutrient, for multiple nutrients of equal preference, and for multiple nutrients of different preference.

Throughout this chapter, the term "optimal" is meant to refer to mathematical optimal, unless otherwise noted. Mathematically optimal scenarios or outcomes, of course, are not necessarily "optimal" from any other perspective, especially given the variety of considerations and objectives that inform fisheries decisions. The term "yield", unless otherwise noted, refers to nutrient yield, and the term "landings" refers to retained catch.

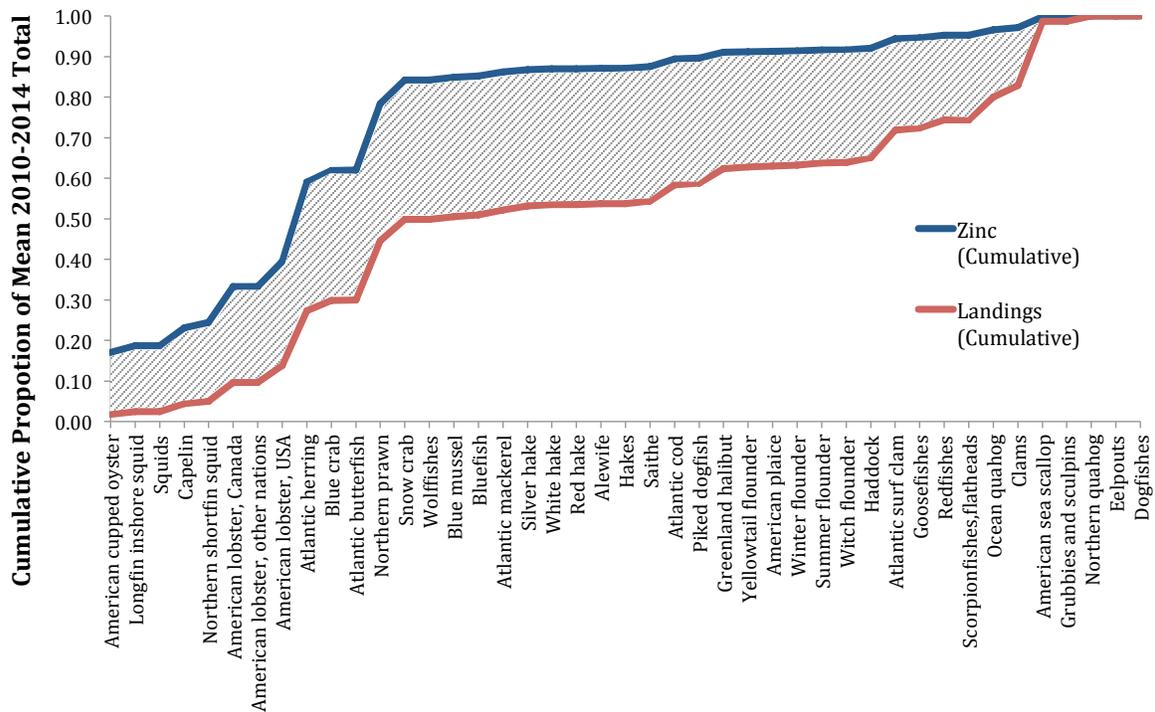
The approach described here is based on the comparison of nutrient yields to landings biomass; however, it can be used to find optimal yields of nutrients relative to any impact that the relevant fisheries have on the environment, from impacts that are proximate in time and space (e.g., seafloor disturbance) to those that are more distant (e.g., contributions to greenhouse gas emissions).

## 4.2. Determining optimal nutrient yields

### 4.2.1. Optimization potential

In the approach described in this chapter, the objective function to be maximized is the “optimization potential”, which is the difference between the cumulative yield of a nutrient, or a group of nutrients, and cumulative landings biomass.

The optimization potential is illustrated in Figure 4.1. In this example, mean annual zinc yields and landings are shown for 43 taxa landed in the Northwest Atlantic Fisheries Organization (NAFO) region during the years 2010-2014 (zinc yield and landings data obtained as described for Chapter 2).



**Figure 4.1.** Optimization potential (shaded area) for mean annual zinc yields and landings of 43 taxa caught in NAFO fisheries, 2010-2014. Taxa are arranged in order of descending ratio of nutrient yield to landings. Yields and landings are cumulative and are expressed in terms of proportion of the respective mean annual NAFO total. For any point  $T$  on the x axis, the optimization potential is the difference between the cumulative zinc yield at  $T$  and the cumulative landings at  $T$ .

Formally, the optimization potential is found as follows. For a set of taxa  $S$ , total landings and total yield of nutrient  $V$  are the sum of all taxon-specific landings ( $l$ ) and yields ( $y_V$ ). Each taxon  $i$  thus contributes a proportion of the total landings ( $PL$ ) and a proportion of the total yield of  $V$  ( $PY_V$ ). The proportions of total landings and total yield of  $V$  obtained from taxon  $i$  are found as shown in Equations 4 and 5, respectively.

$$PL_i = \frac{l_i}{\sum_{n=1}^S l_i} \quad \text{Equation 4}$$

$$PY_{V,i} = \frac{y_{V,i}}{\sum_{n=1}^S y_{V,i}} \quad \text{Equation 5}$$

Figure 4.1 shows taxa organized on the x axis by descending order of their ratio of yield contributions to landings contributions (i.e.,  $PY_{V,i}/PL_i$ ). When taxa are ordered in this way, the optimization potential for any ordered taxon  $T$  is the cumulative difference between taxon-specific incremental changes in yields and landings at  $T$  (Equation 6).

$$\text{Optimization potential}_T = \sum_{i=1}^T (PY_V - PL)_i \quad \text{Equation 6}$$

Using Figure 4.1 as an example, the optimization potential at the point “northern prawn” is the cumulative sum of the incremental changes in yields minus the incremental changes in landings for the 12 taxa up to and including northern prawn. Or, in other words, the optimization potential is simply the blue line minus the red line at the point “northern prawn”.

#### 4.2.2. Relating optimization potential to taxon-specific yields and landings

From the example shown in Figure 4.1, it is apparent that any taxon that contributes more to nutrient yield ( $PY_V$ ) than to landings weight ( $PL$ ) will increase the optimization potential, any taxon that contributes less to yield than to landings weight

will diminish the optimization potential, and any taxon that contributes to both equally will neither increase nor diminish the optimization potential.

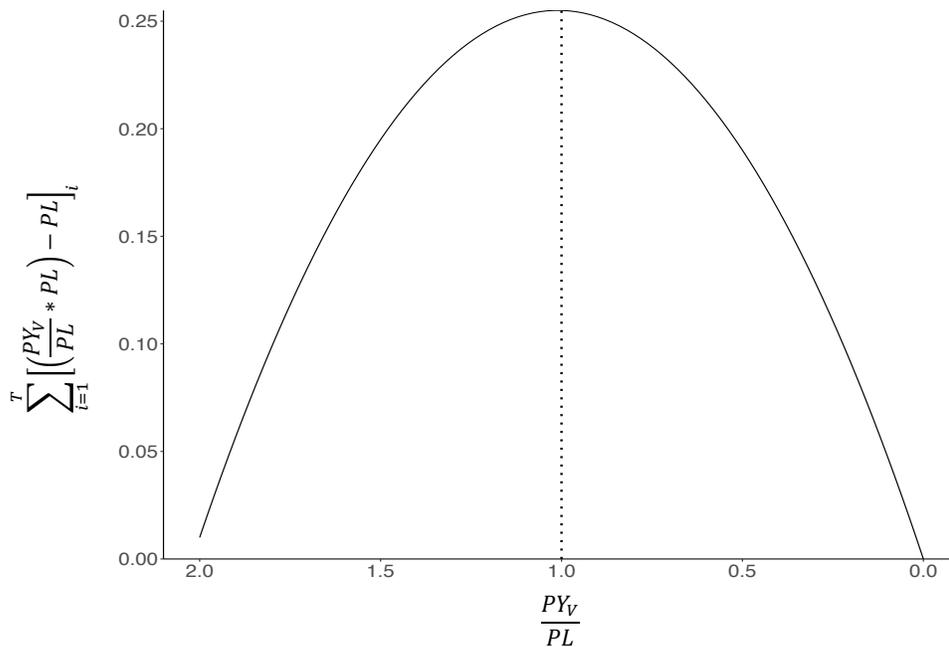
The question at this point is how to establish a taxon's contributions to nutrient yields vs. landings. From Equation 6, it seems that this could be found by calculating  $PY_V - PL$  for each taxon. However, the resulting ranking of taxa would be based on the absolute difference between  $PY_V$  and  $PL$ , and this would favor taxa that are caught in high volumes, so long as they produce the nutrient at a slightly greater rate than they contribute to landings. Taxa that are caught in minor amounts would be diminished, even if they produce the nutrient at a high rate relative to their landings. For example, a taxon that contributed 55% of yields and 50% of landings would have a higher  $PY_V - PL$  value than a taxon that contributed 5% of yields at 1% of landings. However, as the intent of this optimization exercise is to maximize the difference between nutrient yield and landings weight, the latter taxon is clearly the one that should be considered "more optimal" in this analysis.

The ratio  $\frac{PY_V}{PL}$  removes the effect of differences in catch volume, allowing taxa to be ordered by a scale-free indication of their relative contributions to total nutrient yields. Once taxa are put in order of decreasing  $\frac{PY_V}{PL}$  values, the optimization potential can then be found as a function of this ratio, for any ordered taxon  $T$ . Since  $\frac{PY_V}{PL} * PL = PY_V$ , Equation 6 can be modified as:

$$Optimization\ potential_T = \sum_{i=1}^T \left[ \left( \frac{PY_V}{PL} * PL \right) - PL \right]_i \quad Equation\ 7$$

A key implication of Equation 7 is that when  $\frac{PY_V}{PL} = 1.0$ , the change to the optimization potential is 0. This marks the maximum optimization potential. The optimization potential is also at its maximum for the interval immediately preceding

1.0, because the interval for which  $\frac{PY_V}{PL} = 1.0$  adds equally to both cumulative yield and cumulative landings, and thus does not change the difference between the two. Thus, for a set of taxa arranged by decreasing  $\frac{PY_V}{PL}$  values, the optimization potential will be at its maximum value for the last taxon to have a value that exceeds or equals 1.0. Figure 4.2 shows the relationship of the optimization potential to  $\frac{PY_V}{PL}$ , for a scenario in which  $PL=0.01$  and  $\frac{PY_V}{PL}$  continuously declines from 2.0 to 0.



**Figure 4.2.** Optimization potential (y axis) as a function of  $\frac{PY_V}{PL}$ , for a scenario in which  $\frac{PY_V}{PL}$  continuously declines from 2.0 to 0, and  $PL=0.01$ . The dotted line indicates the maximum optimization potential.

#### 4.2.3. Optimality index, $R$

The  $\frac{PY_V}{PL}$  ratio, when calculated for each taxon  $i$ , can thus be used to differentiate taxa based on their relative optimality for a given nutrient  $V$ , and ultimately determine the maximum optimization potential. Here, this ratio is called the optimality index,  $R$  (Equation 8).

$$R_V = \frac{PY_V}{PL} \quad \text{Equation 8}$$

#### 4.2.4. Defining the optimal set of taxa: comparing $R$ to $\alpha$

Once  $R$  values have been determined for all assessed taxa, the next step in the optimization process can occur: the differentiation of the optimal and sub-optimal sets of taxa. This is done by comparing each taxon's  $R$  value to a desired degree of optimality,  $\alpha$ .

As shown in Figure 4.2, the maximum optimization potential is obtained from the cumulative yields and landings of all taxa for which  $R \geq 1.0$ . Thus, an  $\alpha$  value of 1.0 will identify the set of taxa that will yield the maximum possible optimization potential. However, the selection of a value for  $\alpha$  is subjective, allowing users to increase or relax the stringency of the requirement for inclusion in the optimal set of taxa in order to accommodate additional objectives, e.g.  $\alpha > 1$  for increasingly stringent selection of the highest-yielding taxa, or  $\alpha < 1$  to allow for the inclusion of taxa that contribute less to yields than to landings.

For a set of taxa  $S$ , the optimal subset for nutrient  $V$  ( $P^{OPT}_V$ ) consists of the taxa that have  $R_V$  values  $\geq \alpha$  (Equation 9), and the sub-optimal subset ( $P^{SUB}_V$ ) consists of those taxa with  $R_V$  values  $< \alpha$  (Equation 10).

$$P^{OPT}_V \ni \{S_{R_V} | R_V \geq \alpha\} \quad \text{Equation 9}$$

$$P^{SUB}_V \ni \{S_{R_V} | R_V < \alpha\} \quad \text{Equation 10}$$

#### 4.2.5. Optimal landings and optimal nutrient yield

The total optimal landings  $L^{OPT}_V$  are then found as the sum of the landings  $l$  of all taxa  $i$  in set  $P^{OPT}_V$  (Equation 11):

$$L^{OPT}_V = \sum_{n=1}^{P^{OPT}_V} l_i \quad \text{Equation 11}$$

The optimal nutrient yield ( $Y^{OPT}_V$ ) for nutrient  $V$  is then determined by applying taxon-specific edible biomass yield and nutrient content conversion rates to the landings that compose  $L^{OPT}_V$ .

#### 4.2.6. Integral of the optimization potential

Another informative value is the integral of the optimization potential; i.e., the total area between the cumulative yield and cumulative landings curves for the entire set of taxa. This value provides information about the possible flexibility for developing configurations of taxa that are optimal relative to broader social objectives, beyond mathematical optimization of nutrient yields. For two regions that have equal maximum optimization potentials, one may have a substantially larger integral, indicating that it has much broader potential to maintain relatively high nutrients in relation to landings (despite changes in taxa fished or to the level of exploitation of

fished taxa). This is because the lower integral implies less overall difference between cumulative yields and cumulative landings across most taxa, and thus any deviation from the mathematical optimal nutrient-yielding configuration will be more likely to result in a substantial reduction in the difference between cumulative yield and landings. In order to be comparable across scenarios that differ in the number of exploited taxa, the integral must be divided by (S-1). The resulting indicator is called the “optimization interval”.

Formally, for a set of taxa  $S$ , the optimization interval for nutrient  $V$  is the sum of optimization potentials for all individual points  $T$  on the x axis (with  $S=\sum T$ ), divided by  $S-1$  (Equation 12):

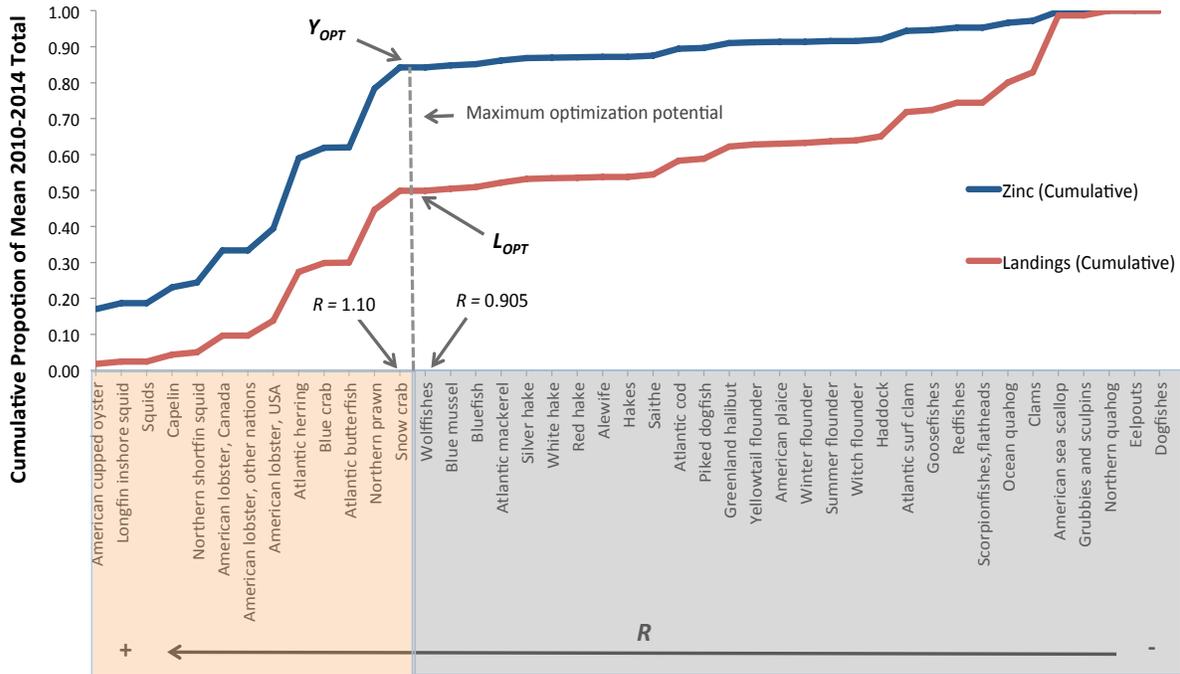
$$\text{Optimization Interval}_V = \frac{\sum_{i=1}^S (\text{Optimization Potential}_V)_T}{S-1} \quad \text{Equation 12}$$

### 4.3. Illustration of optimal landings and nutrient yields

Figure 4.3 illustrates the preceding concepts, again using the example of mean annual zinc yields and landings from NAFO landings during the years 2010-2014.

In this example, as in Figure 4.1, taxa are arranged on the x axis in order of decreasing mean  $R$  value for zinc during these years. The selected  $\alpha$  value is 1.0. The resulting  $P^{OPT}$  and  $P^{SUB}$  sets of taxa are shaded in orange and gray, respectively. As previously mentioned, an  $\alpha$  value of 1.0 identifies the taxa that result in the maximum optimization potential; optimal nutrient yield ( $Y^{OPT}$ ) and the associated optimal landings ( $L^{OPT}$ ) are found as the cumulative yield and cumulative landings, respectively, associated with the last taxon in the optimal set (snow crab in Figure 4.3). The difference between  $Y^{OPT}$  and  $L^{OPT}$  is the optimization potential, which in this example reaches its maximum value following landings of snow crab (0.343671) but prior to landings of wolffish (0.34366). In this example, the mean annual  $Y^{OPT}$  for zinc in the years 2010-2014 is approximately 0.843 of mean annual total zinc yields, while the associated  $L^{OPT}$  is approximately 0.499 of mean annual

landings. The optimization interval is 0.247; as this is a novel indicator, it requires contextualization (e.g., comparison to other scenarios) in order to be informative, and is presented here as an illustration.



**Figure 4.3.** Optimal yield plot for zinc yields in NAFO fisheries, 2010-2014. Taxa are arranged on x axis by  $R$  value, in descending order. The orange-shaded taxa are the optimal set ( $R \geq \alpha$ , where  $\alpha = 1.0$ ). The gray-shaded taxa are the sub-optimal set ( $R < \alpha$ ). The mathematical optimum yield occurs where the optimization potential reaches its maximum, and can also be found as the cumulative yield for the last taxon with an  $R$  value  $\geq 1.0$  (snow crab, in this case). The optimization interval is the area between the blue and orange lines, divided by  $S-1$ .

#### 4.4. Aggregate optimal yield across multiple nutrients

To this point, the analyses have been based on a single nutrient. It may also be useful to understand the optimal and sub-optimal sets of taxa when aggregated across multiple nutrients, and to estimate nutrient yields in such scenarios.

#### 4.4.1. Non-weighted aggregate index of optimality

The  $R_V$  value derived from Equation 7 is the basis for the aggregating exercise. In the absence of any preference for one nutrient over another in the aggregation, the aggregate  $R$  for a taxon, aggregated across all assessed nutrients  $Z$ , ( $AR_Z$ ), is simply the arithmetic mean of the taxon's  $R_V$  values across  $Z$  (Equation 13). This approach allows maximization of the mean optimization potential across the assessed nutrients.

$$AR_Z = \frac{\sum_{i=1}^Z R_V}{Z} \quad \text{Equation 13}$$

#### 4.4.2. Weighted aggregate index of optimality

In many cases, certain nutrients may be prioritized over others: for example, Hicks and colleagues (2019) identified multiple nutrient deficiencies that threaten Namibia's population, and found that approximately 9% of the fish caught in Namibia's exclusive economic zone would be sufficient to supply enough of one of these nutrients (iron) for the nation's coastal population. Thus, iron yields might be prioritized, among multiple other nutrients, in any effort to estimate an aggregate optimal nutrient-yielding set of taxa for that nation.

In this case, Equation 13 can be modified with the following weighting procedure. For a given set of  $Z$  nutrients, the total weighting available is 1.0. For each nutrient  $V$ , a subset of the total available weighting is subjectively assigned ( $W_V$ ). As the total weighting available is 1.0, an equal weighting of all nutrients is  $1/Z$ ; any weighting greater than  $1/Z$  represents an increased preference for that nutrient, and a weighting  $< 1/Z$  represents a diminished preference. Again, the sum of all  $W_V$  values in the set must equal 1.0, regardless of how many nutrients are in the set.

For each taxon  $i$ , the weighted aggregate  $R$  across all nutrients  $Z$  ( $WAR_Z$ ) is the sum of the taxon's incremental contribution to each nutrient's yields, weighted with each nutrient's weighting score, divided by the taxon's incremental contribution to total

landings biomass, as shown in Equation 14. This approach will maximize the weighted mean optimization potential across the assessed nutrients.

$$WAR_Z = \frac{\sum_{n=1}^Z [(PY_{V,i}) * W_V]}{(PL_i)} \quad \text{Equation 14}$$

#### 4.4.3. Aggregate optimal sets

The optimal set of taxa, aggregated across all analyzed nutrients  $Z$  ( $AP^{OPT}_Z$ ), are the subset of taxa  $S$  that have  $AR$  or  $WAR$  values (i.e., un-weighted or weighted aggregate  $R$  values) equal to or greater than  $\alpha$  (Equation 15). Those taxa with  $AR$  or  $WAR$  values less than  $\alpha$  represent the aggregated sub-optimal set of fisheries ( $AP^{SUB}_{Zj}$ , Equation 16).

$$AP^{OPT}_Z \ni \{S_{AR_Z} | AR_Z \geq \alpha\} \quad \text{Equation 15}$$

$$AP^{SUB}_Z \ni \{S_{AR_Z} | AR_Z < \alpha\} \quad \text{Equation 16}$$

## 4.5 Discussion

This chapter presents an approach by which yields of specific nutrients may be optimized, in a mathematical sense, relative to landings biomass. For each assessed nutrient, this approach produces an index of relative optimality ( $R$ ) for each taxon, identifies the set of taxa that are optimal relative to a desired degree of optimality, generates estimates of both the optimal nutrient yield and optimal landings weight for that desired degree of optimality, and provides an optimality interval to aid in the assessment of the potential for flexibility in decision-making. For a degree of optimality of 1.0, the mathematical optimal set of taxa will produce the maximum yield of the nutrient relative to landings biomass – i.e., the maximum optimization potential. Aggregate optimal scenarios across multiple nutrients may be

modeled with all assessed nutrients weighted equally, or nutrients of particular concern receiving greater weighting; aggregate optimization results in scenarios that maximize the weighted or unweighted mean optimization potential across all assessed nutrients. The methods developed in this chapter can be used retroactively, to assess optimal scenarios in years past, or to assess optimal scenarios for projected landings levels in future years.

While the methods in this chapter were written to optimize yields relative to landings, these same methods can be used to optimize fisheries' nutrient yields relative to other environmental impacts of fisheries as well. For example, there is a small but growing body of research on the greenhouse gas emissions of fisheries (e.g., Driscoll & Tyedmers, 2010, Tyedmers et al., 2005, Tyedmers & Parker 2012). A simple adjustment of this chapter's optimization methods – essentially, inserting greenhouse gas emissions estimates in place of landings weights – would result in estimates of optimal nutrient yields relative to emissions. Similar efforts could conceivably be undertaken to optimize nutrient yields relative to other impacts of fisheries, including the area of seafloor swept by mobile fishing gear or the catch of overfished and/or at-risk taxa.

It is worth reiterating that the term “optimal”, as used in this chapter and throughout this dissertation, is narrow: it refers only to what is mathematically optimal if the objective is to maximize nutrient yield relative to landings. The outcomes generated by the methods in this chapter should not be interpreted or presented as being “optimal” in a broader sense, e.g., as indicating what is socially or ecologically optimal, but rather understood as representing one of many potential perspectives on fisheries outcomes and performance.

Still, the insights gained from determining the characteristics of mathematically optimal fisheries scenarios may serve a number of purposes. The optimization process does not even have to be completed to be useful: the information that is generated at each step can inform our understanding of a region's fisheries. The

calculation of  $R$  provides unique, context-specific insight into the relative importance of a species' contributions to a region's yields of a nutrient – insight that cannot be gained from catch weights or from intrinsic nutrient content information alone. The subsequent ordering of taxa by descending  $R$  value provides a simple means of understanding and visualizing the relative importance of each taxon to yields of the nutrient in question. Furthermore, the ordering of taxa by descending  $R$  also enables us to identify the suite of species that will result in the mathematical maximum optimization potential (all taxa with  $R$  values  $\geq 1.0$ ). Finally, if we plot cumulative nutrient yield and landings for these taxa arranged by descending  $R$ , we can see the maximum optimization potential, optimal nutrient yield, and optimal landings as those associated with the last taxon for which  $R \geq 1.0$ . These values (maximum optimization potential, optimal nutrient yield, and landing associated with optimal nutrient yield) can serve as formal or informal reference points for assessing the consequences of different fisheries options.

In this way, even if a region's fisheries are not explicitly managed to result in optimal nutrient yields, decisions at various steps in the management process may be informed by nutrient-related information that is specific to the context – the species, their catch levels relative to other species, and the nutrients of concern. The formal approach to determining optimal nutrient-yielding scenarios described in this chapter therefore does not require similarly formal incorporation of its results in order to be useful; it may incrementally and informally improve the consideration of nutrient yields at multiple points in the fisheries decision-making process. While the optimization approach is necessarily binary in its delineation of taxa as optimal or sub-optimal, its usefulness does not require that the final exploitation decisions are similarly binary. As such, the approach described in this chapter may be of particular use in situations where conservation concerns require reducing landings in multi-species fisheries that are known to provide key nutrients to dependent human populations.

## **Chapter 5. Optimization analysis of fisheries yields in the Northwest Atlantic, 1950-2014**

### **5.1. Introduction**

International objectives for ending hunger and malnutrition (United Nations Sustainable Development Goal 2; DSDG, 2019) and ceasing overfishing (Aichi Biodiversity Target 6; CBD, 2019) (United Nations Sustainable Development Goal 14; DSDG, 2019) face a fundamental challenge: while seafood from fisheries makes critical contributions to the food security and dietary nutrition of at-risk populations, humanity's exploitation of many marine ecosystems and species must be reduced to allow for the recovery necessary for the long-term sustainability of fisheries (Chan et al., 2019).

Fisheries have long been an overlooked component of the larger food system, but a growing body of literature is establishing fisheries as a key contributor to food security and human nutrition (Thilsted et al., 2016). In particular, recent research has highlighted fisheries' role as a source of micronutrients for people whose diets place them at risk of micronutrient deficiencies (Golden et al., 2016a; Hicks et al., 2019). However, while health authorities may suggest increased fish consumption for health purposes, the adoption of such advice may lead to unintended consequences for fish stocks and marine ecosystems (NRC, 2015).

Multiple lines of evidence converge to indicate that limits to global fisheries production have been reached (Worm & Branch, 2012). A recent reconstruction of total fisheries catches suggests that catches have been declining by an average of 1.22 million tons per year since 1996 (Pauly & Zeller, 2016a). This has occurred despite the expansion of fishing effort across most of the world's oceans (Watson et al., 2013) and into deeper waters (Morato et al., 2006). Landings from the new stocks that have been discovered and exploited in the course of this expansion have not been sufficient to offset declining landings from stocks that have been fished for

longer periods of time (Swartz et al., 2010). High trophic level species have been particularly affected by fisheries exploitation. In the North Atlantic, for example, high trophic level fish biomass declined by an estimated 90% during the 20<sup>th</sup> century (Christensen et al., 2003). Globally, substantial abundance declines have been noted for many large demersal fish species (Worm et al., 2009). In some regions, enhanced management capacity has reduced overexploitation and allowed stocks to rebuild in recent years, but stocks in regions with less-robust management remain at high risk of overexploitation (Hilborn et al., 2020).

The challenge is evident: global catches are not likely to increase in the near future, due to declining stock abundance, mandated catch reductions put in place to rebuild over-exploited stocks (Worm & Branch, 2012), and adoption of ecosystem-based management objectives that may limit the catches of specific ecosystem components (Smith et al., 2011) or ecosystem-wide catches (Patrick & Link, 2015). Where stocks have been rebuilt through effective management, catches have been reduced (Hilborn et al., 2020). Thus, enhancing fisheries management in regions where it is currently lax would be expected to result in reduced catches. This may present a challenge to human health, as declines in fisheries catches may result in enhanced risk of nutrient deficiencies in at-risk populations (Golden et al., 2016a).

One potential policy option that has been suggested to address this challenge is to strategically reduce catches, in a manner that seeks to maintain yields of nutrients that are of particularly high concern for dependent human populations (Hicks et al., 2019). This may be possible because nutrient yields are not evenly distributed across taxa; rather, there is substantial heterogeneity in species-specific concentrations of different nutrients. From a policy standpoint, this heterogeneity provides an opportunity to address at least some of the tension between objectives for conservation and food security: by analyzing species' nutrient yield profiles and incorporating this information into decision-making processes, it may be possible to mitigate nutrient yield declines that result from catch reductions.

To this end, In Chapter 4, I developed a theoretical approach to determining optimal scenarios for nutrient yields from fisheries landings. Here, I apply this approach to fisheries landings in the NAFO region for the 1950-2014 period. This analysis is meant to be illustrative: by retrospectively examining the degree to which nutrient yields in those years may have been maximized relative to landings biomass, this exercise will illustrate a) the application of the concepts presented in Chapter 4, and b) the potential for fisheries landings to be optimized for yields of specific nutrients. It should be emphasized, however, that the optimization approach described in Chapter 4 is not limited to retrospective analyses, and can be applied to estimates of future catch levels.

There are three components to this chapter:

1. An evaluation of broad trends over the 1950-2014 period,
2. A detailed analysis of optimal scenarios for the most recent years in the data (2010-2014), and
3. A comparison of possible approaches to optimizing landings for multiple nutrients, simultaneously.

## **5.2. Methods**

The data for 1950 - 2014 NAFO landings, along with all edible biomass yield rates and nutrient content rates, were obtained as described for Chapter 2. The variables and equations used in this chapter are introduced and explained in depth in Chapter 4; here, they are introduced and briefly described in Tables 5.1. and 5.2.

Throughout this chapter, the term “yields” refers to nutrient yields, while “landings” refers to retained catch biomass. To facilitate comparisons across nutrients and across time, yields and landings are expressed as proportions of that year’s total NAFO yields and total landings, respectively. For example, a yield of 0.78 is 78% of that year’s total NAFO yields of that particular nutrient. (Since all analyses in this chapter are retrospective, yields and landings can be expressed as proportions of the amounts that were actually achieved in the assessed year; for analysis of future

scenarios, yields and landings could similarly be expressed as proportions of total predicted amounts that would occur in the absence of optimization efforts).

Also, throughout this chapter, the term “optimal” refers to the levels of yield and landings obtained from a set of taxa that has been defined by a specified degree of optimality,  $\alpha$  (Table 5.1). For all analyses in this chapter,  $\alpha = 1.0$ . Thus, the optimal set of taxa comprises those taxa that contribute more to yields than to landings, or equally to yields and landings, while the “sub-optimal” set consists of taxa that contribute less to yields than to landings.

**Table 5.1.** Variables used in this study.

Variable	Description
$\alpha$	<i>Degree of optimality</i> : the threshold value for differentiating taxa into optimal and sub-optimal sets for a given nutrient, such that those taxa with $R \geq \alpha$ are part of the optimal set. An $\alpha$ value of 1.0 will result in the set of taxa that yield the mathematical maximum optimization potential
$PL_i$	Proportion of total landings that are obtained from a taxon $i$
$PY_{V,i}$	Proportion of total yields of nutrient $V$ that are obtained from a taxon $i$
$AR$	<i>Aggregate index of optimality</i> : for a given taxon, $AR$ is the mean of all nutrient-specific indices of optimality ( $R$ ) across all assessed nutrients
$i$	A single taxon
$j$	A single year
$I$	Landings associated with one taxon
$L$	Landings associated with a set of taxa
$L_{OPT}$	Landings at $Y_{OPT}$ (see below)
$P$	A subset of a larger set of taxa
$R$	<i>Index of optimality</i> : the ratio of a taxon's incremental nutrient yield contribution to that taxon's incremental landings contribution
$S$	The set of all available taxa
$T$	When taxa are arranged on an x axis in descending order of their $R$ values, point $T$ is any point on the x axis
$V$	The nutrient being assessed
$W$	Subjective weighting applied to multiple nutrients in a weighted aggregate optimization scenario; for a set of nutrients, the sum of all nutrient-specific $W$ values must equal 1.0
$WAR$	<i>Weighted aggregate index of optimality</i> : for a given taxon, $WAR$ is the weighted mean of all nutrient-specific indices of optimality ( $R$ ) across all assessed nutrients.
$y$	The yield of a nutrient obtained from one taxon
$Y$	The yield of a given nutrient or set of nutrients
$Y_{OPT}$	<i>Optimal yield</i> : the yield associated with the set of optimal taxa, defined by a specified degree of optimality, $\alpha$ .
$Z$	A set of assessed nutrients

**Table 5.2.** Equations used in this study.

Equation Number	Equation	Description
5.1	$PL_{i,j} = \frac{l_{i,j}}{\sum_{n=1}^S l_{i,j}}$	Proportion of total landings in year $j$ that are obtained from landings $l$ of taxon $i$
5.2	$PY_{V,i,j} = \frac{y_{V,i,j}}{\sum_{n=1}^S y_{V,i,j}}$	Proportion of total yields of nutrient $V$ in year $j$ that are obtained from yields $y$ from taxon $i$
5.3	$R_{V,i,j} = \frac{(PY_{V,i,j})}{(PL_{i,j})}$	Index of optimality: ratio of yield proportion to landings proportion associated with taxon $i$ in year $j$
5.4	$P^{OPT}_{V,j} \ni \{s_{R_{V,i,j}}   R_{V,i,j} \geq \alpha\}$	Subset of taxa with $R_V$ values that equal or exceed specified degree of optimality $\alpha$
5.5	$P^{SUB}_V \ni \{s_{R_{V,i,j}}   R_{V,i,j} < \alpha\}$	Subset of taxa with $R_V$ values that are less than specified degree of optimality $\alpha$
5.6	$Optimization\ potential_T = \sum_{i=1}^T \left[ \left( \frac{PY_V}{PL} * PL \right) - PL \right]_i$	Optimization potential: when taxa are arranged in descending order of $R_{V,i,j}$ , the optimization potential is the cumulative difference between taxon-specific yield proportions and landings proportions at any ordered taxon $T$
5.7	$AR_{Z,i,j} = \frac{\sum_{i=1}^Z R_{V,i,j}}{Z}$	For a taxon $i$ , the aggregated mean $R_{V,i,j}$ value across all assessed nutrients $Z$
5.8	$WAR_{Z,i,j} = \frac{\sum_{n=1}^Z [(PY_{V,i,j}) * W_{V,j}]}{(PL_{i,j})}$	For a taxon $i$ , the weighted aggregated $R_{V,i,j}$ across all assessed nutrients $Z$
5.9	$1.0 = \sum_{i=1}^Z W_{V,j}$	For a set of nutrients $Z$ being analyzed by weighted aggregate optimization, the sum of all nutrient-specific weightings must be 1.0

### 5.2.1. Optimal yields and landings, 1950-2014

The following steps were used to determine optimal yield and landings for each nutrient and year during the 1950-2014 period.

1. First, for each taxon  $i$  and nutrient  $V$  in year  $j$ , the  $PL_{i,j}$  and  $PY_{V,i,j}$  were found (Equations 5.1 and 5.2, respectively).
2. Then, for each taxon,  $R_{V,i,j}$  was derived from  $PL_{i,j}$  and  $PY_{V,i,j}$  (Equation 5.3).
3. All taxa were then arranged in order of decreasing  $R_{V,i,j}$ .
4. The optimal and sub-optimal sets of taxa were then found as shown in Equations 5.4 and 5.5; as the objective in this study is to find the maximum optimization potential, the  $\alpha$  value was set at 1.0.
5. The maximum optimization potential was then found as the optimization potential associated with the last taxon for which  $R_{V,i,j} \geq 1.0$  (Equation 5.6).

6. The yield and landings associated with this optimization potential were then identified as the optimal yield ( $Y_{OPT}$ ) and optimal landings ( $L_{OPT}$ ), respectively, for nutrient  $V$  in year  $j$  (see Table 5.1).

### 5.2.2. Optimal yields, landings, and sets of taxa, 2010-2014

To determine optimal yields, landings, and sets of taxa for the 2010-2014 period, the steps described in Section 5.2.1 were applied to mean annual 2010-2014 landings and nutrient yields data.

### 5.2.3. Aggregate approaches, 2010-2014

This section explored several approaches to developing sets of taxa to meet objectives for multiple nutrients.

#### 5.2.3.1. *Aggregate approaches, all nutrients*

Three different approaches were developed for building aggregate sets of taxa for all assessed nutrients (i.e.,  $Z = 6$ ), without preference for any one nutrient.

#### *AR: Aggregate optimization*

In this approach, Equations 5.1, 5.2, 5.3, and 5.7 were followed to arrive at  $AR_{Z,i,j}$  values for each taxon  $i$ . Equations 5.4 and 5.5 were then followed, with  $AR_{Z,i,j}$  in place of  $R_{V,i,j}$ , to determine optimal and sub-optimal sets relative to an  $\alpha$  value of 1.0. The maximum optimization potential was thus found at the last taxon with an  $AR_{Z,i,j}$  value  $\geq 1.0$ .

#### *Top 10: Top 10 taxa for each nutrient, 2010-2014.*

In this approach, the aggregate set of taxa is composed of all taxa that had at least one 2010-2014  $R_{V,i,j}$  value that placed within the top 10 for any nutrient. As many taxa were in the top 10 for multiple nutrients, this method resulted in the selection of 26 taxa.

### *3 Nutrients: Taxa with 2010-2014 $R_{V,ij}$ values $\geq 1.0$ for $\geq 3$ nutrients*

For this approach, taxa were selected for inclusion if their 2010-2014  $R_{V,ij}$  values put them in the optimal set for  $\geq 3$  nutrients.

#### *5.2.3.2. Aggregate approaches, subset of nutrients*

Two different approaches were developed to evaluate potential approaches for building aggregate sets of taxa for two nutrients (i.e.,  $Z = 2$ ).

##### *WAR: Weighted aggregate optimization, equal weighting between iron and zinc*

In this approach, Equations 5.1, 5.2, 5.8, and 5.9 were followed, with equal weightings ( $W_{Vj} = 0.5$ ) for both iron and zinc. Equations 5.4 and 5.5 were then followed, with  $WAR_{Z,ij}$  in place of  $R_{V,ij}$ , to determine optimal and sub-optimal sets relative to an  $\alpha$  value of 1.0.

##### *Iron + Zinc: Combined optimal sets for iron and zinc*

This approach combined the nutrient-specific optimal sets for both iron (Figure 5.2.C and Appendix E) and zinc (Figure 5.2.D and Appendix E) into one aggregate set of taxa.

## **5.3. Results**

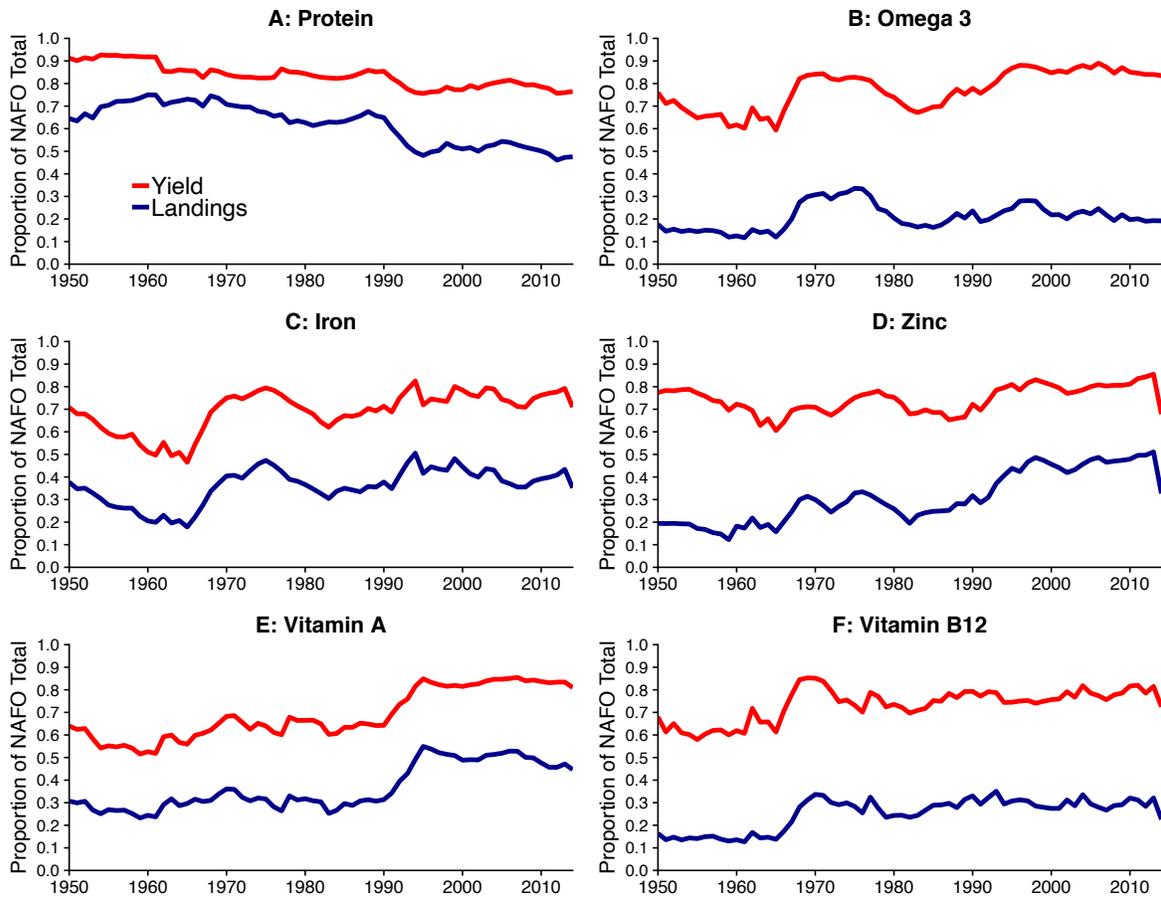
### **5.3.1. Optimal yields and landings, 1950-2014**

Annual optimal yields ( $Y_{OPT}$ ) and associated optimal landings ( $L_{OPT}$ ) for the period 1950-2014 are shown in Table 5.3 and Figure 5.1. Four of the six assessed nutrients had 1950-2014 mean optimal yield values that exceeded 0.73; only vitamin A (0.69) and iron (0.69) fell below this threshold. The overall means of the nutrient-specific mean annual optimal yield and optimal landings were 0.74 (S.E.M.  $\pm$  0.02) and 0.35 ( $\pm$  0.06), respectively. This means that when landings were optimized for one nutrient at a time, approximately 74% of each nutrient could be obtained, on average, from only 35% of the landings.

The mean optimization potential across all nutrients and years was therefore 0.39 ( $\pm 0.05$ ). Protein demonstrated the highest mean optimal yield (0.83); however, relatively high mean optimal landings (0.62) rendered this nutrient's optimization potential the lowest of all assessed nutrients, at 0.21. The nutrients that showed the greatest optimization potentials ( $>0.4$ ) were the omega-3 fatty acids, vitamin B<sub>12</sub>, and zinc, meaning that the majority of these nutrients were provided from taxa that represented only small portions of NAFO total landings.

**Table 5.3.** Yields and landings provided by the optimal set for each nutrient ( $\alpha = 1.0$ ), and associated optimization potentials (the difference between optimal yields and optimal landings) for NAFO fisheries, 1950-2014. Note that the sets of taxa that result in optimal yields and landings are unique to each nutrient.

	<b>Optimal yields</b> (obtained from nutrient-specific optimal set)		<b>Optimal landings</b> (nutrient specific optimal set)		<b>Optimization potential</b>	
	<i>Mean</i>	<i>S.D.</i>	<i>Mean</i>	<i>S.D.</i>	<i>Mean</i>	<i>S.D.</i>
<b>Protein</b>	0.83	0.05	0.61	0.09	0.21	0.06
<b>Omega-3</b>	0.77	0.09	0.21	0.06	0.56	0.06
<b>Iron</b>	0.69	0.09	0.36	0.08	0.33	0.02
<b>Zinc</b>	0.74	0.06	0.31	0.12	0.43	0.08
<b>Vitamin A</b>	0.69	0.11	0.37	0.10	0.32	0.03
<b>Vitamin B<sub>12</sub></b>	0.73	0.07	0.25	0.07	0.48	0.03



**Figure 5.1.** Estimated optimal yields and landings provided by the optimal set for each nutrient ( $\alpha = 1.0$ ), NAFO, 1950-2014. Note that the sets of taxa that result in optimal yields and landings are unique to each nutrient.

No taxon had  $R$  values that exceeded 1.0 for all nutrients across all years during the 1950-2014 period. However, for five of the six nutrients, Atlantic herring (*Clupea harengus*) had  $R$  values  $\geq 1.0$  for all years in the study, falling below 1.0 for only zinc. Atlantic herring was thus a component of the optimal sets of taxa for five of the six nutrients throughout the entire study period.

The optimal and sub-optimal sets of taxa exhibited varying degrees of consistency in their species composition over time. For protein, of the 19 taxa that had  $R$  values  $\geq 1.0$  in the first year of the study (1950), 16 remained above 1.0 for the entire duration of the assessed period, and thus were part of the optimal set for protein for the entire 1950-2014 period. Similarly high consistency was observed for iron (10 of 12 original

optimal taxa). Consistent placement in the optimal set was present to a lesser extent for omega-3 fatty acids (five of the original 10 optimal taxa), vitamin A (11 of 19), and vitamin B<sub>12</sub> (four of nine). This relative consistency of optimal and sub-optimal sets, observed as it was over six and a half decades in which the species composition of landings underwent substantial change, suggests that the relative optimality of a species for yields of a given nutrient does not change rapidly.

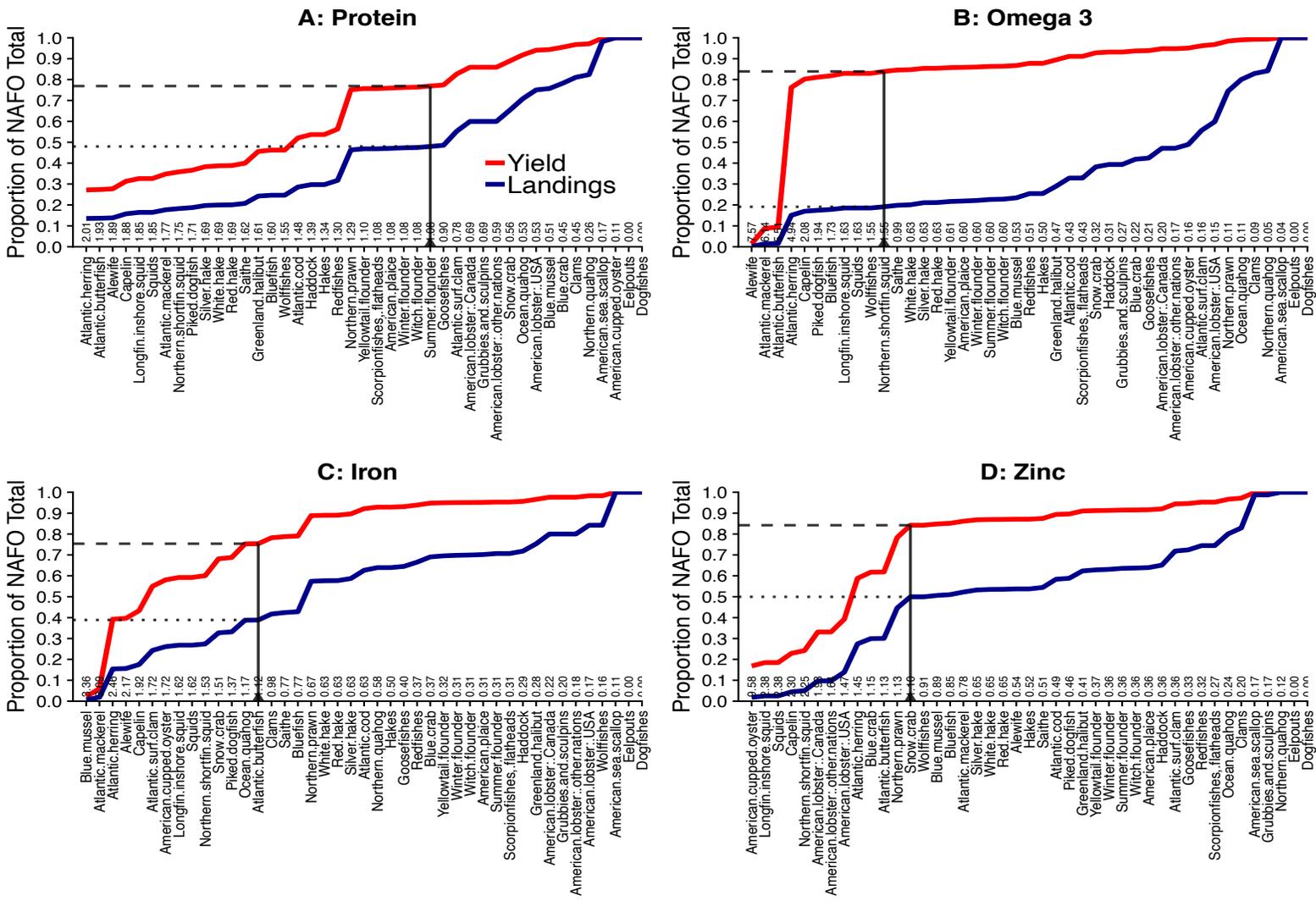
Zinc was a notable outlier: in 1950, the “optimal set” was composed of only one species, American cupped oyster (*Crassostrea virginica*). Due to its exceptionally high zinc content and relatively high landings at the time, oysters dominated zinc yields at that time, even in the absence of any intentional optimization efforts, contributing nearly 80% of annual zinc yields in the early 1950s (Chapter 2). Over the following decades, declining oyster landings led to an overall decrease in zinc yields, and allowed for the zinc contributions from other taxa to take on increasing relative importance (see Chapter 2). As a result, the optimal set for zinc was composed of nine taxa by the end of the study period. The zinc example is an illustration of how the optimality of a taxon’s contributions can only be understood relative to contributions from other taxa: a species, by itself, is not inherently “optimal” or “sub-optimal” for a nutrient, and thus the relative optimality of a taxon can change due to fluctuations in landings of other taxa.

### 5.3.2. Optimal yields, landings, and sets of taxa, 2010-2014

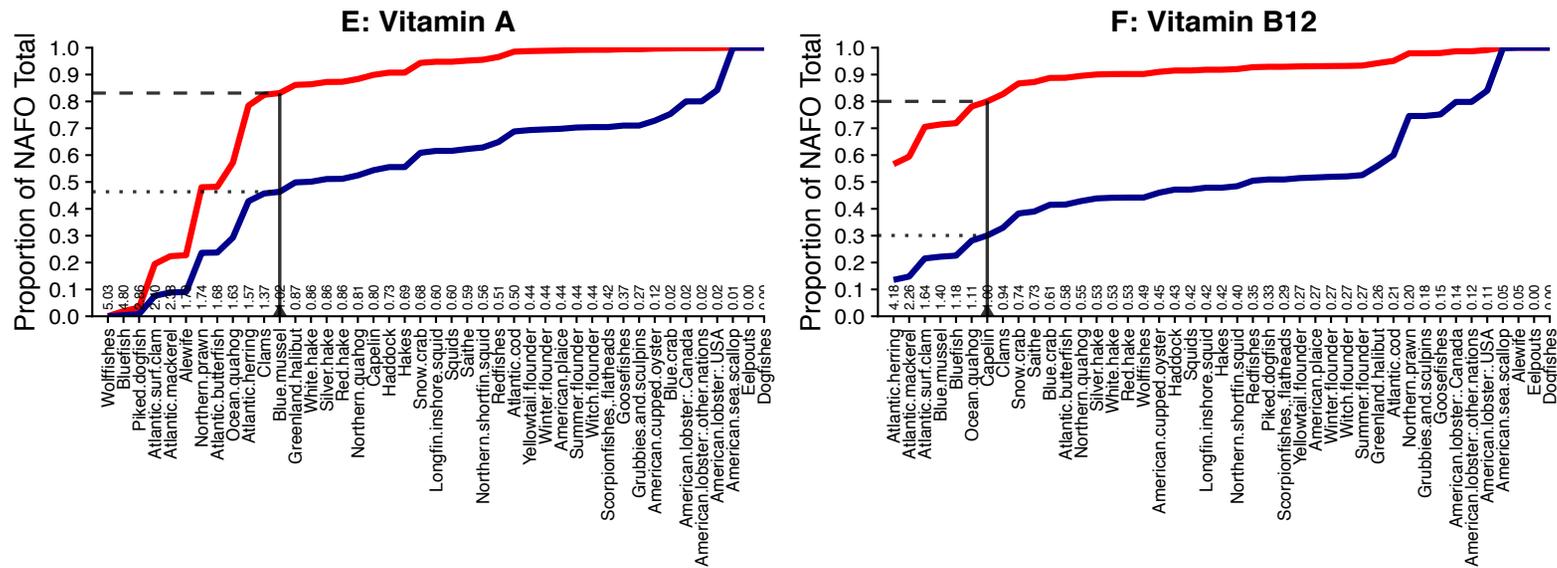
In the second part of the analysis, the most recent years in the data (2010-2014) were analyzed in detail, with particular attention paid to the optimal and sub-optimal sets and to potential approaches to aggregate optimization across nutrients.

Optimal yields, optimal landings, and optimal and sub-optimal sets of taxa are shown for the most recent years in the study (2010-2014) (Figure 5.2). Mean annual optimal yields ranged from approximately 75-84% of total yields. Optimal landings displayed a wider range, from approximately 19% of total mean annual landings (omega-3 fatty acids) to approximately 50% (protein, zinc). The optimization potential for these

nutrients over this period thus ranged from approximately 0.29 (protein) to 0.65 (omega-3).



**Figure 5.2.** 2010-2014 mean optimal yields, optimal landings, and optimal and sub-optimal sets of taxa ( $\alpha = 1.0$ ). Optimal yields are indicated by dashed line, optimal landings by dotted line, and the space between is the optimization potential. Taxa arranged on x axis from highest  $R$  value (left) to lowest (right), with  $R$  values shown above x axis. Optimal set of taxa is to left of solid black line (including the taxon touching the line), sub-optimal set is to the right.



**Figure 5.2. cont'd.** 2010-2014 mean optimal yields, optimal landings, and optimal and sub-optimal sets of taxa ( $\alpha = 1.0$ ). Optimal yields are indicated by dashed line, optimal landings by dotted line, and the space between is the optimization potential. Taxa arranged on x axis from highest  $R$  value (left) to lowest (right), with  $R$  values shown above x axis. Optimal set of taxa is to left of solid black line (including the taxon touching the line), sub-optimal set is to the right.

Optimal sets of taxa varied across the nutrients (Figure 5.2, Appendix E). For the years 2010-2014, only one taxon (Atlantic herring, *Clupea harengus*) had mean  $R$  values  $\geq 1.0$  for all nutrients, and thus only Atlantic herring were present in all optimal sets. Nearly half of all taxa (20 of 43) had  $R$  values  $\geq 1.0$  for one nutrient only; these taxa were thus present in optimal sets for one nutrient only. Thirteen taxa had  $R$  values  $\geq 1.0$  for three to five of the assessed nutrients, and six taxa had  $R$  values  $< 1.0$  across all nutrients and thus were not a part of the optimal set for any nutrient. Overall, for all nutrients other than protein, nutrient-specific optimal yields were achieved without contributions from the majority of the 43 assessed taxa. The number of taxa required for optimal yields ranged from seven (omega-3 fatty acids) to 14 (iron), while protein optimal yields required landings of 27 taxa (Figure 5.2, Appendix E).

### 5.3.3. Aggregate approaches, 2010-2014

In order to meet objectives for human health, nutrition-informed fisheries management may have to consider multiple nutrients at the same time. In this section, several potential approaches are compared.

#### 5.3.3.1. Aggregate approaches, all nutrients

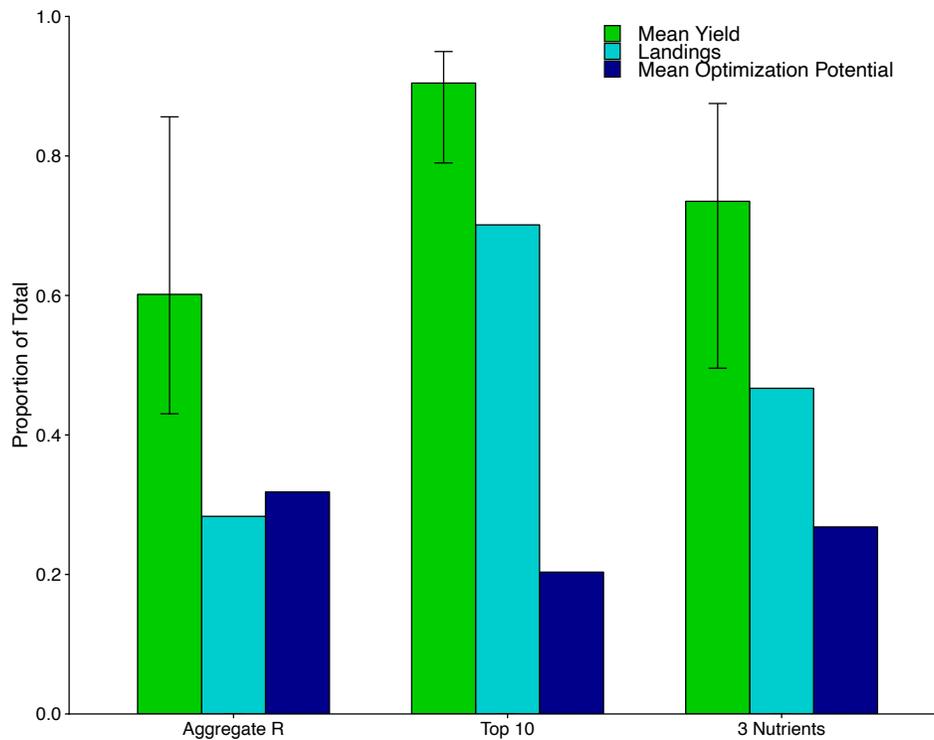
The following example illustrates the simultaneous consideration of multiple nutrients, with a hypothetical objective to develop an aggregate set of taxa for all nutrients in the 2010-2014 data, with no differences in preference for any nutrient. Three approaches addressed this challenge, with slightly different logics.

The formal approach is aggregate optimization ( $AR$ ). In the  $AR$  approach, Equations 5.1, 5.2, 5.3, and 5.7 were followed to generate  $AR_{z,ij}$  values, which were then used to determine the aggregate optimal set via Equation 5.4 (Table 5.2). Two informal approaches were also explored: “Top 10”, in which the aggregate set was composed of all taxa that had at least one  $R_{v,ij}$  value that was within the top 10 for any nutrient, and “3 Nutrients”, in which the aggregate set was composed of all taxa that had  $R_{v,ij}$  values  $\geq 1.0$  for  $\geq 3$  nutrients (see Methods section 5.2.3.1). The details of each

approach's optimal yields, landings, and sets of taxa, and optimization potentials are shown in Appendix F. Figure 5.3 provides a summary of the results.

Of the three aggregate sets of taxa that resulted from these selection criteria, the set formed by the *AR* approach had the highest maximum cumulative optimization potential (mean, across all nutrients) (Figure 5.3). Thus, the maximum difference between yields and landings, averaged over all nutrients, was highest with the mean *AR* approach than with either of the other approaches. The optimization potential is the objective function that the *AR* optimization measures developed in Chapter 4 are designed to maximize, and thus this example illustrates the apparent efficacy of the *AR* approach vs. two potential alternative approaches. However, as Figure 5.3 makes evident, the *AR* approach does not necessarily result in the highest mean nutrient yields, and it is also characterized by high variability between nutrients.

In contrast, the two informal approaches resulted in substantially higher yields, at higher levels of landings and lower optimization potentials, relative to the *AR* approach. The Top 10 approach, in particular, resulted in approximately 90% of nutrient yields across all nutrients, with much less variation between nutrients than did the *AR* approach (Figure 5.3). These results suggest that informal development of aggregate sets of taxa, informed by *R* values derive from single-nutrient optimization, may provide a pathway for addressing multiple nutrients simultaneously, while ensuring that yields and landings meet broader social objectives.



**Figure 5.3.** Mean nutrient yield, landings, and optimization potential for three aggregate approaches: the formal *AR* optimization approach, Top 10 (all taxa with a top-10 *R* value for any nutrient), and 3 Nutrients (all taxa with an *R* value >1.0 for  $\geq 3$  nutrients). Yield error bars show the minimum and maximum values of the nutrient-specific yields that compose the mean.

### 5.3.3.2. Shortcomings of the *AR* approach

From these results, it is apparent that the *AR* method, which again maximizes the *mean* difference between yields and landings across all nutrients, will allow for substantial variation in the yields of each nutrient. In this case, the *AR* approach resulted in yields of 0.43-0.85 (Appendix F.3). This variability is because the *AR* method rests on the selection of taxa based on their arithmetic mean *R* values across all selected nutrients. This method may therefore select taxa that have a mean *R* value that meets or exceeds the selection threshold, but that nonetheless make very little contribution to several nutrients.

The case of American cupped oyster is illustrative. While the oyster’s mean 2010-2014 *AR* value (2.02) was fourth-highest among all taxa, placing it well inside the *AR* optimal set, it was actually a poor contributor for four of the six nutrients, with

nutrient-specific  $R$  values of 0.11, 0.12, 0.16, and 0.45 for protein, vitamin A, the omega-3 fatty acids, and vitamin B<sub>12</sub>, respectively (Appendix E). However, the oyster's very high  $R$  value for zinc (9.58) overrode these multiple low values to result in an  $AR$  value that easily exceeded the  $\alpha$  value of 1.0.

In contrast, northern prawn had  $R$  values that exceeded 1.0 for three nutrients, yet it ultimately fell outside of the  $AR$  optimal set, with a mean 2010-2014  $AR$  value of 0.85 (Appendix E). Due to northern prawn's high levels of landings, its exclusion from the aggregate optimal set resulted in substantial decreases in that set's yields of protein, zinc, and vitamin A (Appendix F.3).

The broader implication is that nutrients that tend to be highly concentrated in a small subset of taxa may be disproportionately represented in an aggregate optimal set of taxa based on mean  $R$ , in comparison to nutrients that are more evenly distributed. This is because the former will result in very high  $R$  values for the few taxa in which they are concentrated, which may in turn override low  $R$  values for multiple other nutrients, while the latter will result in lower  $R$  values across more taxa, and thus be less likely to override low  $R$  values for other nutrients.

In the NAFO case, this may be illustrated by comparing two nutrients that show highly concentrated distribution (omega-3 fatty acids and vitamin B<sub>12</sub>) to a nutrient that shows a more even distribution (protein). As demonstrated by the Simpson's  $E$  values for these nutrients in Chapter 2's Figures 2.3 and 2.6, both the omega-3 fatty acids and vitamin B<sub>12</sub> are highly concentrated in relatively few taxa, while protein is more evenly distributed. This difference in distribution is reflected in the number of taxa that have mean 2010-2014  $R$  values  $\geq 1.0$  for these nutrients: while there were 11 and seven such taxa for the omega-3 fatty acids and vitamin B<sub>12</sub>, respectively, there were 27 such taxa for protein (Appendix E). However, those taxa that were optimal for protein were much less likely to be represented in the  $AR$  optimal set: whereas all 11 of the optimal taxa for the omega-3 fatty acids were represented in the aggregate optimal set, and six of the seven optimal taxa for vitamin B<sub>12</sub>, only 11

of protein's 27 optimal taxa were included (Appendix E). As a result, the aggregate optimal set's yields of the omega-3 fatty acids and vitamin B<sub>12</sub> were the highest of all nutrients, and its yields of protein were the lowest (Appendix F.3).

However, this variability of nutrient-specific outcomes does not change the conclusion that the selection of an aggregate optimal set of taxa based on mean  $R$  across selected nutrients will apparently result in the maximum arithmetic mean optimization potential, across the selected nutrients. Thus, if there is no preference for one nutrient over another, and if the only objective is purely one of maximizing the mean difference between yields and landings weights, the selection of taxa for which the mean  $R$  across all selected nutrients equals or exceeds the  $\alpha$  of 1.0 will result in mathematically optimal yields of the suite of nutrients as a whole.

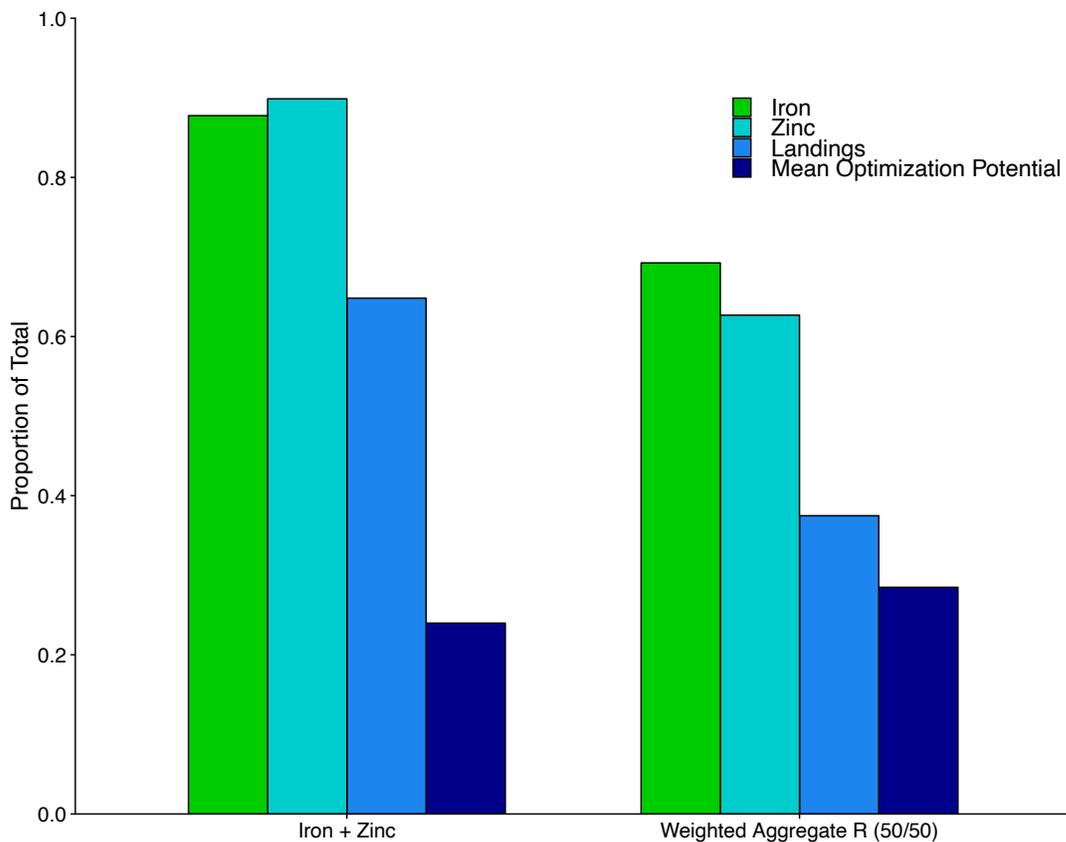
#### 5.3.3.3. Aggregate optimization, weighted

In the preceding example, all of the assessed nutrients were considered when calculating the  $AR$  values to build the optimal set. However, in cases where only a subset of nutrients are of concern, the aggregate optimal set of taxa may be developed to optimize yields of only that subset of nutrients. Furthermore, a subjective weighting system may be applied to more heavily weight those nutrients that are of highest concern (e.g., based on the needs of the dependent human population), thus ensuring that the resulting optimal set will return optimization potentials that are similarly weighted.

To illustrate the weighted aggregate optimization process, consider an example in which stakeholders wish to use the 2010-2014 NAFO data to optimize iron and zinc yields relative to landings biomass, with an equal preference for each nutrient and with no consideration of consequences for other nutrients. They develop two approaches, described in section 5.2.3.2: the weighted aggregate ( $WAR$ ) method and "Iron + Zinc", in which the aggregate set of taxa is composed of all taxa in the iron and zinc optimal sets (see Figure 5.2).

The details of the optimal yields, landings, and sets of taxa for the Iron + Zinc and *WAR* approaches are shown in Appendices F.4 and F.5, respectively. Figure 5.4 summarizes these two approaches' iron and zinc yields, landings, and optimization potentials. What is immediately apparent is that the Iron + Zinc approach resulted in higher yields of iron and zinc, with yields of both nutrients approaching 90% of their mean annual 2010-2014 NAFO values. However, landings were also higher in this approach, at approximately 66% of the mean annual 2010-2014 NAFO total. The maximum optimization potential achieved by the Iron + Zinc approach (averaged over iron and zinc) was thus approximately 0.24 (Figure 5.4). In comparison to scenario 3, the *WAR* approach resulted in lower yields of iron and zinc, but also required substantially lower landings to achieve these yields. As a result, the maximum optimization potential (when weighted equally between the nutrients as per the initial conditions of the weighted model) was approximately 0.28. Compared to the approach of combining both nutrients' optimal sets (scenario 3), the *WAR* model, as predicted, produced an optimal set of taxa that achieved a higher maximum optimization potential across the selected nutrients (Figure 5.4).

This example illustrates the efficacy of the *WAR* approach to maximize the mean optimization potential across multiple nutrients. However, it also illustrates the potential for *WAR* approaches to result in scenarios that may not achieve other objectives (e.g., yields that do not meet a minimum threshold).



**Figure 5.4.** Yields, landings, and optimization potentials for the Iron + Zinc and weighted aggregate (WAR) approaches.

## 5.4. Discussion

The results of single-nutrient optimization exercises show that yields of single nutrients could be maintained at relatively high levels even if NAFO landings were reduced substantially. For the 1950-2014 period, when evaluated on a single-nutrient basis, the average annual optimal yield (across all nutrients) was approximately 75% of total mean annual yields, and the associated mean optimal landings was approximately 35%. Thus, on average, the last 25% of nutrient yields required 65% of the landings biomass, when optimized on a single-nutrient basis. Similarly, single-nutrient optimization for the 2010-2014 period resulted in 75-84% of total yields, while requiring 19-50% of annual landings (depending on the nutrient).

Optimizing across multiple nutrients results in a more complicated picture. Strictly mathematical optimization of multiple nutrients returns the set of taxa that will yield

the maximum average difference between yields and landings, but may do so at yield and/or landings levels that are insufficient for other social objectives. When multiple nutrients are to be considered at the same time, flexibility in the approach, informed by but not limited to a purely mathematical optimization approach, may allow for aggregate scenarios that best achieve multiple social objectives.

In such efforts, the information that can be obtained from single-nutrient optimization analysis – particularly, the index of optimality  $R$  – can be used to build sets of taxa that achieve multiple objectives. For example, one multi-nutrient approach that seemed to strike an informal balance between mathematical optimization and maintaining high nutrient yields was simply to exclude those taxa that had relatively low  $R$  values across all nutrients (the “Top 10” approach). This method resulted in nutrient yields that were consistently near or above 90% of average annual levels, while requiring approximately 70% of average annual landings (Appendix F.1).

The relative efficacy of the “Top 10” approach (i.e., excluding lowest contributors across nutrients) hints at another conclusion: many taxa were relatively minor contributors to nutrient yields. For all nutrients other than protein, single-nutrient optimization for the years 2010-2014 resulted in optimal sets that included less than half of assessed taxa. For multiple nutrients, the “Top 10” approach (which excluded any taxon that was not in the top 10  $R$  values for any nutrient) resulted in the exclusion of 17 of the study’s 43 taxa. Exploitation of these minor contributors could therefore be reduced with disproportionately low consequences for the assessed nutrients.

However, by focusing nutrient yields on smaller subsets of taxa, the evenness of nutrient yields across taxa is reduced, with potential consequences for the resilience of nutrient yields. Analysis of changes to nutrient evenness across taxa (e.g., by monitoring Simpson’s  $E$ ) and concentration in biomass (e.g., by monitoring  $B_{80}$ ) could be a component of optimization efforts. Simpson’s  $E$  and  $B_{80}$  are developed, and their relevance to nutrient yields is discussed, in Chapter 2.

Several lines of evidence suggest that optimal scenarios for the NAFO region may require relatively little adjustment on a year-by-year basis. For many nutrients, the optimal sets of taxa showed little turnover from one year to the next: for all nutrients other than zinc, 44% - 84% of the taxa that were “optimal” in 1950 remained optimal for the entire duration of the study. Many decade-scale mean optimal yields, optimal landings, and optimization potentials showed relatively small standard deviations (Table 5.3). These results suggest that assessment of NAFO fisheries’ performance relative to nutrient optimization models may not require annual updates and may instead be accomplished with updates at broader time intervals.

Improvements in data needed for optimization will improve the applicability and usefulness of the results for real-world situations. The information available to estimate edible biomass yield rates and nutrient content rates, in particular, often ranges from poor to non-existent; any efforts to integrate nutrient optimization into a fisheries science and/or management apparatus must ensure that these types of data are available or will be obtained.

## **5.5. Conclusion**

This approach to nutrient optimization analysis of fisheries is binary in its definition of optimal and sub-optimal taxa, but its utility is not predicated on binary exploitation decisions. The nutrient optimization approach developed in Chapter 4 and illustrated here in Chapter 5 identifies a single point at which the optimization potential is at its maximum. As such, this approach requires the separation of species into two groups – one optimal, and one sub-optimal. However, such stark binary separation is unlikely to be a feasible or desirable basis for exploitation decisions. Complex relationships between species, fisheries, and fisheries infrastructure preclude the simple excision of a group of species from a multi-species, multi-fishery context. Optimal and sub-optimal species may be caught alongside one another, across multiple fisheries. Fishing, processing, and market capacity may be sustained by exploitation of sub-optimal species, to the extent that their sudden exclusion would

preclude fisheries for the optimal species. Some of the apparently optimal species may simply be minor species, with catches or values that are insufficient to support directed exploitation. Furthermore, of course, in any real-world context, there will be myriad social and cultural arguments against the wholesale exclusion of multiple taxa based on nutrient yield outcomes alone.

Thus, this approach is not intended to result in binary exploitation decisions, nor is its utility predicated on such outcomes. Rather, the methods presented in this chapter may have a variety of useful applications. At the most basic level, nutrient optimization analysis can yield a wealth of information regarding the nutrition yield consequences of different decisions. The calculation of  $R$  values provides a simple index of each taxon's relative contribution to nutrient yields. Arranging taxa by descending  $R$  value and calculating the cumulative nutrient yield and landings at each taxon provides an easily-understood depiction of nutrient yield capacity, and associated landings, across all assessed taxa. This information can be easily accessed to support decision-making processes. The optimal nutrient yield, found at the last taxon for which  $R \geq 1.0$ , can serve as a formal or informal reference point against which different exploitation scenarios can be compared. By virtue of containing all of this information, the single-nutrient optimal nutrient yield curves (Figure 5.2) would seem to provide a simple, coherent tool for incorporating nutrient yield considerations into fisheries decision-making processes.

Overall, the results of this chapter suggest that declines in catches do not necessarily have to result in an equal decline in nutrient yields. If catch reductions are required for conservation reasons, optimization analysis can inform the decision-making process, potentially mitigating nutrient yield declines. However, the reverse is also true: declining catches of a taxon that is of particularly high importance for a given nutrient may result in disproportionate reductions in total yields of that nutrient. Declining landings of Atlantic herring, for example, would have disproportionately large negative consequences for yields of multiple nutrients in the NAFO region (Figure 5.2). For this reason, a key use of nutrient optimization analysis is to identify

important sets of taxa for specific nutrient yields, and thus focus research and management efforts.

## **Chapter 6. Summary and Conclusions**

### **6.1. Context for the dissertation**

Fisheries make a variety of key but often overlooked contributions food security and human nutrition (HLPE, 2014; Béné et al., 2016). Perhaps the clearest direct connection between fisheries and human diets is in fisheries' contributions to the availability of dietary nutrients for at-risk human populations. Recent research indicates that the strategic use of fisheries products could address specific instances of malnutrition, particularly that associated with micronutrient deficiencies (Hicks et al., 2019). However, this important social function of fisheries is challenged by the threats to the sustainability of fisheries, as potentially millions of people may be exposed to micronutrient deficiencies due to downturns in catches (Golden et al., 2016a).

The rising awareness of the importance of fisheries for dietary nutrient availability is occurring against a backdrop of uncertainty regarding the future of marine fisheries. Globally, the future of fisheries in the short and medium-term is fraught with a variety of challenges caused by overexploitation (both historic and ongoing) and climate change. The available information does not suggest that global catches will increase in the near future.

The emerging context for global fisheries, then, is one of guiding fisheries toward a future in which exploitation is broadly reduced for many stocks and ecosystems, and thereafter is maintained at sustainable levels. However, the steps required to achieve this future may have negative effects on dependent human populations if they are undertaken without consideration of their consequences for the availability of key nutrients. Just as the scope of fisheries management has been expanded to include fisheries' broader ecological impacts via ecosystem-based management (Pikitch et al., 2004; Curtin & Prellezo, 2010), so too may the development and adoption of "food-informed management", comprising information and objectives for nutrition yields as well as the variety of other dimensions of fisheries' contributions to

food security, help to inform fisheries management (and science) with crucial insight – insight that is currently missing.

There are a number of considerations that may be included in a fisheries management approach that is informed by food security insights (McClanahan et al., 2015). In my review of the literature, I recognized a gap that seemed to be fundamental to addressing the nutrition aspect of such management: fisheries yields are currently characterized, quantified, and analyzed in terms of the weight of the catch, as well as in economic terms. Neither catch weights nor economic information can be assumed to indicate nutrient yields, though, due to heterogeneity in edible biomass yield rates and nutrition content rates across taxa. Thus, despite the increasingly apparent need to inform our fisheries decisions with nutrition information, we are still “flying blind” when it comes to understanding key information about the nutrients yielded by fisheries: their trends, the important taxa that support them, and so on. This dissertation’s research is meant to be a starting point for addressing this gap.

## **6.2. Findings and contributions to knowledge**

In Chapter 2, I sought to address research questions 1 and 2 by analyzing nutrient yields from seven decades of fisheries landings in the Northwest Atlantic Fisheries Organization (NAFO) region. This chapter truly represents starting from scratch: as the first study, to my knowledge, to analyze trends and patterns in nutrient yields from a specific region over time, the methods and indicators were invented and/or adapted from other disciplines as the need for them became apparent. The indicator  $B_{80}$ , which is a measure of how much of a region’s total landings biomass is required to obtain the first 80% of a nutrient’s yield, is novel. Simpson’s evenness ( $E$ ) was, of course, a long-established concept in assessments of biodiversity, but to my knowledge its adaptation for use as an indicator of the evenness of nutrient yields across taxa is novel. The distribution of nutrient yields across trophic levels and the calculation of a resulting mean trophic level (MTL) is a simple adaptation of a method for analyzing catch distribution across trophic levels, and as such is

closest to a well-established method. All told, Chapter 2 is best viewed as an introductory exploration of a potentially new way of looking at fisheries and their yields.

The exploration undertaken in Chapter 2, being essentially a novel exercise, yielded a number of insights that to my knowledge will be new to the relevant literature. Relative to their initial yields at the start of the study period, the assessed nutrients increased more and subsequently declined further than did landings weights. Landings became more evenly distributed across taxa over time, but in comparison, nutrient yields remained more concentrated in fewer taxa, with the omega-3 fatty acids and vitamin B<sub>12</sub> exhibiting particularly high concentration in few taxa. Of the 43 assessed taxa, Atlantic herring (*Clupea harengus*) emerged as the cornerstone of NAFO nutrient yields, to a degree that suggests the potential for a lack of resilience in the ongoing yields of several nutrients. Some taxa were identified as contributing to nutrient yields at a disproportionately high level when compared to their landed weights (e.g., Atlantic herring, zinc yields derived from American cupped oyster, *Crassostrea virginica*), while others were found to yield disproportionately low amounts of assessed nutrients (e.g., American sea scallop, *Placopecten magellanicus*). The indicator  $B_{80}$  suggests that NAFO's yields of many nutrients in recent years could be largely maintained with substantial catch reductions.

Chapter 3 evaluated the nutrient yield consequences that result from the use of Atlantic herring as bait in the Maine fishery for American lobster (*Homarus americanus*). Despite substantial uncertainty, the results indicate that the Maine lobster fishery is likely a net consumer of most of the assessed nutrients. To my knowledge, this is a novel finding in fisheries research – the possibility that a fishery can consume more nutrients, in the form of bait, than it ultimately produces for humans. In light of the aforementioned status of Atlantic herring as the cornerstone of recent NAFO nutrient yields (Chapter 2) and the current declines in Atlantic herring recruitment, stock status, and catches (NEFSC, 2018a), the scale of Atlantic herring bait use in the Maine lobster fishery, and the consequences for net nutrients

made available to humans, is troubling. This chapter thus partially answers the second research question: by evaluating these two fisheries from a nutrient yield perspective, we learn that it may be possible for a fishery to consume more nutrients, in the form of potentially edible biomass used as bait, than it produces through its own landings.

In Chapter 4, I sought to develop an approach by which nutrient yields might be optimized. I began from Gordon's (1954) definition of the "optimum degree of utilization" of a fishing grounds as being that which maximized the net economic yield (the difference between revenues and costs). I attempted to re-cast this definition in terms of nutrient yields and catch weights, such that the mathematical optimum degree of utilization would be that which maximized the difference between yields and catch weights. The result of this effort was the development of an approach that organizes taxa based on the ratio ( $R$ ) of their contributions to the nutrient vs. their contributions to total landings weights. By organizing taxa in descending order of  $R$ , and taking the cumulative nutrient yield and landings weight at each successive taxon, the mathematical optimal nutrient yield, landings weight, and set of taxa are found at the last taxon for which  $R$  exceeds or equals 1.0. I concluded this chapter by showing how the  $R$  approach can be modified to produce optimal scenarios for more than one nutrient.

Finally, in Chapter 5, I applied this approach to the NAFO landings and nutrient yields data that were used in Chapter 2. The results of the optimization exercises in Chapter 5 support the initial insight presented in Chapter 2: NAFO's nutrient yields in recent years could be largely maintained even with substantial reductions in catch levels. Furthermore, recent optimal yields of most nutrients required landings from less than half of assessed taxa. Over a longer time period (1950-2014), single-nutrient optimization returned optimal yields that were, on average, 74% of mean annual yields, while requiring only 35% of mean annual landings. The results of this chapter indicate that the optimization approach defined in Chapter 4 appears to achieve the objective, which is to maximize the optimization potential (i.e., the

difference between nutrient yields and catch weights) for both single nutrient and multiple nutrient scenarios. However, for scenarios in which multiple nutrients are simultaneously considered, there are informal approaches to developing aggregate sets of taxa that may achieve near-optimal levels of optimization potential maximization, while also resulting in higher total yields of the selected nutrients.

Taken as a whole, it is my hope that my dissertation provides a road map for the analysis, monitoring, and optimization of nutrient yields from fisheries. It is, emphatically, an exploration: there was no existing conceptual outline, much less any accepted methods or indicators, to guide this research. This dissertation may provide the start of such a guide and thus catalyze the development of methods and indicators for the analysis, monitoring, and/or optimization of nutrient yields in fisheries.

### **6.3. Implications and applications**

It cannot be assumed that commercial fisheries are currently organized and executed in a way that is aligned with sound nutrient yield outcomes. Efforts to improve this alignment can be undertaken by a range of actors involved in the assessment, management, production, and distribution of seafood, from policy-writing bureaucrats to chefs.

In some situations, it may be beneficial to increase demand (and thus prices) for highly nutritious species. Commercial fisheries are fundamentally a profit-seeking enterprise (Sethi et al., 2010) and therefore fisheries for highly nutritious species may not fully develop if prices are too low. Or, as in the case of Atlantic herring in the Maine lobster fishery, the cheap yet highly nutritious species may be used for purposes other than direct human consumption. Where highly nutritious species are currently undervalued by seafood consumers, demand can particularly be motivated by tastemakers: chefs and restaurants, for example, may play a key role in developing and driving popular adoption of novel tastes and species. Cheap sources of highly nutritious seafood may be critical to the diets of food-insecure people,

however, so care must be taken that such highly nutritious species are not priced out of reach of dependent populations. To this end, the development and promotion of nutritious foods from currently under-valued components of seafood (e.g., fish bone broths) could simultaneously improve fisheries' nutrient retention while providing affordable sources of nutrition for at-risk populations.

Non-governmental organizations may have a variety of roles to play in improving the alignment of commercial fisheries with nutrient yield outcomes. On the demand side, these organizations can work directly with chefs, seafood retailers, processors, and fishing fleets to improve demand for, and/or utilization of, highly nutritious species. Organizations that produce third-party sustainability assessments or certifications can incorporate nutrition information into their evaluation criteria. On the policy side, these organizations can evaluate a region's current fisheries approaches against theoretical optimal scenarios in order to identify areas for improvement (e.g., by ensuring that key species for specific nutrients are recognized by science and management efforts).

For industry, efforts to improve alignment with nutrient yield outcomes may present both opportunities and challenges. Wherever highly nutritious species are drastically undervalued in the market, opportunities may exist for improving demand. In these scenarios, collaborations with chefs, restaurants, and non-governmental organizations may present opportunities for improvement. Conversely, industry may also identify, within their own activities, actions that are leading to the waste or inefficient use of nutrients. Fisheries that use whole fish for bait may be prime candidates for improvement: the Maine lobster fishery, for example, may substantially improve its nutrient yield profile by adopting alternative baits. At-sea discarding of species that are highly nutritious but of low economic value may also represent a key vector by which fisheries waste nutrients. Similarly, processing decisions that result in the waste of potentially edible parts of the animal – e.g., at-sea processing that results in large-scale discards of fish heads – may be revisited. Although neither of these latter two issues were directly assessed in this

dissertation, it stands to reason that improved retention of nutrient-rich species and processing by-products would improve fisheries' alignment with nutrition objectives. In such scenarios, partnerships and support from non-governmental organizations or governments could be pursued if market demand for such products is so low that retention would otherwise be economically prohibitive. By proactively addressing these and other sources of inefficiency, industry can demonstrate a commitment to improving the alignment of their operations with sound nutrient yield outcomes, while also adding value to the products obtained from their fishing effort.

The issue of post-processing waste was not explicitly assessed in this dissertation, but it likely presents a promising avenue for improving the retention of nutrients obtained from fisheries. This is particularly likely in situations where heads, roe, and small bones are routinely discarded, such as in at-sea processing. These components of the body may be nutrient-dense: fish eyes can provide high amounts of vitamin A, for example (Roos et al., 2002), and edible fish bones can be an excellent source of calcium (Larsen et al., 2000). In this dissertation, an example of the consequence of eating some parts of the animal and not others may be found in the comparison of nutrients obtained from cupped oysters and sea scallops. While both the meat and the viscera of oysters are consumed, only the meat of scallops is consumed; this may contribute to the oyster's much greater contributions to NAFO nutrient yields (see Chapter 2). Depending on the context, a variety of approaches may be employed to improve retention and use of nutritious components of fish and shellfish. In the case of at-sea processing, for example, partnerships between industry and community food organizations could support the retention of fish heads that would otherwise be discarded.

Finally, there are implications for fisheries policy. When policy fails to require explicit consideration of nutrient yields, the resulting science and management apparatus will be "flying blind" when it comes to this issue. In such situations, exploitation decisions may result in any number of unforeseen consequences for nutrient yields. More broadly, fisheries policy that fails to consider nutrients may itself encourage the

coupled fishery-ecosystem to proceed along trajectories leading to increasingly sub-optimal nutrient yield scenarios. This issue may be particularly pronounced in the case of subsidies. Clearly, capacity-enhancing subsidies (Sumaila et al., 2016) may drive fisheries toward less-ideal nutrient yield outcomes in a number of ways, such as encouraging the expansion of fleets targeting species of low nutrient yield, supporting fisheries that engage in the incidental catch and discard of nutritious species, and supporting the use of highly nutritious species as bait. Even beneficial subsidies, such as the use of public funds to support science and management efforts (Sumaila et al., 2016), may have unintended negative consequences if they are allocated to species and/or fisheries that are economically lucrative but of relatively low nutrient value.

The good news is that it may not be difficult to incorporate nutrient considerations into policy, and subsequently into management and science. The information and methods necessary to assess and monitor nutrient yields from fisheries may be relatively simple and straightforward, particularly when compared to the complexity of the information sources, models, and assumptions that characterize other aspects of fisheries and oceans science and management. The approaches that I develop in this dissertation could be employed by scientists, managers, and stakeholders to form a composite, multi-faceted conceptualization of the relationships between a region's exploited taxa and its yields of important nutrients. Including this nutrient-yield information could inform decision-making processes at multiple levels, from the policy level (e.g., identifying priorities for a region's fisheries) to management strategy evaluation (e.g., assessing different management options for their consequences for nutrient yields) to in-season management (e.g., the setting of Total Allowable Catches). To this end, several components of this dissertation may be particularly useful.

At the policy level, nutrition-informed management may consider the issue of adaptive capacity for the purposes of community resilience (McClanahan et al., 2015). Whereas a strong reliance on few sources of seafood or few fisheries may

expose the community to uncertainty-associated risk, diversifying the seafood supply may help to make affected communities more resilient to environmental, economic, and social shocks (Bell et al., 2009). Simpson's  $E$  (Chapter 2) can serve as an indicator of the distribution of nutrients across taxa, which has consequences for adaptive capacity and resilience in nutrient yields.

One consideration for both risk assessment and nutrient-informed management strategy evaluation may be to monitor the degree to which landings can be reduced before affecting nutrient yields. The indicator  $B_{80}$  (Chapter 2) conveys information about the relative concentration of nutrients across landed biomass, which has implications for the relationship of landings to nutrient yields. A lower  $B_{80}$  value indicates greater potential for nutrient yields to remain comparatively unaffected even in the case of declining total landings; conversely, a relatively high  $B_{80}$  value suggests less potential for total landings and nutrient yields to be decoupled.

Risk assessment and management strategy evaluation may also be informed by understanding and monitoring differences in specific nutrients' ecological origins. This may be of particular importance when a nutrient is distributed across many taxa: when yields are aggregated by trophic level, complex distributions across many taxa can be simplified and coherent patterns may become apparent. In the case of NAFO, for example, protein yields are relatively evenly distributed across many taxa (relatively high Simpson's  $E$ ) and across biomass (relatively high  $B_{80}$ ), but show a noted reliance on higher trophic level finfish (high MTL). Recent zinc yields have similar Simpson's  $E$  values as protein, and slightly lower  $B_{80}$ , but show a different trophic pattern, with greater reliance on invertebrates. This insight is informative because it suggests that yields of these two nutrients may be affected by different sets of factors: changes in finfish catches due to warming waters, for example, may affect protein yields more than zinc yields. This information may be useful to guide research and management efforts (e.g., by identifying nutrients that may be at higher risk of decline due to environmental changes), and to inform stakeholders of potential risks to ongoing availability of specific nutrients.

Finally, optimal nutrient yield analysis (Chapters 4 and 5) may inform management strategy evaluation by management agencies, and may also be used by non-governmental organizations and academics to evaluate status quo and/or alternative fisheries arrangements. In these contexts, the use of optimal nutrient yield analysis would be to provide a theoretical nutrient-optimized configuration of catches, against which alternatives can be assessed. While the method takes a binary approach to the designation of optimal vs. sub-optimal sets of species, it is not meant to result in similarly binary exploitation decisions, nor are its results limited to such use. Arranging taxa by descending contributions to nutrient yield ( $R$  value), and calculating cumulative nutrient yield and landings at each step, results in an easily-understood depiction of how a region's nutrient yields are obtained across all taxa, and allows for rapid assessment of the consequences of different exploitation options. The mathematical optimal nutrient yield and associated optimal landings can serve as formal or informal reference points against which different exploitation scenarios can be assessed, and the mathematical optimal and sub-optimal sets of taxa can likewise serve as references for different catch options. These applications of optimal nutrient yield analysis may be particularly useful in situations where enhanced management efforts will reduce catches in fisheries that supply key nutrients to highly-dependent human populations.

#### **6.4. Limitations and areas for future research**

In my research, the dominant challenge was in the availability and quality of data. In particular, the available information for edible biomass yield rates (i.e., the proportion of a taxon's biomass that is ultimately edible) was extremely poor. For most edible biomass yield rate estimates, I relied upon a decades-old monograph from the United Nations Food and Agriculture Organization (FAO, 1989). The author of that monograph repeatedly noted the poor quality of much the information available at the time. Given the importance of the issue of seafood for human nutrition, and the implications of even small changes in estimates of edible biomass yield rate for resulting estimates of nutrient yields (given the vast amounts of catch to which some

of these edible biomass yield rates are applied), it is remarkable that the data are so poor. A key “next step” in research is thus to compile reliable estimates of the amount of biomass that is ultimately edible for major marine taxa, ideally broken down into sub-categories to account for different nutrient concentrations for different types of edible biomass (e.g., eyes, flesh, bones, roe), all of which must be attended by clear and consistent documentation of the methods used to arrive at the estimates, as well as estimates of sampling variability.

An associated challenge was, essentially, uncertainty about uncertainty. As discussed in greater detail in Appendix A.3, uncertainty was treated in a variety of ways for the estimates of catch, edible biomass yield rates, and nutrient content rates that I obtained from the literature. For example, there were simply no estimates of uncertainty available for many taxon-specific edible biomass yield rates. Faced with this challenge, I endeavored to develop an approach to characterizing uncertainty (Appendix A.3) that could be applied across all cases and result in uncertainty bounds that would be conservative (i.e., err on the side of producing wider uncertainty boundaries, rather than narrower, as the available information deteriorated in availability, quality, and/or agreement). However, there is the possibility that this approach did not capture the full range of uncertainty in all cases, and thus that it may sometimes give false confidence.

The approach that I developed in Chapter 4 for determining mathematical optimal nutrient yields, landings, and groups of taxa appears to be successful relative to the stated objective: to maximize the difference between nutrient yield and catch weight. This definition of optimal, as discussed in Chapters 4 and 5, has a narrow focus on mathematical optimization of yields relative to landings; therefore, its results should not be construed as indicating “optimal” outcomes for any broader set of social or ecological objectives. Furthermore, the formal approach to optimization across multiple nutrients may result in much greater yields of some optimized nutrients than others. The result may be “optimal” in the sense that the mean optimization potential across all assessed nutrients is at its maximum point, yet may not achieve nutrient-

specific outcomes. For these reasons, the optimization approach that I put forward in Chapter 4, and illustrate in Chapter 5, should be understood to generate outcomes that are “optimal” only relative to one specific objective: the mathematical maximization of the difference between nutrient yields and catch weights.

The research presented in Chapters 2-5 was simplified in some regards. One notable simplification was that I did not include any consideration of the dietary needs of the people who actually depend on NAFO fisheries for nutrition. In real-world scenarios, the evaluation of nutrient yields, and in particular efforts to define optimal nutrient yield scenarios, would ideally be informed from the start by information about the dependent population’s nutrition needs, and by a larger suite of information regarding the various ways in which their food security is interrelated with the assessed fisheries.

## **6.5. Conclusion**

Nutrient-specific analyses of fisheries, using methods such as those put forward in this dissertation, may help shape fisheries decision-making across scales of space and time. Over the short term, as discussed throughout this dissertation, these analyses may help scientists, managers, and stakeholders understand the key species for yields of specific nutrients in localized contexts, and by doing so, improve their understanding of the consequences of their decisions. In the medium term, the application of these approaches may shift science, management, and conservation efforts toward a greater focus on those taxa that support human dietary needs. They may also provide another perspective into the consequences of potentially harmful subsidies. Ultimately, over the longer term, the cumulative effect of these short and medium-term changes may be to shift fisheries’ focus toward species that provide nutrition for humans with the least associated environmental impact.

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## Appendix A. Methods for Chapter 2

### A.1. Species selection

Landings data were obtained from the reconstructed catch data for the northwest Atlantic available through the Sea Around Us database (SAU RFMO 10 v45) (Pauly & Zeller, 2016b). The global catch dataset to which the northwest Atlantic set belongs is “reconstructed” in that it represents the results of an effort to quantify all fisheries catches, including those that have typically been excluded from the United Nations Food and Agricultural Organization (FAO) global landings data (Pauly & Zeller, 2016a). The FAO data, for example, do not include catches that are discarded at sea, and may underestimate small-scale commercial and subsistence fisheries (Pauly & Zeller, 2016a).

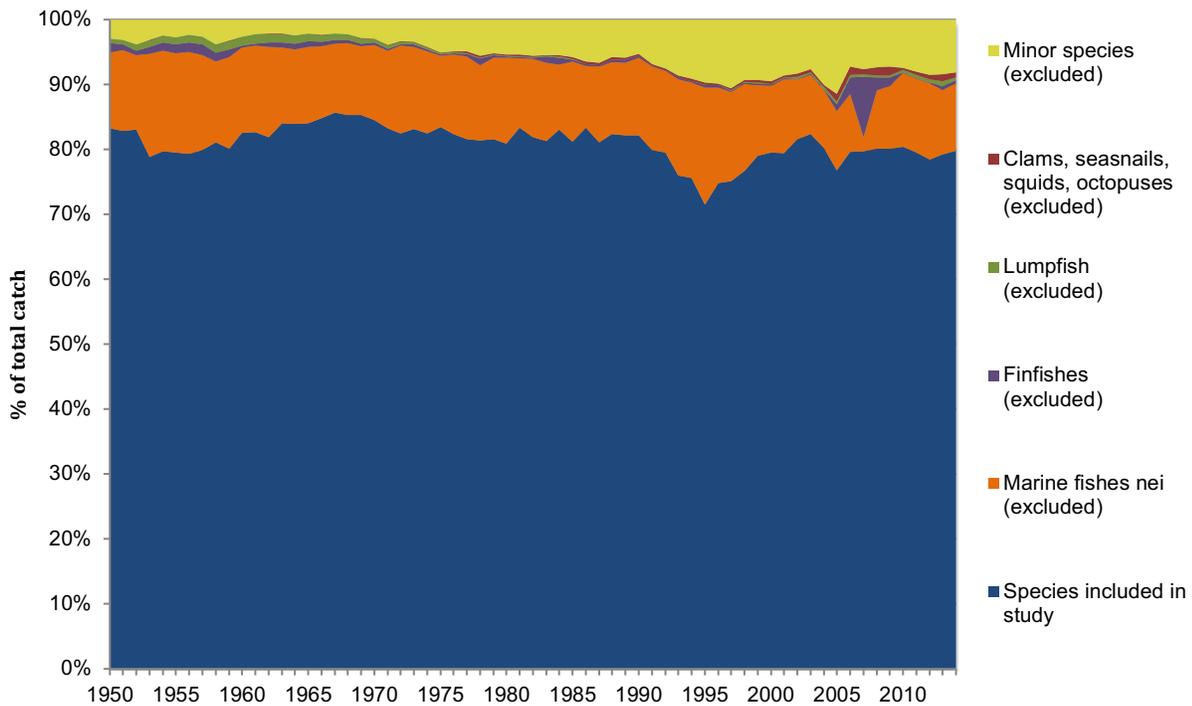
Taxa were selected for inclusion in this study based on their contributions to total catches (i.e., landed catch + catches discarded at sea), edibility (to humans), and the availability of nutrient information at the reported level of taxonomic resolution. There were 383 species and species groups with catch records in the region’s 1950-2014 reconstructed catches. However, 95% of the total 1950-2014 catch was supplied by a total of only 44 species/species groups. These 44 species/species groups were initially selected for inclusion in this study. In addition, goosefishes (Lophiidae) were included, as their catches contributed >1.0% of total catches for a number of years beginning in the 1990s. None of the species excluded at this step contributed more than 1.0% of any one year’s total catches, with the exception of roundnose grenadier (*Coryphaenoides rupestris*) in 1971 (1.08% of total catches) and the multi-species group “herrings, shads, and anchovies” (3.0% and 3.8% in 2004 and 2005).

Menhaden (*Brevoortia tyrannus*) were not included in this analysis, as this taxon is fished entirely for reduction to fishmeal and fish oil and is widely considered inedible due to its boniness, odor, and taste. Indeed, we were unable to find published estimates of edible biomass yield rates, or nutrient content per unit of edible

biomass, for menhaden. While we therefore cannot estimate the nutrient yields that could theoretically be obtained from menhaden if it were to be directly consumed, we note that total menhaden landings in a later version of the SAU NAFO data (SAU RFMO 10 v47) were similar to those of Atlantic herring over the entire assessed period. Thus, we speculate that the nutrients embodied in the edible flesh of menhaden would rival yields from other major NAFO taxa, if menhaden landings were to be directly consumed by humans.

In order to apply nutrient conversion factors to catches, the reconstructed catches must be identified to an appropriate level of taxonomic resolution. Of the 45 species and species groups that qualified based on contribution to total catch, three were excluded due to insufficient taxonomic resolution. The largest group, by total catch, that did not allow for the application of conversion factors, “marine fishes nei”, contributed the second-most total biomass to the 1950-2014 total catch (12.07% of the total). It is not possible to know which species were recorded in this generic category, and therefore no nutrient conversion factors could be applied to its catches. The other two groups, “finfishes” and “clams, seasnails, squids, octopuses” contributed 0.70% and 0.20% of total 1950-2014 catches, respectively. Conversion factors cannot be applied to these groups due to their lack of sufficient taxonomic resolution. Thus, these groups were removed from both the reconstructed catch data and the converted estimates of nutrient yield. For similar reasons, lumpfish (*Cyclopterus lumpus*), were also excluded due to excessive uncertainty regarding the nutrient content of their catches and landings.

The result of these removals is that nutrient conversion factors were applied to 41 taxa (see Appendix B; note that American lobster are divided into those caught in USA, Canadian, and other nations’ fisheries, for a total N=43). These taxa cumulatively contributed approximately 81.8% of the region’s total catch, excluding menhaden catch, from 1950-2014 (annual range: 71.5% - 85.7%; Figure A.1.).



**Figure A.1.** Annual catches included and excluded from study.

## A.2. Conversion factors

To estimate yields of nutrients from fisheries catches requires the application of two sets of conversion factors. The first factor converts the weight of the catch to the weight of edible biomass that is available for human consumption. The second set of factors converts this edible biomass yield to yields of specific nutrients. For the remainder of this manuscript, the former will be referred to as “edible conversion factors”, and the latter, “nutrient conversion factors”. The specific conversion factors will be referred to as  $E_{ED}$  (edible conversion factor) and  $E_{PRO}$ ,  $E_{EPA}$ ,  $E_{DHA}$ ,  $E_{IRON}$ ,  $E_{ZINC}$ ,  $E_{VitA}$ , and  $E_{VitB12}$  for protein, EPA, DHA, iron, zinc, vitamin A, and vitamin B<sub>12</sub> conversion factors, respectively. The values selected for use in this study are presented in Appendix B.

In the literature,  $E_{ED}$  estimates are characterized in a variety of ways. For finfish, common variants include fillet (both skin-on and skinless) weights as a percentage of gutted, headless weight and as a percentage of whole weight, and total edible meat as a percentage of both gutted, headless weight and whole weight (FAO, 1989). For this analysis, the  $E_{ED}$  value that was selected was for total edible biomass yield (i.e., yields of edible meat) as a percentage of whole body weight. This conversion factor was selected because it precludes the potential for the results to be influenced by changes in filleting technology (e.g., a shift from hand-filleting to machine filleting, which can result in slightly lower fillet yields; (FAO, n.d.), or consumer preference for fillets vs. whole fish. This decision accords with the intent of this analysis, which is to evaluate changes in nutrient yield that result from changes in the species composition of fisheries catches, rather than from fish processing or consumer decisions.

The primary source for  $E_{ED}$  values was the FAO document *Yield and Nutritional Value of the Commercially More Important Fish Species* (FAO, 1989). This document presents a series of monographs on the subject of edible biomass yield and protein and total fat content for a suite of high-profile species and species groups. To develop these monographs, the authors conducted an extensive review of the literature as it existed at the time. Each monograph presented estimates of edible biomass yield, in fillet and total flesh terms for finfish and in edible meats for shellfish. As this information was derived from a number of sources that used a variety of methods to analyze varying sample sizes from different regions, the authors also included their best estimate of an overall “selected value” for the species or species group. The authors of this document note several problems associated with their sources, and while the authors attempted to provide detail to support an informed understanding of the relative strengths and weaknesses of the data, they were limited in this effort by their source material. The values derived from this source are thus characterized by substantial uncertainty; uncertainty analyses for  $E_{ED}$  values are described in the Supplementary Information.

The other major source of information for conversion rates was the Food Composition Database maintained by the United States Department of Agriculture (USDA) (USDA, 2017). The data included in the USDA database are obtained from analyses conducted by the USDA's National Food and Nutrient Analysis Program. This database was the major source for  $E_{\text{PRO}}$ ,  $E_{\text{EPA}}$ ,  $E_{\text{DHA}}$ ,  $E_{\text{IRON}}$ ,  $E_{\text{ZINC}}$ ,  $E_{\text{VIT A}}$ , and  $E_{\text{VIT B12}}$  estimates.

These two sources did not present edible and nutrient conversion rates for all of the species included in this study. In such situations, information had to be found from additional sources, and/or estimates had to be obtained for similar species. Species-specific information for these situations is described in the Supplementary Information.

#### A.2.1. Gap-filling procedures for yield conversion factors

When taxon-specific estimates were not available, alternate sources and/or methods were used. In general, the order of preference by which yield estimates were obtained was as follows:

- a) Estimates for the same taxon for a similar region;
- b) Estimates for the same taxon from a different region;
- c) Estimates for closely related taxa in a similar region;
- d) Estimates for closely-related taxa in a different region.

For the purposes of this study, it was assumed that  $E_{\text{ED}}$  is largely a function of body type, and thus that in the absence of  $E_{\text{ED}}$  information for related taxa, information for morphologically similar (but taxonomically distant) taxa may serve as a proxy value. This approach was not followed for nutrient yield estimates, however.

#### *Alewife*

*Alewife* (*Alosa pseudoharengus*) is a member of family Clupeidae. For alewife, the sources did not provide information for  $E_{\text{ED}}$  or any nutrient yields. Thus, proxies had to be determined. For  $E_{\text{ED}}$ , the value that was used (0.63) was that of the most

prominent clupeid species in the catch data, Atlantic herring (*Clupea harengus*). For nutrient composition, the values that were used were those of a member of the same genus, American shad (*A. sapidissima*) (note: there were no estimates of edible biomass yield available for *A. sapidissima*; hence the use of the edible biomass yield value for *C. harengus*).

#### *American lobster*

For American lobster (*Homarus americanus*),  $E_{ED}$  varies with molt stage. Estimates vary and there is no single authoritative source, but generally soft-shell lobsters are said to yield 12.5% - 18%, whereas hard-shell lobsters yield 20% - 28%. For example, Dow and colleagues stated that hard-shell lobster edible biomass yield is approximately 25%, and soft-shell lobster yield is approximately 12.5% (Dow et al., 1975). For the purposes of this study, soft-shell lobsters are assumed to yield 15%, and hard-shell lobsters 25%.

Those lobsters caught in the winter and early spring (i.e., November through May) are mostly hard shell, while many caught in the summer and fall months are soft shell (Acheson, J., pers. comm.). In Maine, most lobsters are caught in the summer and fall months (Dow et al., 1975). In southwest Nova Scotia, the proportion of lobster landings that are soft-shelled seems to have grown in recent years. While historical data are not available, anecdotal evidence suggests that soft-shelled lobsters historically composed approximately 5-10% of landings in this region (Aquatic Science & Health Services, 2013). Since the beginning of the 21<sup>st</sup> century, however, this has increased to approximately 30-40%, with catches in the first several weeks of the season (i.e., late November and early December) particularly affected (Aquatic Science & Health Services, 2013).

Overall, there are no comprehensive data on the molt status of historical lobster catches in the USA and Canada lobster fisheries. For the purposes of this study, it was assumed that 70% and 30% of the USA's annual lobster catches were soft-shell

and hard-shell, respectively, and that 85% of Canada's lobster catches were hard-shell and 15% soft-shell.

As a result of these two assumptions (edible biomass yield of hard and soft-shell lobsters, and molt composition of USA and Canadian lobster catches), the  $E_{ED}$  values for the USA and Canadian lobster catches are 0.180 and 0.235, respectively. Any lobster catches outside of the USA and Canada were assumed to be 50% hard-shell and 50% soft-shell, and thus had an  $E_{ED}$  value of 0.200. Due to the unknown molt status of the lobster catches, and the associated uncertainty regarding their edible biomass yields, lobster  $E_{ED}$  values were assigned the highest uncertainty value (30%; see A.3. "Uncertainty in edible biomass yield rates").

#### *Atlantic butterfish*

For Atlantic butterfish (*Peprilus triacanthus*), there were no species-specific  $E_{ED}$  estimates available in the literature. The mean  $E_{ED}$  of two morphologically similar species (Atlantic herring and alewife) was used as a proxy.

For butterfish, estimates of  $E_{EPA}$  and  $E_{DHA}$  were not available, but estimates of total fat were available. To develop an estimate of EPA and DHA content of butterfish, the average EPA and DHA contents of all finfish in the study were determined. The ratios of these average EPA and DHA values to average total fat content were then determined. The average ratios of EPA and DHA to total fats were then multiplied by the species-specific total fat estimates for butterfish, and the results were used as this species' respective  $E_{EPA}$  and  $E_{DHA}$  values.

#### *Blue crab*

For blue crab (*Callinectes sapidus*), an estimate of  $E_{ED}$  (14%) was obtained from a brief summary written by an agent of the Florida Sea Grant College Program (Sweat, 2007).

### *Bluefish*

For bluefish (*Pomatomus saltatrix*), an  $E_{ED}$  estimate of 45% was obtained from a recent cookbook (Green, 2007).

### *Capelin*

Capelin (*Mallotus villosus*) are a member of the order Osmeriformes, family Osmeridae. The USDA nutrition data did not include information for capelin. Taxonomically, the most similar species in the USDA data were rainbow smelt (*Osmerus mordax*), also of the family Osmeridae. Thus, the nutrition parameters for rainbow smelt were used as a proxy for capelin.

### *Clams*

The  $E_{ED}$  estimate for the generic “clams” catch category was the mean  $E_{ED}$  value (0.1714) for the six taxa in the “Molluscs” commercial group in the SAU data.

### *Eelpouts*

In the SAU data, “eelpouts” is used as the term for catches of *Lycodes* spp., family Zoarcidae. The FAO 1989 data do not include any information on zoarcid fish. There is no information for  $E_{ED}$  for eelpouts (*Lycodes* spp.), or related taxa from other regions. The  $E_{ED}$  estimate for “hakes” is thus used for eelpouts, as hakes and eelpouts are elongated, somewhat eel-like groundfish. It should be noted that the  $E_{ED}$  value for hakes (53%) is similar to a published estimate of approximately 50% edible biomass yield from farm-raised spotted wolffish, *Anarhichas minor* (Green, 2007), which is another elongated, eel-like groundfish of somewhat similar morphology to eelpouts.

For the nutrition conversion factors, the only information in the USDA database for a zoarcid is for the ocean pout, *Zoarces americanus*, a pout that is found in the northwest Atlantic between the mid-Atlantic states and Labrador (Fishbase, 2018b). Ocean pouts are represented in the SAU catches, albeit as a relatively minor species when viewed over the entire 1950-2014 period. The USDA database’s

information for ocean pouts was used as values for the “eelpout” parameters  $E_{PRO}$ ,  $E_{IRON}$ ,  $E_{ZINC}$ ,  $E_{VitA}$ , and  $E_{VitB12}$ . However, there was no information in the USDA database for  $E_{EPA}$  or  $E_{DHA}$  for ocean pout. Thus, the means of all  $E_{EPA}$  and  $E_{DHA}$  values used in this study were used for eelpout  $E_{EPA}$  and  $E_{DHA}$  values.

### *Goosefishes*

There is scant information available for edible biomass yield and omega-3 fatty acid content of goosefish (Lophiidae), aka monkfish. The estimate of  $E_{ED}$  for goosefish (35%) was obtained from a recent cookbook (Green, 2007). Estimates of total EPA and DHA content (combined; 0.12% of total body weight) were obtained from (Strobel et al., 2012). The fraction of DHA to EPA in the tail was calculated to be 14:1 (Table 4 in (Prego et al., 2012)). Thus, the  $E_{DHA}$  and  $E_{EPA}$  conversion factors for goosefish edible mass was estimated to be:  $(14/15 \cdot 0.0012 = 0.00112)$  and  $(1/15 \cdot 0.0012 = 0.00008)$ , respectively; this assumes that DHA and EPA are regularly distributed throughout all of the fish’s mass. Values for goosefish  $E_{PRO}$ ,  $E_{IRON}$ ,  $E_{ZINC}$ ,  $E_{VitA}$ , and  $E_{VitB12}$  were obtained from the USDA database.

### *Grubbies/sculpins*

See “Scorpionfishes/flatheads”.

### *Hakes*

The three primary species of hake in the catch data were silver hake (*Merluccius bilinearis*), white hake (*Urophycis tenuis*), and red hake (*U. chuss*). Until the late 1970s, the generic species group “hakes” dominated hake catches, but this group’s catches declined to essentially zero by 1978. Of the hake catch identified to species during the 1950-1975 period, approximately 81% was silver hake, 10.3% white hake, and 8.9% red hake.

For all hakes, species-specific estimates of  $E_{ED}$  could not be determined. Thus, the FAO 1989 selected value for all *Merluccius* spp. (0.53) was used for the three

species in the catch data as well as the generic “hakes” category.

While the USDA nutrition database had information for “whiting” (likely *Merlangius merlangus*, in family Gadidae), it did not have any entries specifically for hakes in the gadoid families Merlucciidae (e.g., *M. bilinearis*) or Phycidae (e.g., *U. tenuis* or *U. chuss*). The closest shared taxonomic grouping for these three species is the order Gadiformes. The USDA nutrition database had information for four taxa in this order: Atlantic cod, haddock, saithe, and the aforementioned “whiting”. The three former species, along with hakes, are also grouped together under the “cod-likes” commercial group in the SAU reconstructed catch data. Thus, the decision was made to use the arithmetic mean of the USDA nutrient conversion factors for Atlantic cod, haddock, saithe, and “whiting” for the three hake species and the general “hake” category.

#### *Scorpionfishes/flatheads and Grubbies/sculpins*

Scorpionfishes/flatheads and grubbies/sculpins belong to the order Scorpaeniformes. The other members of the order Scorpaeniformes that are included in this study consist of the redfish (*Sebastes* spp.); lumpfish (*Cyclopterus lumpus*) are also a member of this order, but data for lumpfish were not considered to be sufficiently reliable to serve as proxies for other taxa.

Much data is missing for scorpionfishes/flatheads and grubbies/sculpins. Due to the taxonomic similarity of redfish to these groups, the  $E_{\text{PRO}}$ ,  $E_{\text{EPA}}$ ,  $E_{\text{DHA}}$ ,  $E_{\text{IRON}}$ ,  $E_{\text{ZINC}}$ ,  $E_{\text{VIT A}}$ , and  $E_{\text{VIT B12}}$  values for redfish were used as proxies. For grubbies and sculpins, the selected  $E_{\text{ED}}$  was the mean of the fillet and dressed head-off yields for several genera of Alaskan sculpins (State of Alaska, 2018). For scorpionfishes and flatheads, the mean of the  $E_{\text{ED}}$  values for redfish and the Alaskan sculpin species was used.

### *Shrimp*

There were no estimates for  $E_{VitA}$  or  $E_{VitB12}$  for raw shrimp in the USDA database, but there were estimates for raw shrimp of mixed species, to which moisture-retaining additives may or may not have been added. These were the values that were used for  $E_{VitA}$  and  $E_{VitB12}$ .

### *Snow crab*

For snow crab (*Chionoecetes opilio*), an  $E_{ED}$  estimate of 17% was obtained from an industry website (Pacific Seafood, 2017).

### *Wolffish*

There is no information for  $E_{ED}$  from wild-caught wolffish (*Anarhichas* spp.), or related taxa from other regions. The  $E_{ED}$  estimate for wolffish is taken from a recent cookbook (Green, 2007), which reported yields of approximately 50% for farm-raised spotted wolffish, *A. minor*.

## **A.3. Uncertainty**

Landings estimates, edible biomass yield rate estimates, and nutrient content estimates were all associated with uncertainty. In some cases, no published estimate of uncertainty was available, and proxies had to be developed as described below. Uncertainty also had to be propagated across estimates of landings, edible biomass yield rates, and nutrient contents to generate estimates of nutrient yield uncertainty specific to a given year, taxon, and nutrient. Finally, these year- and taxon-specific uncertainty estimates had to be combined to generate an overall estimate of uncertainty for each year  $j$ 's estimated total yield  $X$  of nutrient  $V$  across all taxa  $S$ . The methods and approaches used in these steps are described below.

### A.3.1. Uncertainty in catch estimates

Uncertainty values in the SAU database were derived from uncertainty scores assigned to a region's fishing sectors (e.g., industrial, artisanal) for the time periods 1950-1969, 1970-1989, and 1990-2010. The uncertainty scores, which ranged from

1 (lowest confidence) to 4 (highest confidence), were assigned by the catch reconstruction's authors based upon their judgment of the quality of the evidence for the reconstructed catch value, and of the agreement of that evidence. Catches assigned uncertainty scores of 1, 2, 3, or 4 then received "percent uncertainty" values of  $\pm 50\%$ ,  $\pm 30\%$ ,  $\pm 20\%$ , and  $\pm 10\%$ , respectively (Pauly & Zeller, 2016a). These percent uncertainty estimates were not presented with confidence levels (D. Palomares, pers. comm.). For the purposes of this study, it was assumed that the implicit confidence level used by the authors in assigning uncertainty scores was 95%, in keeping with custom.

In the SAU catch reconstruction database, estimates of uncertainty were available only for catches at the scale of nations' exclusive economic zones (EEZs). The catch data aggregated for NAFO, which comprises parts of the EEZs of Canada, the United States, Greenland, and France, as well as waters beyond national jurisdiction, thus did not come with ready-made uncertainty estimates. To estimate uncertainty for NAFO-scale catches, catch-weighted uncertainty scores were determined for each species' NAFO catches from their catches in EEZs off of the east coast of Canada and the United States. Any catches beyond those obtained from these two EEZs (i.e., NAFO catches that exceeded the sum of catches in the EEZs of Canada and the USA) were assumed to come from the high seas in NAFO and were assigned the highest uncertainty score (i.e., a score of 1 as defined in (Pauly & Zeller, 2016a)).

A section of Greenland's EEZ also contributes catches to NAFO. However, in the SAU data, catches that occurred in that portion of Greenland's EEZ that is within the NAFO "area of competence" (i.e., Greenland west coast) could not be differentiated from catches on Greenland's east coast, which is outside of NAFO. Thus, the SAU uncertainty scores assigned to catches taken from Greenland's EEZ were not used in the determination of NAFO-scale uncertainty scores. Effectively, this meant that any NAFO catches that originated from Greenland's western EEZ were lumped in with assumed "high seas" catches and were assigned an uncertainty score of 1.

While this likely overestimated the uncertainty for some of Greenland's catches, it should be noted that in the SAU data, significant proportions of catches in Greenland's EEZ were "inferred foreign catch", which did not have uncertainty scores. These catches would have been assigned an uncertainty score of 1 even if Greenland-specific uncertainty scores had been included in the calculation of NAFO-scale scores.

Finally, there is a small amount of France's EEZ present in NAFO waters, courtesy of the French-controlled islands of Saint Pierre and Miquelon. Catches from this EEZ are minimal relative to total NAFO catches; for example, the highest reported catches of any one taxon during the 1950-2014 period were the approximately 6.8 tons of Atlantic cod caught in 1987, and total catches of all taxa never exceeded 15 tons in any one year (Pauly & Zeller, 2018). Due to the relatively small catches in this EEZ, they were excluded from the catch-weighted uncertainty calculations.

Additional assumptions and decisions taken in the calculation of catch-weighted uncertainty scores for NAFO are as follows:

- Any EEZ-scale catches that lacked an uncertainty score (e.g., "inferred foreign catch") was assigned the highest uncertainty score (i.e., 1)
- Occasionally, the sum of Canada and USA catches exceeded the sum of NAFO catches. In these cases, it was assumed that the excess catches came from the southern portion of the USA's eastern EEZ, which extends farther south than the NAFO boundary. This assumption rests on the fact that those catches taken from the USA EEZ waters south of the NAFO boundary would be represented in the USA EEZ catch data, but not in the NAFO catch data. In these cases, the "excess" USA catches (i.e., the sum of Canada's and the USA's catches minus NAFO catches) were removed from the catch weighting calculations and the NAFO high seas catch was assumed to be zero.

### A.3.2. Uncertainty in edible biomass yield rates

The source for most of the  $E_{ED}$  values used in this study (FAO, 1989) generally did not present estimates of sampling variability alongside edible biomass yield estimates. For most taxa, only one or a small number of edible biomass yield estimates were presented, and often there was no clear taxon-specific estimate. Thus, it was not possible to generate taxon-specific estimates of uncertainty for edible biomass yield rates. Therefore, an alternative method had to be used. For consistency, the selected alternative was to employ a similar approach as the one used in the SAU catch data, via the assignment of uncertainty rates. As with the SAU catch data, an implicit 95% confidence level was assumed for these uncertainty rates.

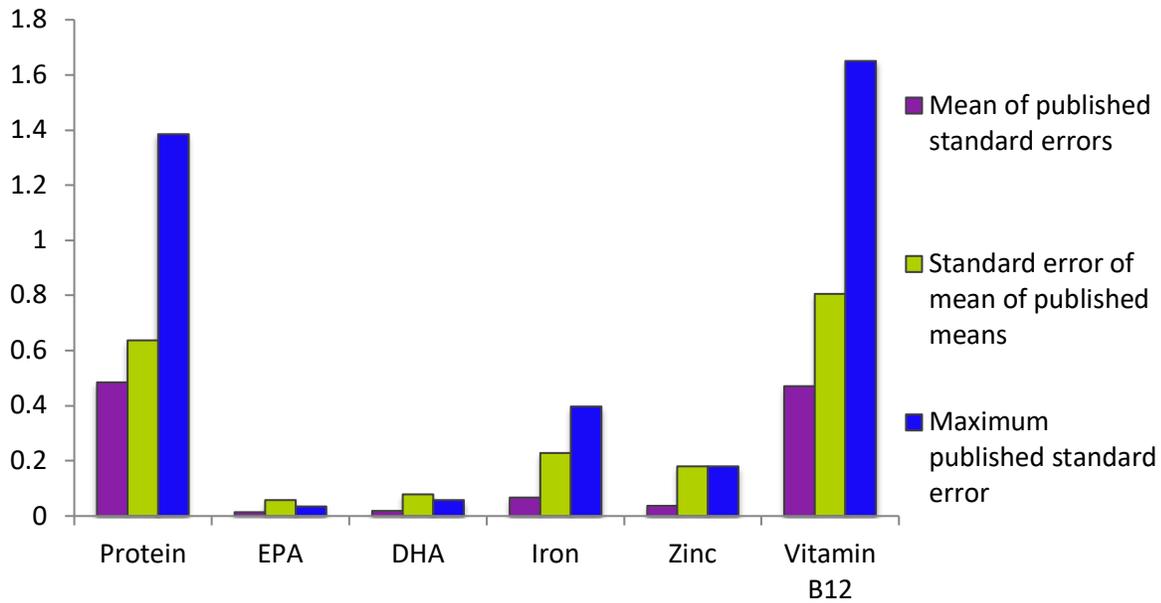
More specifically, the method used for  $E_{ED}$  uncertainty consisted of this study's lead author applying one of three different uncertainty rates, based upon the perceived quality and agreement of the available data for each taxon's edible biomass yield estimate (this approach was similar to the SAU database's use of four uncertainty levels and associated uncertainty rates). The three uncertainty rates were 10%, 20%, and 30%, with 95% confidence (i.e., that the true value is within 10% of the mean, 19 times out of 20). The 10% rate was reserved for several well-studied taxa, generally characterized by several estimates that converged to indicate <10% variability between estimates. The 20% uncertainty rate was generally used when one estimate was available for a taxon, but multiple estimates were not available. The 30% rate was used when there was no taxon-specific estimate, when there was a single taxon-specific estimate characterized by substantial uncertainty, and/or when a species' edible biomass yield is known to fluctuate substantially over the course of a year (e.g., lobsters).

### A.3.3. Uncertainty in nutrient conversion rates

Where the USDA database presented taxon-specific  $E_{PRO}$ ,  $E_{EPA}$ ,  $E_{DHA}$ ,  $E_{IRON}$ ,  $E_{ZINC}$ , and  $E_{VitB12}$  mean estimates along with a standard error of the mean, that standard error was used to indicate the uncertainty of that particular estimate.

Where a taxon-specific published standard error was not available, it could not be known how much estimates of nutrient  $V$  varied for that particular taxon. For  $E_{\text{PRO}}$ ,  $E_{\text{EPA}}$ ,  $E_{\text{DHA}}$ ,  $E_{\text{IRON}}$ ,  $E_{\text{ZINC}}$ , and/or  $E_{\text{VitB12}}$ , when no taxon-specific standard error was available, the mean standard error for the relevant nutrient over all taxa in the study was used as a proxy. We assumed that the variability in the taxon without a standard error estimate was similar to the variability in estimates for other taxa.

For  $E_{\text{VitA}}$ , there were no published standard errors for any taxa. In this case, the standard error of the mean  $E_{\text{VitA}}$  estimates across taxa was used as a proxy. The assumption here is that nutrient variation across taxa is greater than sample-to-sample variation within a taxon, such that a standard error of the mean of published means across taxa should provide a conservative proxy, bounded by natural variation of that nutrient across taxa, when taxon-specific standard error is not available. Support for this assumption is provided by the fact that, for  $E_{\text{PRO}}$ ,  $E_{\text{EPA}}$ ,  $E_{\text{DHA}}$ ,  $E_{\text{IRON}}$ ,  $E_{\text{ZINC}}$ , and  $E_{\text{VitB12}}$ , the standard error of the mean of published single-taxon means (i.e., the proposed proxy for vitamin A) was always somewhat greater than the mean of the published single-taxon standard errors (Figure A.2). This suggests that the use of the standard error of the mean of published means is conservative in that it may somewhat overestimate average standard error across all taxa (Figure A.2). While this proxy does not seem likely to underestimate average standard error across all taxa, it may not capture taxon-specific instances of high standard error (Figure A.2).



**Figure A.2.** Comparison of three indicators of error used for estimating uncertainty in nutrient conversion rates. The Y axis units are: grams (protein, EPA, and DHA), milligrams (iron and zinc), and micrograms (vitamin B12). The average of the published single-taxon standard errors used in this study is shown as the blue column for each nutrient; this was used as the proxy value when a taxon-specific value was not available for EPRO, EEPA, EDHA, EIRON, EZINC, or EVitB12. For EVitA, no published estimates of error were available for any taxon; the proxy that was used was the standard error of the mean of the published single-taxon means for EVitA, indicated in red. For reference, maximum published single-taxon standard errors for each nutrient are represented in green.

#### A.3.4. Propagating uncertainty

To arrive at estimates of edible nutrient yield derived from catches of a given taxon involves the multiplication of a catch estimate with an edible biomass yield estimate and estimates of the nutrient content of the edible portion. As discussed in this section, each of these estimates is associated with a degree of uncertainty. The estimate of edible nutrient yield is thus informed by uncertainty at all three steps. Equation 17 shows how uncertainty is typically propagated for  $f = x*y$ , where  $\sigma$  is standard deviation (Lyons, 1991).

$$\left(\frac{\sigma_f}{f}\right)^2 = \left(\frac{\sigma_x}{x}\right)^2 + \left(\frac{\sigma_y}{y}\right)^2 \quad \text{Equation 17}$$

In Equation 17, the uncertainties to be combined (and the result that they produce) are all based on the same confidence level, because one  $\sigma$  for normally distributed data corresponds to a confidence level of approximately 68.3%. Therefore, to combine uncertainty estimates as shown in Equation 17, each estimate must represent equivalent confidence levels. As described previously, 95% confidence levels were assumed for the landings uncertainty estimates and the  $E_{ED}$  rate uncertainty estimates, allowing Equation 17 to be used to propagate uncertainty across estimates of landings,  $E_{ED}$ , and nutrient content.

Thus, the following process was followed to propagate uncertainty:

1. Assuming a normal distribution of sample means around the (unknown) population mean, the upper and lower 95% confidence intervals for the population mean are found by adding and subtracting 1.96 x the standard error of the mean (SEM), respectively, from the sample mean (Altman & Bland, 2005). Thus, to calculate a “proxy standard error of the mean” (PSEM) for each catch estimate and  $E_{ED}$  estimate, the percent uncertainties were first multiplied by the associated catch or  $E_{ED}$  estimate (to give a “proxy 95% confidence interval” for that estimate), and the result was divided by 1.96 to give a PSEM.
2. The resulting PSEM values for the SAU catch data and taxon-specific  $E_{ED}$  estimates could then be combined with SEM estimates for taxon-specific nutrient content rate estimates. Equation 17 was thus used as the basis for propagating uncertainty across taxon  $i$ -specific estimates of catch  $C$ , edible biomass yield  $E_{ED}$ , and nutrient content  $E_V$ . Essentially, this involved the combination of the PSEM for catches and  $E_{ED}$  estimates with the SEM for  $E_V$  to yield a propagated PSEM for yield  $x$  of nutrient  $V$  from taxon  $i$  in year  $j$  (Equation 18):

$$\frac{PSEM_{x_{V,i,j}}}{x_{V,i,j}} = \sqrt{\left(\frac{PSEM_{C,i,j}}{C_{i,j}}\right)^2 + \left(\frac{PSEM_{E_{ED,i}}}{E_{ED,i}}\right)^2 + \left(\frac{SEM_{E_{V,i}}}{E_{V,i}}\right)^2} \quad \text{Equation 18}$$

To calculate the associated proxy 95% confidence interval (CI) from these PSEM values, Equation 19 was followed.

$$\text{Proxy 95\% } CI_{x_{V,i,j}} = \left(\frac{PSEM_{x_{V,i,j}}}{x_{V,i,j}}\right) * x_{V,i,j} * 1.96 \quad \text{Equation 19}$$

3. At this stage, there are  $S$  taxon-specific estimates of yield  $x$  for nutrient  $V$  in year  $j$ , each with an associated proxy 95% confidence interval. The errors for each taxon-specific estimate were not assumed to be independent of one another, because the SAU catch reconstructions necessarily employed a variety of assumptions that affected catch estimates across taxa. Thus, for a given year, it is possible that catch errors for any number of taxa would display some degree of covariance. As it was not possible to estimate this covariance, the most conservative approach to propagating uncertainty across taxon-specific estimates of nutrient yield was employed: the proxy 95% confidence interval for total yield  $X$  of nutrient  $V$  across all taxa  $S$  in year  $j$  was simply the sum of all taxon-specific proxy 95% confidence intervals (Equation 20):

$$\text{Proxy 95\% } CI_{X_{V,S,j}} = \sum_{n=1}^S \text{Proxy 95\% } CI_{x_{V,i,j}} \quad \text{Equation 20}$$

#### **A.4. Recommended Daily Amount estimates**

In order to better understand the human nutritional importance of the nutrient yields of the studied fisheries, as well as the consequences of changes in these yields, the nutrient yields were re-expressed in terms of the number of adult humans who could be supported for a full year at their Recommended Daily Allowance (RDA) of the nutrient in question. The RDAs used in these calculations are shown in Appendix C.

## A.5. Simpson's Evenness

Theoretically, the degree to which a fishing region's yield of nutrient  $V$  is distributed across taxa may indicate greater or lesser resilience of  $V$  production to unforeseen changes, due to a greater or lesser diversity of potential responses to unforeseen change (Elmqvist et al., 2003). A region that is heavily dependent on one species to produce  $V$  would thus be considered to be less resilient than one in which  $V$ 's yields are broadly distributed across one dozen species, for example. Thus, it is proposed that a simple index for nutrient distribution across taxa may be a useful indicator for changes in resilience. The index selected for use in this study is Simpson's evenness ( $E$ ).

In ecology, Simpson's  $E$  is a commonly used indicator of species evenness, which along with species richness is one of the basic elements of species diversity. Species evenness is the distribution of a community's total number of individuals across species, such that high evenness reflects a more equal distribution of individuals across species and low evenness indicates clustering of individuals within fewer species<sup>14</sup>. Here, Simpson's  $E$  is used to indicate the distribution of nutrient  $V$  yield across the taxa in the study. As a preliminary step, Simpson's dominance ( $D$ ) was first calculated as shown in Equation 21 (Morris et al., 2014):

$$D_{X_{V,S,j}} = \frac{1}{\sum_{n=1}^S \left( \frac{x_{V,i,j}}{X_{V,S,j}} \right)^2} \quad \text{Equation 21}$$

Where  $X_{V,S,j}$  is the total yield  $X$  of  $V$  from all taxa  $S$  in year  $j$ , and  $x_{v,i,j}$  is the yield that was derived from catches of taxon  $i$ . Simpson's  $E$  is then obtained as shown in Equation 22 (Morris et al., 2014):

$$E_{X_{V,S,j}} = \frac{D_{X_{V,S,j}}}{S} \quad \text{Equation 22}$$

## A.6. Mean Trophic Level

The mean trophic level (MTL) of a region's catches is the sum of that region's taxon-specific trophic levels weighted for each taxon's contribution to total catches (Pauly & Palomares, 2005). For this study, the MTL for yield  $X$  of nutrient  $V$  across all taxa  $S$  in year  $j$  was the sum of taxon-specific trophic levels ( $TL_i$ ) weighted for each taxon  $i$ 's contribution  $x$  to yields of the nutrient, as shown in Equation 23:

$$MTL_{X_{V,S,j}} = \sum_{n=1}^S \left( \frac{x_{V,i,j}}{X_{V,S,j}} * TL_i \right) \quad \text{Equation 23}$$

In addition to MTL, the distribution of nutrient  $V$  across trophic levels in year  $j$  was also determined (Figure 2.5). This was found by summing the yield  $X$  of  $V$  at each 0.1 trophic level increment. For both the distribution of nutrients across taxa and MTL calculations, species-specific trophic levels were obtained from Fishbase (Fishbase, 2018a) and SeaLifeBase (SeaLifeBase, 2018).

## A.7. $B_{80}$ : an index nutrient yield concentration in landed biomass

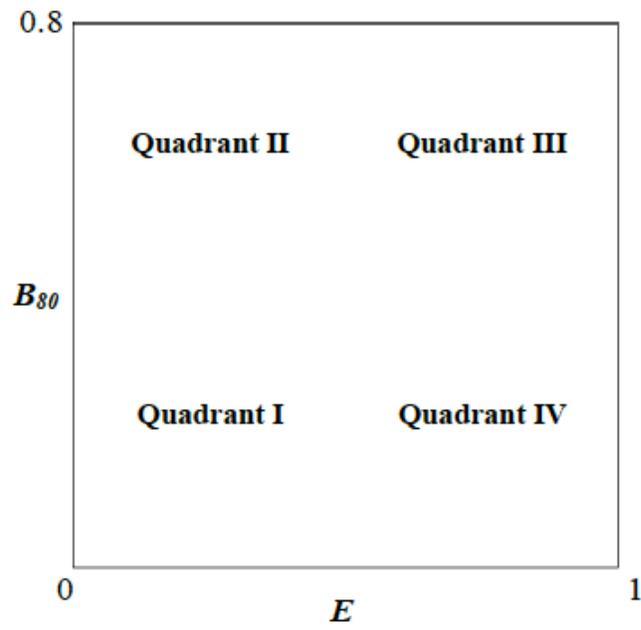
To support the assessment and monitoring of nutrient concentrations in landed biomass, we developed a simple index,  $B_{80}$ . To determine this index for nutrient  $V$ , the intrinsic nutrient content,  $C$ , is first determined for each taxon  $i$ , from taxon-specific estimates of edible biomass yield rate ( $E_{ED}$ ) and nutrient  $V$  content rate ( $E_V$ ) (Equation 24). The nutrient yields and landings weights of all taxa are then put in order of descending  $C_{V,i}$  values, and are summed starting with the taxon that has the highest  $C_{V,i}$  value and proceeding toward that with the lowest. When the sum of this sequentially increasing nutrient yield equals 80% of total nutrient yields, the corresponding landings weight is the  $B_{Y80}$  value. The  $B_{80}$  index is the ratio of  $B_{Y80}$  to total landings biomass ( $B_{TOT}$ ) (Equation 25):

$$C_{V,i} = E_{ED,i} * E_V \quad \text{Equation 24}$$

$$B_{80} = \frac{BY_{80}}{B_{TOT}} \quad \text{Equation 25}$$

By viewing  $B_{80}$  relative to  $E$  (Figure A.3), important summary characteristics of the fisheries' nutrient yields can be quickly ascertained. A position in quadrant I shows that the nutrient's yields are dominated by few taxa (low  $E$ ) and are highly concentrated in relatively little biomass (low  $B_{80}$ ). Quadrant II indicates that yields are dominated by few taxa (low  $E$ ) and that nutrient yields are distributed more evenly across total biomass (high  $B_{80}$ ); positioning in this quadrant may occur when a nutrient is dependent upon a small number of taxa that dominate catch biomass. Positioning in quadrant III occurs when yields are relatively evenly distributed across many taxa (high  $E$ ) and across landings biomass (high  $B_{80}$ ); this scenario may result when nutrient concentrations are relatively steady across many taxa (e.g., protein). Finally, a position in quadrant IV indicates that yields are distributed across many taxa (high  $E$ ), but that nutrient yields are highly concentrated in relatively little biomass (low  $B_{80}$ ). A position in quadrant IV may occur when the nutrient is primarily obtained from multiple species that are minor from a catch weight perspective.

Of these four quadrants, three (I, II, and IV) indicate the best options for reducing total catch weight while minimizing effects on yields of the single nutrient in question. In quadrant I and potentially in quadrant IV, catches can be reduced for those taxa that dominate total catch biomass but contribute relatively little to nutrient yields. For quadrant II, it may be possible to reduce catches of many taxa that are minor for both catch weights and nutrient yield. Conversely, quadrant III represents a scenario in which yields of the nutrient will likely be quickly affected by attempts to reduce catches. Consideration of consequences for the resilience of nutrient yields (e.g., through modeling changes to  $E$ ) should inform any effort to strategically reduce catches while minimizing effects on nutrient yields.



**Figure A.3.** Quadrants of the  $B_{80}/E$  plot.

## Appendix B. Species groups and nutrient conversion rates selected for inclusion in the study.

Note that the names of the species/species groups come from the Sea Around Us database, with the exception of the USA/Canada/other nations differentiation of American lobster. The E<sub>ED</sub> conversion rate converts the weight of the catch to the weight of edible biomass obtained from the catch. All other conversion rates are relative to the edible amount and convert on a weight:weight basis (i.e., convert the weight of the edible amount to the weight of the nutrient embodied in that edible amount, in the same units used to quantify the catch).

Taxon		Nutrient Conversion Rates							
Common name	Scientific name	E <sub>ED</sub>	E <sub>PRO</sub>	E <sub>EPA</sub>	E <sub>DHA</sub>	E <sub>IRON</sub>	E <sub>ZINC</sub>	E <sub>VITA</sub>	E <sub>VIT B12</sub>
Alewife	<i>Alosa pseudoharengus</i>	0.63	0.1693	1.09E-02	1.32E-02	9.70E-06	3.70E-06	3.20E-07	1.50E-09
American cupped oyster	<i>Crassostrea virginica</i>	0.105	0.0571	1.77E-03	1.36E-03	4.61E-05	3.93E-04	1.30E-07	8.75E-08
American lobster (USA)	<i>Homarus americanus</i>	0.18	0.1652	1.02E-03	6.80E-04	2.60E-06	3.53E-05	1.00E-08	1.25E-08
American lobster (Canada)	<i>Homarus americanus</i>	0.235	0.1652	1.02E-03	6.80E-04	2.60E-06	3.53E-05	1.00E-08	1.25E-08
American lobster (Other nations)	<i>Homarus americanus</i>	0.2	0.1652	1.02E-03	6.80E-04	2.60E-06	3.53E-05	1.00E-08	1.25E-08
American plaice	<i>Hippoglossoides platessoides</i>	0.49	0.1241	1.37E-03	1.08E-03	1.80E-06	3.20E-06	1.00E-07	1.13E-08
American sea scallop	<i>Placopecten magellanicus</i>	0.079	0.1206	4.20E-04	6.10E-04	3.80E-06	9.10E-06	1.00E-08	1.41E-08
Atlantic butterfish	<i>Peprilus triacanthus</i>	0.63	0.1728	7.35E-03	1.02E-02	5.00E-06	7.70E-06	3.00E-07	1.90E-08
Atlantic cod	<i>Gadus morhua</i>	0.47	0.1781	6.40E-04	1.20E-03	3.80E-06	4.50E-06	1.20E-07	9.10E-09
Atlantic herring	<i>Clupea harengus</i>	0.63	0.1796	7.09E-03	8.62E-03	1.10E-05	9.90E-06	2.80E-07	1.37E-07
Atlantic mackerel	<i>Scomber scombrus</i>	0.535	0.186	8.98E-03	1.40E-02	1.63E-05	6.30E-06	5.00E-07	8.71E-08

Taxon		Nutrient Conversion Rates							
Common name	Scientific name	E <sub>ED</sub>	E <sub>PRO</sub>	E <sub>EPA</sub>	E <sub>DHA</sub>	E <sub>IRON</sub>	E <sub>ZINC</sub>	E <sub>VITA</sub>	E <sub>VIT B12</sub>
Atlantic surf clam	<i>Spisula solidissima</i>	0.3	0.1467	4.30E-04	6.40E-04	1.62E-05	5.10E-06	9.00E-07	1.13E-07
Blue crab	<i>Callinectes sapidus</i>	0.14	0.1806	1.70E-03	1.50E-03	7.40E-06	3.54E-05	2.00E-08	9.00E-08
Blue mussel	<i>Mytilus edulis</i>	0.24	0.119	1.88E-03	2.53E-03	3.95E-05	1.60E-05	4.80E-07	1.20E-07
Bluefish	<i>Pomatomus saltatrix</i>	0.45	0.2004	2.52E-03	5.19E-03	4.80E-06	8.10E-06	1.20E-06	5.39E-08
Capelin	<i>Mallotus villosus</i>	0.6	0.1763	2.75E-03	4.18E-03	9.00E-06	1.65E-05	1.50E-07	3.44E-08
Clams	<i>Bivalvia</i>	0.171	0.1467	4.30E-04	6.40E-04	1.62E-05	5.10E-06	9.00E-07	1.13E-07
Dogfishes	<i>Squalus</i>	0.46	0.2098	3.16E-03	5.27E-03	8.40E-06	4.30E-06	7.00E-07	1.49E-08
Eelpouts	<i>Lycodes</i>	0.53	0.1664	2.41E-03	3.33E-03	2.80E-06	1.03E-05	1.20E-07	9.00E-09
Goosefishes	<i>Lophiidae</i>	0.35	0.1448	8.00E-05	1.12E-03	3.20E-06	4.10E-06	1.20E-07	9.00E-09
Greenland halibut	<i>Reinhardtius hippoglossoides</i>	0.49	0.1856	6.60E-04	1.28E-03	1.60E-06	3.60E-06	2.00E-07	1.10E-08
Grubbies and sculpins	<i>Myoxocephalus</i>	0.315	0.1531	6.20E-04	1.53E-03	2.20E-06	2.90E-06	1.20E-07	1.50E-08
Haddock	<i>Melanogrammus aeglefinus</i>	0.48	0.1632	4.20E-04	8.90E-04	1.70E-06	3.20E-06	1.70E-07	1.83E-08
Hakes	<i>Merluccius</i>	0.53	0.1797	6.75E-04	1.73E-03	3.38E-06	5.30E-06	1.83E-07	2.06E-08
Longfin inshore squid	<i>Doryteuthis pealeii</i>	0.67	0.1558	1.46E-03	3.42E-03	6.80E-06	1.53E-05	1.00E-07	1.30E-08
Northern prawn	<i>Pandalus borealis</i>	0.362	0.201	3.00E-04	3.10E-04	5.20E-06	1.34E-05	5.40E-07	1.11E-08
Northern quahog	<i>Mercenaria mercenaria</i>	0.101	0.1467	4.30E-04	6.40E-04	1.62E-05	5.10E-06	9.00E-07	1.13E-07
Northern shortfin squid	<i>Illex illecebrosus</i>	0.635	0.1558	1.46E-03	3.42E-03	6.80E-06	1.53E-05	1.00E-07	1.30E-08
Ocean quahog	<i>Arctica islandica</i>	0.2	0.1467	4.30E-04	6.40E-04	1.62E-05	5.10E-06	9.00E-07	1.13E-07

Taxon		Nutrient Conversion Rates							
Common name	Scientific name	E <sub>ED</sub>	E <sub>PRO</sub>	E <sub>EPA</sub>	E <sub>DHA</sub>	E <sub>IRON</sub>	E <sub>ZINC</sub>	E <sub>VITA</sub>	E <sub>VIT B12</sub>
Piked dogfish	<i>Squalus acanthias</i>	0.46	0.2098	3.16E-03	5.27E-03	8.40E-06	4.30E-06	7.00E-07	1.49E-08
Red hake	<i>Urophycis chuss</i>	0.53	0.1797	6.75E-04	1.73E-03	3.38E-06	5.30E-06	1.83E-07	2.06E-08
Redfishes	<i>Sebastes</i>	0.479	0.1531	6.20E-04	1.53E-03	2.20E-06	2.90E-06	1.20E-07	1.50E-08
Saithe	<i>Pollachius virens</i>	0.47	0.1944	7.10E-04	3.50E-03	4.60E-06	4.70E-06	1.40E-07	3.19E-08
Scorpionfishes, flatheads	<i>Scorpaeniformes</i>	0.397	0.1531	6.20E-04	1.53E-03	2.20E-06	2.90E-06	1.20E-07	1.50E-08
Silver hake	<i>Merluccius bilinearis</i>	0.53	0.1797	6.75E-04	1.73E-03	3.38E-06	5.30E-06	1.83E-07	2.06E-08
Snow crab	<i>Chionoecetes opilio</i>	0.17	0.185	2.59E-03	1.13E-03	2.50E-05	2.80E-05	4.50E-07	9.00E-08
Squids	<i>Teuthida</i>	0.67	0.1558	1.46E-03	3.42E-03	6.80E-06	1.53E-05	1.00E-07	1.30E-08
Summer flounder	<i>Paralichthys dentatus</i>	0.49	0.1241	1.37E-03	1.08E-03	1.80E-06	3.20E-06	1.00E-07	1.13E-08
White hake	<i>Urophycis tenuis</i>	0.53	0.1797	6.75E-04	1.73E-03	3.38E-06	5.30E-06	1.83E-07	2.06E-08
Winter flounder	<i>Pseudopleuronectes americanus</i>	0.49	0.1241	1.37E-03	1.08E-03	1.80E-06	3.20E-06	1.00E-07	1.13E-08
Witch flounder	<i>Glyptocephalus cynoglossus</i>	0.49	0.1241	1.37E-03	1.08E-03	1.80E-06	3.20E-06	1.00E-07	1.13E-08
Wolffishes	<i>Anarhichas</i>	0.5	0.175	3.07E-03	3.16E-03	9.00E-07	7.80E-06	1.13E-06	2.03E-08
Yellowtail flounder	<i>Limanda ferruginea</i>	0.5	0.1241	1.37E-03	1.08E-03	1.80E-06	3.20E-06	1.00E-07	1.13E-08

**Appendix C. Recommended Daily Allowance values (adults), or similar values, used in this study.**

<b>Nutrient</b>	<b>Recommended Dietary Allowance (RDA)</b>	<b>Value selected for use as RDA</b>	<b>Source</b>
Protein	0.80 g protein per kg body weight (males and females)	51 g (assuming 50% male, 50% female, and .80 g/kg body weight, using reference body weights)	(Food and Nutrition Board, 2011b)
Omega 3 fatty acids <sup>1</sup>	0.250 g <sup>1</sup>	0.250 g <sup>1</sup>	(FAO/WHO, 2008)
Iron	8 mg (male and post-menopausal female)	10.5 mg (assuming 50% male, 25% post-menopausal female, 25% pre-menopausal female)	(Food and Nutrition Board, 2011a)
	18 mg (pre-menopausal female)		
Zinc	11 mg (male)	9.5 mg (assuming 50% male and 50% female)	(Food and Nutrition Board, 2011a)
	8 mg (female)		
Vitamin A	900 µg RAE (male)	800 µg RAE (assuming 50% male and 50% female)	(Food and Nutrition Board, 2001)
	700 µg RAE (female)		
Vitamin B12	2.4 µg	2.4 µg	(Food and Nutrition Board, 2011c)

1 = In the United States, there is no established RDA value for EPA or DHA; the selected value is the value recommended by a joint United Nations Food and Agriculture Organization/World Health Organization working group.

**Appendix D. Uncertainty estimates for American lobster and Atlantic herring.**

SEM = standard error of the mean. PSEM = “proxy” standard error of the mean, determined when SEM was not available (see Chapter 2 for details).

	<i>Edible biomass yield (PSEM)</i>	<i>Protein (SEM)</i>	<i>EPA (SEM)</i>	<i>DHA (SEM)</i>	<i>Iron (SEM)</i>	<i>Zinc (SEM)</i>	<i>Vitamin A (PSEM)</i>	<i>Vitamin B<sub>12</sub> (SEM)</i>
<i>American lobster (70/30 soft/hard shell)</i>	0.0276	0.0049	1.33 e-4	1.86 e-4	6.83 e-7	3.79 e-7	6.453 e-8	4.708 e-9
<i>Atlantic herring</i>	0.0643	0.0024	2.8- e-4	5.7 e-4	6.83 e-7	7.3 e-7	6.453 e-8	5.58 e-9

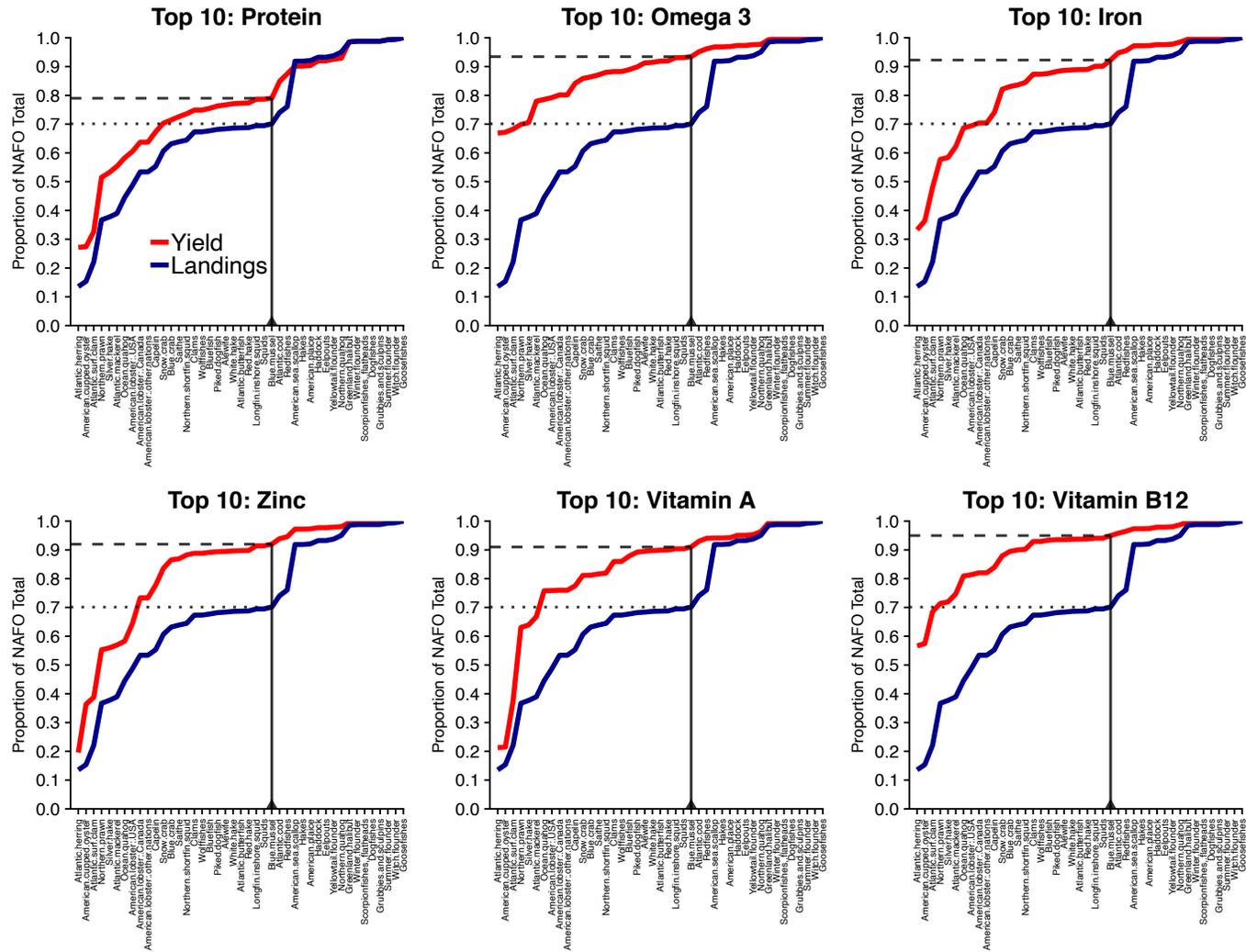
**Appendix E. Each taxon's mean *R* value for each nutrient, and mean *R* value across nutrients, for the years 2010-2014.**

	<b>Protein</b>	<b>Omega-3</b>	<b>Iron</b>	<b>Zinc</b>	<b>Vitamin A</b>	<b>Vitamin B<sub>12</sub></b>	<b>Mean</b>
<b>Atlantic herring</b>	2.01	4.94	2.46	1.45	1.57	4.18	<b>2.77</b>
<b>Atlantic mackerel</b>	1.77	6.14	3.09	0.78	2.38	2.26	<b>2.74</b>
<b>Alewife</b>	1.89	7.57	2.17	0.54	1.79	0.05	<b>2.33</b>
<b>American cupped oyster</b>	0.11	0.16	1.72	9.58	0.12	0.45	<b>2.02</b>
<b>Atlantic butterfish</b>	1.93	5.51	1.12	1.13	1.68	0.58	<b>1.99</b>
<b>Bluefish</b>	1.60	1.73	0.77	0.85	4.80	1.18	<b>1.82</b>
<b>Capelin</b>	1.88	2.08	1.92	2.30	0.80	1.00	<b>1.66</b>
<b>Wolffishes</b>	1.55	1.55	0.16	0.91	5.03	0.49	<b>1.62</b>
<b>Piked dogfish</b>	1.71	1.94	1.37	0.46	2.86	0.33	<b>1.45</b>
<b>Longfin inshore squid</b>	1.85	1.63	1.62	2.38	0.60	0.42	<b>1.42</b>
<b>Squids</b>	1.85	1.63	1.62	2.38	0.60	0.42	<b>1.42</b>
<b>Northern shortfin squid</b>	1.75	1.55	1.53	2.25	0.56	0.40	<b>1.34</b>
<b>Blue mussel</b>	0.51	0.53	3.36	0.89	1.02	1.40	<b>1.29</b>
<b>Atlantic surf clam</b>	0.78	0.16	1.72	0.36	2.40	1.64	<b>1.18</b>
<b>Saithe</b>	1.62	0.99	0.77	0.51	0.59	0.73	<b>0.87</b>
<b>Northern prawn</b>	1.29	0.11	0.67	1.13	1.74	0.20	<b>0.85</b>
<b>White hake</b>	1.69	0.63	0.63	0.65	0.86	0.53	<b>0.83</b>
<b>Silver hake</b>	1.69	0.63	0.63	0.65	0.86	0.53	<b>0.83</b>
<b>Red hake</b>	1.69	0.63	0.63	0.65	0.86	0.53	<b>0.83</b>
<b>Snow crab</b>	0.56	0.32	1.51	1.10	0.68	0.74	<b>0.82</b>
<b>Ocean quahog</b>	0.53	0.11	1.17	0.24	1.63	1.11	<b>0.80</b>

	<b>Protein</b>	<b>Omega-3</b>	<b>Iron</b>	<b>Zinc</b>	<b>Vitamin A</b>	<b>Vitamin B<sub>12</sub></b>	<b>Mean</b>
<b>Clams</b>	0.45	0.09	0.98	0.20	1.37	0.94	<b>0.67</b>
<b>Hakes</b>	1.34	0.50	0.50	0.52	0.69	0.42	<b>0.66</b>
<b>Greenland halibut</b>	1.61	0.47	0.28	0.41	0.87	0.26	<b>0.65</b>
<b>Atlantic cod</b>	1.48	0.43	0.63	0.49	0.50	0.21	<b>0.63</b>
<b>Haddock</b>	1.39	0.31	0.29	0.36	0.73	0.43	<b>0.58</b>
<b>Redfishes</b>	1.30	0.51	0.37	0.32	0.51	0.35	<b>0.56</b>
<b>American lobster, Canada</b>	0.69	0.20	0.22	1.93	0.02	0.14	<b>0.53</b>
<b>Yellowtail flounder</b>	1.10	0.61	0.32	0.37	0.44	0.27	<b>0.52</b>
<b>Winter flounder</b>	1.08	0.60	0.31	0.36	0.44	0.27	<b>0.51</b>
<b>American plaice</b>	1.08	0.60	0.31	0.36	0.44	0.27	<b>0.51</b>
<b>Summer flounder</b>	1.08	0.60	0.31	0.36	0.44	0.27	<b>0.51</b>
<b>Witch flounder</b>	1.08	0.60	0.31	0.36	0.44	0.27	<b>0.51</b>
<b>Blue crab</b>	0.45	0.22	0.37	1.15	0.02	0.61	<b>0.47</b>
<b>Scorpionfishes, flatheads</b>	1.08	0.43	0.31	0.27	0.42	0.29	<b>0.47</b>
<b>American lobster, other nations</b>	0.59	0.17	0.18	1.64	0.02	0.12	<b>0.45</b>
<b>American lobster, USA</b>	0.53	0.15	0.17	1.47	0.02	0.11	<b>0.41</b>
<b>Northern, quahog</b>	0.26	0.05	0.58	0.12	0.81	0.55	<b>0.40</b>
<b>Goosefishes</b>	0.90	0.21	0.40	0.33	0.37	0.15	<b>0.39</b>
<b>Grubbies, sculpins</b>	0.69	0.27	0.20	0.17	0.27	0.18	<b>0.30</b>
<b>American sea scallop</b>	0.17	0.04	0.11	0.17	0.01	0.05	<b>0.09</b>
<b>Eelpouts</b>	0.00	0.00	0.00	0.00	0.00	0.00	<b>0.00</b>
<b>Dogfishes</b>	0.00	0.00	0.00	0.00	0.00	0.00	<b>0.00</b>

## **Appendix F. Optimization figures for different scenarios**

Figure F.1. Yields, landings, and sets of taxa, top 10 taxa for all nutrients



**Figure F.2.** Yields, landings, and sets of taxa, all taxa with  $R \geq 1.0$  for  $\geq 3$  nutrients

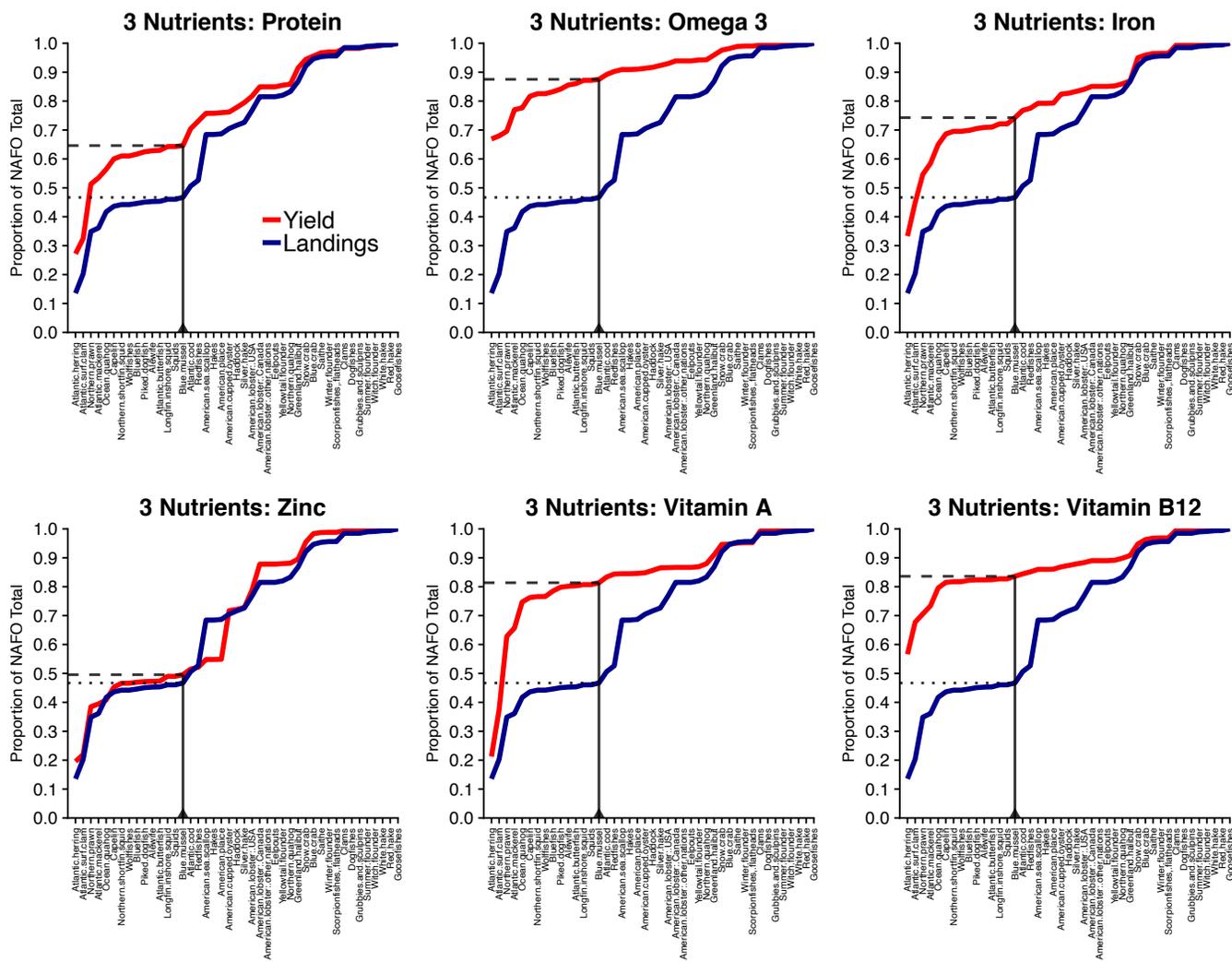
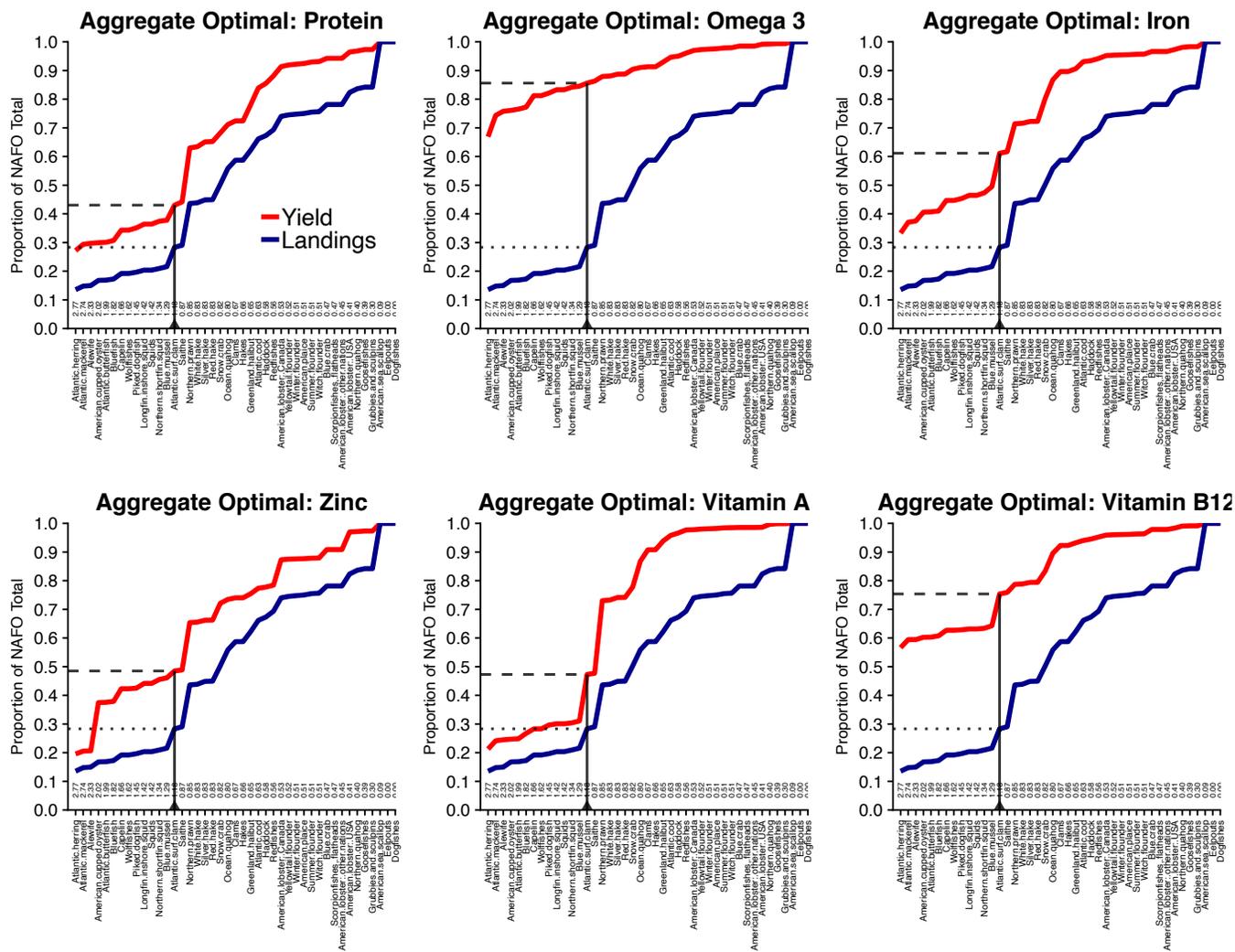


Figure F.3. Yields, landings, and sets of taxa, aggregate  $R$  approach.



**Figure F.4.** Yields, landings, and sets of taxa, iron optimal set + zinc optimal set

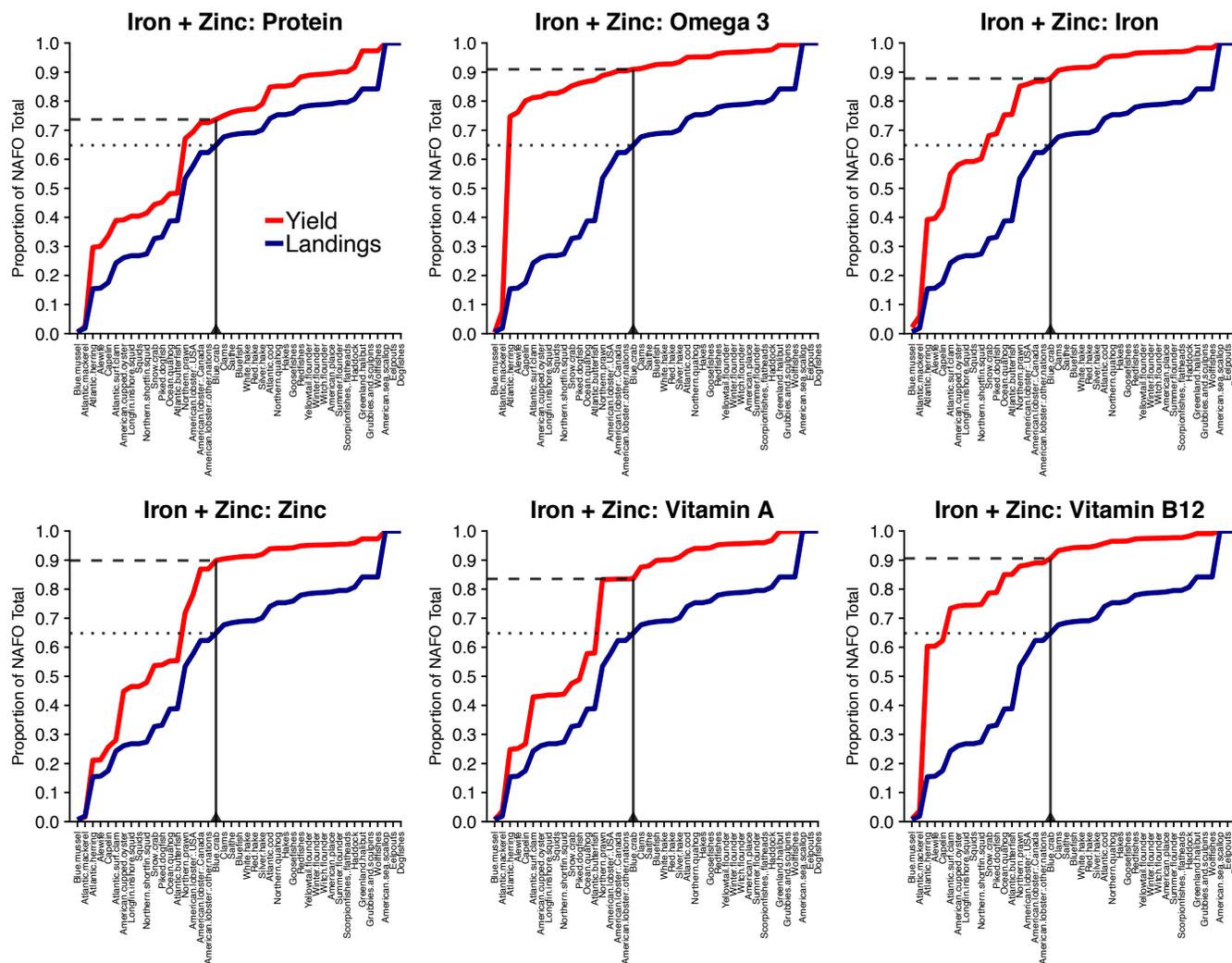


Figure F.5. Yields, landings, and sets of taxa, weighted aggregate  $R$  approach: (iron \* 0.5) + (zinc \* 0.5)

