Aligning Management and Reproductive Strategies in Modern Fisheries Management

By

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ABSTRACT

Numerous approaches to fisheries management exist, and it is paramount that the right management strategy is implemented for a given resource. One tool available in the decision-making process is a management strategy evaluation, which aims to simulate the effects of varying management strategies and harvest control rules on a stock. Management strategies are not one-size-fits-all, and the selected strategy and associated harvest control rules should align with the species' life history, including reproductive output, as well as resource use objectives. The following report provides a review of management strategy implementation in Canada (Chapter 2), fish reproductive strategies (Chapter 3), and matrix projection models for their use in management strategy evaluation (Chapter 4). Chapter 5 includes original research and details the outcome of a management strategy evaluation of the precautionary approach, co-management, and ecosystem-based fisheries management strategies on hypothetical resources with varying life history traits, ranging from extreme *r*-selected to extreme *K*-selected. The management strategy evaluation was completed for each resource and strategy under three distinct scenarios based on stock health, economic pressure, and ecosystem health. The overall finding was that the precautionary approach, when implemented correctly, was the most balanced strategy for meeting the objectives of each scenario. The most influential factor affecting simulation outcomes was economic pressure, with co-management susceptible to both under- and over-fishing at the extremes of low and high economic pressure to fish. The report continues with a comparison of the original research results with real-world fisheries (Chapter 6) and concludes with a summary chapter (Chapter 7). Fisheries management is complex, balancing social, economic, and ecosystem objectives, but a healthy resource ultimately underpins any successful management system. A fisheries management strategy that aligns with the reproductive output of a resource stands a chance at meeting these objectives and results in a sustainable, healthy fishery.

GENERAL SUMMARY

Management strategy evaluations are tools used in fisheries management to examine outcomes of various harvest control rules through closed-loop simulation. Given the variability in reproductive strategies of biological resources, it is important to assess varying management strategies to avoid under- or overexploitation of a resource. The following major report provides a comprehensive review of fisheries management strategies in Canada, reproductive traits of various species, and matrix projection modelling. Following this review, I present the results of original research consisting of a management strategy evaluation of hypothetical resources with various reproductive strategies under different management strategies and differing stock health, economic pressure, and ecosystem health scenarios. I then provide linkages between my findings and real-world examples where such management strategies have been implemented. My findings suggest that the implementation of precautionary approach principles may be the best tool for managing various biological resources and balancing social, biological, and ecological objectives.

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ABBREVIATIONS

CMA	Crab Management Area
CPUE	catch-per-unit-effort
DFO	Department of Fisheries and Oceans Canada
EAFM	ecosystem approach to fisheries management
EBFM	ecosystem-based fisheries management
EBM	ecosystem based management
FSP	fish stock provisions
HCR	harvest control rule
IBM	individual based models
LRP	limit reference point
MSE	management strategy evaluation
MSY	maximum sustainable yield
PA	precautionary approach
SS	single species
SSB	spawning stock biomass
TAC	total allowable catch
TD	test diameter
TEP	total egg production
USR	upper stock reference

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1.0 INTRODUCTION

Management Strategy Evaluation (MSE) involves simulating varying management strategies and assessing outcomes and trade-offs (Punt et al. 2016). A common goal of MSE is to determine the best management strategy to meet fisheries objectives and to inform management decisions. MSEs can help determine the best management choices for addressing various objectives, including stock recovery, maintaining catches, and increasing harvest levels. Management strategies for one resource may not be successful for another due to location (e.g., remote versus near urban development), differences in habitat use (e.g., shallow nearshore versus offshore pelagic), and different life histories and reproductive strategies (e.g., fast-growing and short-lived versus slow growing and long-lived). Management strategies are not universal; therefore, MSE is a tool that should be implemented for a specific resource to determine the best course forward in meeting fishery objectives. Such evaluations have become increasingly important in Canada, considering recent fisheries management changes at the legislative level that prescribe the requirement to use the precautionary approach (PA) and co-management strategies in addition to considering ecosystem interactions (*Fisheries Act* 1985).

This paper attempts to use MSE to assess the effect of the PA, co-management, and ecosystem-based fisheries management (EFBM) strategies on resources with varying reproductive traits under different scenarios. Specifically, the objectives of this report include the following:

- Review modern fisheries management with a focus on Canadian fisheries;
- Review fish reproductive strategies and how they relate to fisheries management;
- Review matrix projection models and their use in fisheries management and MSE;

- Present the methods, results, and discussion of original research, which included simulating five hypothetical populations with varying reproductive strategies under different management strategies and scenarios; and,
- Compare the results of the original research with real-world fisheries.

2.0 MODERN FISHERIES MANAGEMENT

Before describing modern fisheries management ideologies broadly, it is important to briefly discuss the tools used to manage fisheries. Examples of fisheries management tools include spatial management measures such as defined fishing areas, marine protected areas, temporal controls on seasonality, restrictions on gear types, quotas control including total allowable catch (TAC), and effort restrictions, among others (Worm et al. 2009; Hilborn and Ovando 2014). Often, these tools are species or habitat-specific and will vary regionally (Worm et al. 2009). Though management strategies may include a variety of stakeholders, be more precautionary, or focus on ecosystem health, the basic tools used for managing the fishery are similar.

Modern fisheries management can be divided into three broad ideologies herein termed “strategies”: PA, co-management, and EBFM. As part of the Canadian *Fisheries Act* (R.S.C., 1985, c, F-14), decision-making should consider all three forms of management, in addition to maintaining major fish stocks at levels that promote sustainable fishing, setting limit reference points to identify problematic states, and rebuilding stocks that drop below set reference points. Though discussed separately here, these strategies can be used in unison to satisfy policy directives. For example, a co-managed fishery can implement a PA framework that incorporates ecosystem information in assessing stock status.

The success or failure of a management strategy often depends on understanding the resource, including life history, habitat requirements, and an accurate estimate of abundance, size structure, and overall health. Stock assessments aim to quantify these metrics and provide some guidance on developing the management strategy, tools, and Harvest Control Rules (HCRs), which explicitly detail management actions under given circumstances (Worm et al. 2009). An assessment is achieved using fishery dependent (i.e., logbooks, landings) and fishery independent (i.e., standardized scientific survey) data. Fishery data is typically available throughout the fishing season through logbooks, vessel tracking systems and observer data (Karp et al. 2023). Obtaining the data is relatively cost-effective, outside of compensating fisheries observers who take biological measurements of the catch (Karp et al. 2023). Unfortunately, fishery data is often relatively clustered due to the nature of harvesters targeting areas of highest abundances and can thus miss changes in resource abundance in fringe habitat (Karp et al. 2023). In contrast, survey data can be costly; however, the surveys can be replicated annually, and a rigid statistical sampling design can provide information on fringe effects and representative data on population changes through the stocks range (Karl et al. 2023).

2.1 Precautionary Approach

When there is a high level of uncertainty regarding stock health or a general paucity of data, resource managers can take a PA to fisheries management (DFO 2009). PA management strategies aim to maintain caution in setting harvest limits when scientific information about the resource is limited or uncertain (DFO 2009). In recognizing uncertainty in stock health metrics, such as pre-recruit abundance or spawning stock biomass (SSB), managers can move forward cautiously and limit harvest rather than take uninformed risks that could potentially lead to overexploiting the resource.

A global push towards PA management systems began in the early 1990s (FAO 1995, 1996) but wasn't formally introduced into Canadian fisheries management until 2001 (DFO 2009). Between 2001 and the present, a PA management system has been implemented in most major Canadian fisheries (DFO 2009; Oceana Canada 2023). How the PA has been implemented however differs by resource with many still lacking scientifically derived reference points (Oceana Canada 2023). Canadian fisheries are progressively integrating PA frameworks into resource management, but a high proportion of stocks still require them. For example, less than 50% (44%) of 905 Canadian research and management documents between 2000 and 2020 mentioned PA (Boyce et al. 2021).

The framework implementing the PA in Canada is guided by several documents which highlight a focal end goal of promoting transparency in decision-making (e.g., DFO 2006, 2009). However, some have argued that the use of scientific evidence in Canadian fisheries management does not align well (or wholly) with PA principles (Winter and Hutchings 2020). The PA is built upon scientific evidence, but in Canada scientific assessment advice is only part of the decision-making framework, and other factors such as industry demands or politics may take precedence and ultimately minimize scientific advice when determining harvest levels for a stock (Winter and Hutchings 2020). In Canadian fisheries, the Department of Fisheries and Oceans Canada (DFO) undertakes final decision-making for resource use , and these decisions may or may not be based on scientific evidence. Regardless, if implemented correctly, a rigorous PA strategy should include the following principles (Winter and Hutchings 2020):

1. Defining quantifiable reference points that align with fishery objectives;
2. Risk assessment of the fishery to overharvest, environmental factors, and anthropogenic influences;

3. Determine HCRs that align with reference points, objectives, and risk;
4. Peer-review process of the management strategy and scientific evidence; and,
5. Scientific advice process that is transparent with all stakeholders and the public.

The PA to fisheries management is a single-species management tool implemented in Canada through predetermined HCRs aligning with stock status (DFO 2006, 2009). Stock status includes three zones: healthy, cautious, and critical (DFO 2006). The boundaries of the zones are set by a Limit Reference Point (LRP) and an Upper Stock Reference (USR) for a stock health indicator (Figure 1; DFO 2006). In many cases, indicators are tied to abundance or biomass, including the common SSB metric; however, other metrics, such as total egg production (TEP) or recruit biomass, can be used (DFO 2006, 2023). A single indicator (most typically spawning biomass) is sufficient to comply with the *Fisheries Act* and has been the historic norm practice, but there is growing recognition that this is not a holistic approach and recent advocation for multi-indicator approaches to feature more commonly in stock status determination (Mullowney and Baker 2023).

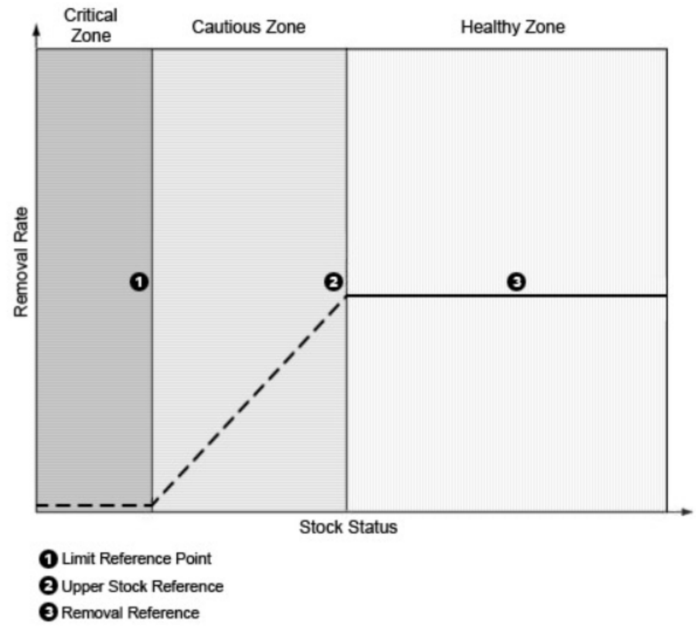

Figure 1 Precautionary approach framework in Canada (DFO 2009).

The LRP is the boundary between the critical and cautious zones. Conceptually, at any point lower than the LRP further harvest or population decline could lead to serious harm to the stock (DFO 2009, 2023). Under DFO (2023), serious harm can be defined as a state where the stock cannot be rebuilt or rebuilding of the stock will occur over a long period (DFO 2009). The LRP forms the basis for preventing serious harm and implementing the PA in accordance with the Fish Stock Provisions (FSPs) as updated as part of the 2019 *Fisheries Act* amendment (DFO 2023). The updated FSPs outline objectives centred on the LRP, including promoting or rebuilding stock status above the LRP (DFO 2009, 2023).

Selecting a stock health indicator can be difficult, as can setting reference points in relation to that indicator (DFO 2023). Since the LRP is typically near the tipping point of serious harm for a stock, it can be easy to underestimate or overestimate the reference value. For example, the point of serious harm can be somewhat estimated for a resource with a detailed history of collapse, given the knowledge of the stock. However, for a resource where past management has been successful and collapse has not occurred, the point of serious harm can be difficult to estimate. Thus, the LRP carries a high degree of uncertainty and is often a "best guess" which itself can be a complicated process (DFO 2023). The selection of the indicator and LRP should be based on the best available science regarding stock health, align with objectives for the fishery (i.e., prevent serious harm, rebuilding recruit biomass), be realistic, and be dynamic through an adaptive management strategy (DFO 2023). Regardless of indicator type, the LRP and USR are typically tied to maximum sustainable yield (MSY), with the USR suggested to align with 80% MSY and the LRP suggested to be 40% MSY if other more specific criteria for a given stock are not defined (DFO 2009).

The removal rate for harvest will be determined by stock status, with an HCR making the link between stock status and removals. If the indicator metric is greater than the set USR, harvest can proceed at the maximum removal reference as the stock is "healthy" (DFO 2006). In contrast, if the indicator is below the LRP and the stock is in the critical zone, harvest should be minimized to protect the stock (DFO 2006). In the cautious zone between the LRP and USR, harvest should be kept at a level that promotes the growth of the stock (DFO 2006). Finally, if the stock is in the critical zone, harvest should be minimal, if any at all, to protect the remaining individuals.

The key benefit of the PA is the prescription of conservative removal rates considering scientific uncertainty with the stock or environment (Winter and Hutchings 2020). Therefore, no harm should be inferred on the stock unless scientific evidence demonstrates that the harvest will not lead to sustained overexploitation or extirpation (Winter and Hutchings 2020). This is especially important in data-deficient stocks, which unfortunately remains the case for many stocks globally (Worm et al. 2009). However, quotas and harvest can fluctuate greatly in stocks with dynamic population growth, such as short-lived species with high reproductive output (Mildenberger et al. 2022).

Fishery independent data collected through surveys may suggest an alternative outlook on stock health compared to catch rates. In such cases, it can be challenging to explain uncertainty in stock size to harvesters, especially when catch rates are stable. It is not uncommon for fisheries catch rates to remain stable while the resource overall declines (Rose and Kulka 1999). Otherwise known as hyperstable catch-per-unit-effort (CPUE), this phenomenon reflects fisheries targeting and continually harvesting the resource from high-quality habitats and not identifying changes occurring in marginal habitats, which often show the first or clearest changes in abundance. Therefore, CPUE would not be proportional to stock abundance, which is often assumed by those using CPUE as an indicator of stock size (Harley et al. 2001). Hyperstable CPUE should be assumed in every fishery given the general preference by harvesters to occupy high-quality habitat where catch rates are greatest (Branch et al. 2006). Furthermore, hyperstable CPUE can actually be engineered in management by implementation of regulation changes that continuously promote spatial fishing patterns that focus on best habitats (Griffeth et al. 2023). In this situation, persuading managers or harvesters to accept a lower quota in response to signals in the survey data can be challenging when catch rates appear stable. Ultimately, hyperstable CPUE is one common example epitomizing the complexity of identifying, interpreting, and applying best stock health indicators in a PA system, particularly for data deficient stocks or those with a high level of user inputs in the management process.

Unfortunately, costs associated with extensive surveys are high which may limit survey data and the amount of scientific evidence available to inform PA management (Little et al. 2016). With high uncertainty in stock health due to a lack of survey data, cautious harvest may be prolonged (Little et al. 2016). However, this may not always be the case, and conversely, uncertainty in stock health can lead to the PA being misused. This was highlighted by Winter and Hutchings (2020) who reviewed Canadian management decisions and found that in the case of many Canadian fisheries, when the stock status is unknown, precaution-oriented management changes were often postponed (Winter and Hutchings 2020). To further complicate the issue, although inadequate management response in circumstances of unknown stock status often leads to overexploitation, the problem can go in the other direction and lead to the potential to underfish and thereby reduce economic yield if stock status is improving. Regardless, examinations of management decisions in Canadian fisheries reveal that although scientific advice and using scientific evidence are key principles of the PA, in reality of application, they often do not appear to be the determining factors of HCRs in Canada (Winter and Hutchings 2020). In most cases to-date, scientific advice appears to feature as a small piece in the decision-making process. It is normally coupled with input from fisheries managers and industry, which can override scientific advice (Winter and Hutchings 2020). Under a rigid PA strategy, scientific advice should be the most important evidence in the determination of HCRs (Winter and Hutchings 2020). Ultimately, PA implementation in fisheries management in Canada appears to be still be in a transitional phase in context of the intended shift from historic less-formalized decision-making practices to the more explicit process created by PA management frameworks

In general, PA management systems rarely account for socioeconomic objectives when stocks are in a sub-optimal state (Frid et al. 2023). Under the Canadian PA framework (DFO 2009), when a stock is in the "middle-ground" Cautious zone, managers are to find a balance between socio-economic and conservation, whereas in the healthy zone socio-economic objectives are prioritized and conservation is prioritized in the critical zone. Some have argued that In Canada, to-date, other considerations, such as socio-economic objectives, fishery-dependent data, and industry interests, appear to be deemed equal throughout the decision-making process regardless of stock status zonation (Winter and Hutchings 2020). Regardless, the prescription of HCRs aligning with the uncertainty of stock health is a key facet of the PA management system and is necessary if the system is to be implemented as designed, with full transparency in decision-making.

2.2 Co-management

Fisheries management is not simply about managing fish and aquatic ecosystems but also people and by extension social and economic objectives (Jentoft and McCay 1995). Since many stakeholders are typically involved in a fishery, and there are likely to be different points of view and ideas on how that fishery should be managed, it can be challenging to satisfy everyone. One management strategy that aims to rectify this is co-management, where there is a shared responsibility for resource use decisions and outcomes between various stakeholders, such as harvesters, managers, politicians, indigenous users, scientists, and conservation groups (Linke and Bruckmeier 2015).

The earliest known use of the concept of co-management in fisheries was in Norway during the 1890s and has since been used in Europe, New Zealand, the USA and Canada, among other countries (Jentoft and McCay 1995). In Canada, the collapse of Atlantic Cod (*Gadus morhua*) in the early 1990s created a unique opportunity for co-management to be implemented in Canada and it has since been broadly introduced into Canadian fisheries (Linke and Bruckmeier 2014; DFO 2019; Mullowney et al. 2020). The successful implementation of fully co-managed fisheries (i.e., consensus-based decision-making) in Canada has been limited outside of Indigenous and non-

Indigenous government land claim agreements (Swerdfager and Armitage 2023). Though DFO maintains final decision-making power in Canadian fisheries management, many fisheries incorporate components of co-management to help arrive at resource use decisions. The inclusion of stakeholders is a key priority for fisheries management in Canada, along with economic viability and environmental sustainability (DFO 2024). Given its importance, many of DFO's integrated fisheries management plans outline not only the biological information regarding a stock but also its economic, social, and cultural importance. This information can then be used to develop stewardship and socio-economic objectives for the fishery between stakeholders.

The conceptual benefit of co-management is that all stakeholders can participate in fisheries management and, in a perfect world, be part of the solution towards a social, economic, and ecologically sustainable fishery (Linke and Bruckmeier 2015). This allows various stakeholders to explain their objectives for the fishery, hoping that all stakeholders will be satisfied. In theory, if stakeholders are included in the development of fisheries management plans, they should be more receptive to any changes as trust builds between parties (Guidetti and Claudet 2009). Consensus-based decision-making has been deemed an important component of a co-managed fishery as it maintains fairness between stakeholders (Davis 2008). Similar to the PA, scientific advice should play a role in decision-making and the development of management plans. To facilitate this, co-management plans should use plain language and provide detailed explanations of technical work to ensure that all stakeholders can understand the science or other sources of information used to help inform or align the various stakeholders on stock status (Pinkerton and John 2008).

One of the key challenges with co-management is the lack of shared perspectives outside of fisheries policy; for example, stakeholders will likely rarely share the same idea of successful social, economic, or ecosystem health objectives (Linke and Bruckmeier 2015). The lack of shared objectives can lead to poor or counterintuitive objectives and inconsistent techniques for applying management rules (Mullowney et al. 2020). This creates a complex dynamic in management decision-making as there are poor or competing objectives which increases uncertainty in the fisheries future (Linke and Bruckmeier 2015). Though all parties usually agree that some level of management or harvest restriction is required, there are often disagreements about the form in which they are implemented (Jentoft and McCay 1995). In addition, there are often discrepancies in biological and fishery knowledge among stakeholders (Wilson et al. 2003). For example, scientists may utilize complex stock assessment or modelling techniques to determine the status of a stock, while alternatively, harvesters will often perceive the status of the resource based on their observations through catch data. Ideally, model-based assessments would match direct user observations, but in cases where there are disagreements in signals of stock health, co-management systems can be challenged. Such disagreements and issues can lead to delays in decision-making and add further complexity to the process of focusing on shared stock goals between key stakeholders.

One common example of complexity created by conflicting survey-based versus fishery-based signals and relevant to co-management is when CPUE, a common fishery index, is hyperstable, as previously described. If quotas or harvest rates are more closely aligned with fishery data and CPUE is hyperstable, it could lead to overexploitation of the resource or unintended negative biological consequences on stock health (Griffeth et al. 2023). One classical example of this occurred upon the aforementioned collapse of Atlantic cod, specifically in the collapse of the

Northern Cod stock off eastern Newfoundland and Labrador (NL), whereby catch rates of the offshore trawler fleet remained high on shrinking aggregations of fish, while surveys and fisheries occurring over the broader stock range were showing declines (Rose and Kulka, 1999). Another factor affecting co-mangement is spatial scale of management versus biological units. Complexities arising from differences in management versus biological scales have been shown in the NL Snow Crab fishery, where a scale mismatch occurs, with the spatial resource management areas not aligned with the spatial distribution of the biological resource (Mullowney et al. 2020). Unfortunately, a lack of shared perspective between managers, industry, and the scientific community has led to an inefficient spatial management system (Mullowney et al. 2020). Localized Crab Management Areas (CMA) developed early in the history of the fishery are small and preferred by managers and industry to control fleet dynamics (Mullowney et al. 2020). Comparatively, scientific examinations of larval drift, genetics, resource synchrony, and movements from tagging studies show that the biological distribution of the stock is much greater than what is perceived within most smaller CMAs (Puebla et al. 2008; Mullowney et al. 2018; Mullowney et al., 2020). It's suggested that from a biological stance the resource would be better managed at a higher level (i.e., fewer CMAs), given the spatial structure of the resource and natural movement through CMAs.

Balancing ecological, economic, institutional, and social trade-offs is a key component of fisheries management (Stephenson et al., 2018). Some management strategies, for example, PA, place an emphasis on the stock (particularly when it is in poor health), whereas co-management can lead to all components of resource management being more equally represented. Conflict can arise when one component begins to outweigh others during decision-making. For example, negating ecological objectives in preference for economic objectives during periods of unrest in the local economy or negating economic objectives in lieu of ecological objectives when stocks initially start showing signs of improvements.

Ultimately, as with any management strategy, clear and adaptive objectives agreed upon by all stakeholders may be the key to a successful co-managed fishery. If there is an incentive within the fishery to collaborate, there may be a higher chance of success in aligning stakeholder objectives (Davis 2008). In addition, integrating PA components, such as scientifically driven reference limits, may assist co-management strategies by limiting the extent to which socio-economic objectives can outweigh other facets (e.g., biological health).

2.3 Ecosystem-Based Fisheries Management (EBFM)

Ecosystem-based management (EBM) is a tool for including natural and anthropogenic factors in management decisions (Link and Browman 2014). The classical approach to fisheries management is Single Species (SS) in absence of applied ecosystem information. When ecosystem inputs are applied to management through SS it is termed Ecosystem Approach to Fisheries Management (EAFM). EBM is much broader than EAFM, attempting to focus on all components of the ecosystem (Link and Browman 2014). Between EBM and EAFM lies ecosystem-based fisheries management (EBFM). The difference is that EBFM typically manages multiple species, whereas EAFM tends to focus on a single stock (Link and Browman 2014) and EBM on all biological and anthropogenic factors affecting a given ecosystem.

Although the idea of ecosystem-based management has existed for more than 100 years, EBFM is a relatively new formalized management strategy and is often considered the future of fisheries management (Cucuzza et al. 2021). Being that EBFM formalization is relatively new, the philosophy comes with inherent challenges, such as a lack of agreement on prescribing such a strategy (Pitcher et al. 2009). The key difference is that PA and co-management are typically focused on the resource, ignoring the ecological complexities of marine and freshwater systems and the interconnection of various species and habitats (Link and Marshak 2021). EBFM aims to rectify the lack of inclusion of ecosystem integrity in fisheries management by taking a holistic approach, including not only the target species but also lower trophic organisms and physical, economic, and social values (Link and Marshak 2021). Target species for management don't have to be aquatic-based, such as pelican management, considering increased predation of juvenile fishes (Sanchirico and Essington 2021). Finally, these components of a fishery are not examined independently in EBFM but how they interact (Link and Marshak 2021).

Given EBFM has only been formalized and implemented in fisheries management since the late 1990s (Link and Marshak 2021), an official ecosystem approach has seldom been used in Canadian fisheries management, with only 1% of 905 research and management documents between 2000 and 2020 mentioning an ecosystem approach (Boyce et al. 2021).

The framework of EBFM will vary by jurisdiction and fishery (ies); however, the core components remain the same. These are focused around ecosystem-level planning, the use of scientific advice, prioritizing species and habitats, examining trade-offs, providing management advice based on ecosystem knowledge (or uncertainty), and the hope of maintaining resilient ecosystems which lead to resilient stocks and fisheries (Link and Marshak 2021). Balancing multiple stakeholders, industries, and fisheries within an area can be highly difficult, with varying objectives in allocating a resource (Link and Marshak 2021). Multiple users also drive various stressors, and often, in single-species management, there is a high focus on the target species. One of the benefits of

EBFM is the holistic approach that incorporates the potential effects of other activities on the ecosystem and resources (Link and Marshak 2021).

In SS management, species of lower value commercially are often not well studied, and their interactions with target species are often misunderstood. EBFM expands the lens of scientists and managers to examine these interactions and the potential adverse impacts of fishing, whether ecological (i.e., removal of predators) or physical (i.e., habitat degradation from gear) (Link and Marshak 2021). Species of lower social or economic value, such as benthic invertebrates, can provide powerful bottom-up effects on an ecosystem, such as being the primary food source for larger predators, such as freshwater salmonids. As is typical within a food web, it has been shown that fewer benthic invertebrates can equal fewer and less-conditioned salmonids (Bruder et al. 2017).

Climate change will profoundly affect an already dynamic environment, potentially negatively impacting ecosystem health, habitat, and population dynamics (Link and Marshak 2021). For example, this was recently observed in the Bering Sea of Alaska, where a historic collapse of more than 10 billion Snow Crab has been linked to a marine heatwave (Szuwalski et al. 2023). Other climate-driven ecosystem changes will occur elsewhere, including potential for unforeseen positive outcomes from climate forcing for a given region or species. Management strategies incorporating ecosystem components, such as ocean temperatures and climate-related variables, could create a more resilient fishery to environmental change. These ecosystem components can affect biomass and catch rates. Therefore, strategies should account for varying top-down and bottom-up effects such as predation or primary production (Howell et al. 2021). In addition, models can consider the potential repercussions of oscillations by aligning environmental parameters with past events such as the North Atlantic Oscillation or El Niño and evaluating outcomes following simulation (Howell et al. 2021; Yin et al 2021). Unfortunately, these models are complex, requiring a variety of climate, fishery, ecosystem, and biological data.

EBFM is not without limitations. The management strategy hinges on the availability of data, functional models, and the buy-in of stakeholders (Link and Marshak 2021). EBFM framework and modelling are inherently complex and can be a difficult learning curve (Fletcher et al. 2010; Gruss et al. 2017). As previously mentioned, a benefit to EBFM is a broader understanding of ecosystem dynamics and interactions; however, that understanding only comes from data and research invested into those interactions. It can be difficult to incorporate enough data in the modelling process to make the model results realistic (Howell et al. 2021). Research can be expensive and extensive for multi-species and environmental studies and can take decades to capture natural temporal variation. This issue is exacerbated when considering climate change and dynamic environmental conditions (e.g., atmospheric oscillations). It is difficult to promote the implementation of such studies when those resources could be used to further study key species. Finally, once multi-species data is available and compiled, models involving species interactions and interactions with the environment can be highly complex (Gruss et al. 2017). This results in models that can overfit the data and be useless for other applications or fisheries. Given the complexity and lack of model utility, the results can also be highly uncertain.

3.0 FISH REPRODUCTIVE STRATEGIES

Fisheries management should incorporate knowledge of reproductive biology into the decision-making process as these characteristics will determine recruitment and productivity (Morgan

2008). Knowledge of a species reproductive strategy will allow for better harvest management (e.g., male-only fishery), season timing (e.g., avoiding spawning), and protection of habitat (e.g., reducing trawling in spawning grounds). Complicating the inclusion of this knowledge in fisheries management is the variability and uncertainty in the reproductive strategies of fish, as well as the lack of information regarding reproductive biology.

At the most basic level of reproductive biology, there are two strategies: *r*- and *K*-selection (Pianka 1970). Species that are *r*-selected are characterized by short lifespans, early maturity, high fecundity, and high population growth rates, whereas *K*-selected species are the opposite, typically long-lived, mature later, have low fecundity with increased parental care, and low population growth rates (Pianka 1970; Reznick et al. 2002). *R*-selected species tend to follow Type 2 or Type 3 survivorship curves, and *K*-selected species are aligned with Type 1 (Reznick et al. 2002; Figure 2). The key difference between the two is density-dependency. Mortality is typically density-independent in *r*-selected species, resulting in rapid population growth and density-dependent for *K*-selected species, capping population growth when abundance reaches carrying capacity (Reznick et al. 2002). Though on a broad scale, the idea of two strategies is ideal for understanding general life histories; most species fall somewhere between the two.

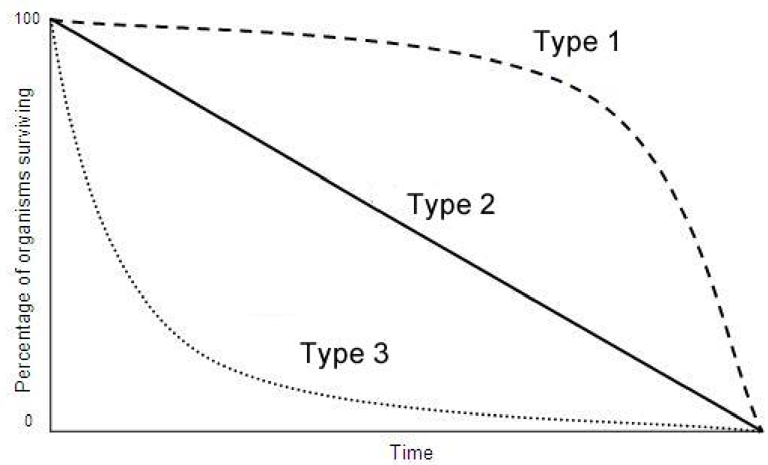

Figure 2 **Type 1, 2, and 3 Survivorship curves (Nahass and Arrington 2023).**

Reproductive strategies in fishes are highly variable, with many unique life histories ranging from *r*- to *K*-selection. Often, the line between *r*- and *K*-selection is blurred in fish. An example being Pink Salmon (*Oncorhynchus gorbuscha*), which have a short lifespan (two years), early to mature (age-2), and high fecundity with egg-juvenile survival as low as 1% (Geiger et al. 2007). Though not technically a fish, but is harvested in Canada's north, Atlantic Walrus (*Odobenus rosmarus rosmarus*) is an example of a species following *K*-selection (COSEWIC 2017). Walrus in Canada's north can live up to 40 years old, reaching maturity between 4 and 13 (COSEWIC 2017). Females have a low reproductive rate and don't usually give birth annually (COSEWIC 2017). An example of where the line between *r*- and *K*-selection can be blurred, are sturgeon. The Amur Sturgeon (*Acipenser schrenckii*) is known to survive up to 60 years, maturing at approximately 9 to 14 years (Daskin and Tilman 2022). Though long-lived with a later age-at-maturity, the species has high fecundity, producing over 150,000 eggs without parental care (Daskin and Tilman 2022). The lack of parental care is typical for most fishes as the majority are oviparous, where a female develops eggs, the female spawns (i.e., releases) the eggs, those eggs are fertilized by a male either pre- or post-spawn, and then eggs hatch with no parental care. Though a relatively straightforward process, there is much variation in frequency, timing, and specific life history traits.

Spawning frequency is a key component of reproductive potential, as the more spawning events occur, the higher potential for enhancing the population. Spawning frequency can be multiple times per year (e.g., White Mullet, *Mugil curema*, Solomon and Ramnarine 2007) to decennial (e.g., Bigmouth Buffalo, *Ictiobus cyprinellus*, Lackman et al., 2023), to once in a lifetime (e.g., Pacific salmon spp.). In the example of Pacific salmon, fisheries management must consider multiple years of potential harvest before a cohort can migrate and spawn. Within a species or stock, spawning frequency can also be variable. For example, the Bull Trout (*Salvelinus confluentus*) are iteroparous, typically spawning once per year; however, skip spawning has been identified in some populations (Sinnatamby et al. 2018). During periods of increased adult density resulting in reduced resource and habitat availability, female Bull Trout were found to skip a spawning season (Sinnatamby et al. 2018). As Bull Trout typically occupy low-productivity systems, skipping a spawning season is beneficial to avoid energy expenditure for egg development, redd construction, and spawning (Sinnatamby et al. 2018). Skip spawning can be sporadic and tied to habitat conditions and population density, so management strategies should incorporate the potential for skip spawning when it's been documented in a population or species. Management plans lacking the inclusion of such known reproductive responses to population dynamics or environmental change that may affect spatial and temporal changes to reproductive output could be prone to overestimate recruitment and potentially overfishing.

Spawning timing within the year has management implications for fishing seasons. Timing can vary throughout the year, such as in the winter (e.g., Burbot, *Lota lota*, Jude et al. 2013), spring (e.g., Walleye, *Sander vitreus*, Izzo et al. 2023), summer (e.g., Pink Salmon, Geiger et al. 2007), or fall (e.g., Lake Whitefish, *Coregonus clupeaformis*, Roseman et al. 2007). Timing can vary within a species; for example, McMillan et al. (2014) identified an early spawning population of Coastal Cutthroat Trout (*Oncorhynchus clarkii clarkii*). The species typically spawns in the spring; however, the authors observed spawning fish in October and November. The early spawning was suggested to be due to the unique habitat characteristics of the system and flow regimes. As an example of management responses to biological variation, in this example the early spawning window should result in a population-specific season closure to protect vulnerable fish rather than the typical blanket spring closure for cutthroat trout.

Multiple reproductive strategies within the same population can increase the complexity of management. For example, Jude et al. (2013) summarized five reproductive strategies for the winter-spawning Burbot inhabiting the Great Lakes. Strategies included migrating into tributaries to spawn with offspring remaining in the tributary to complete their life history; fish migrating and spawning in tributaries with juveniles returning to the lake the year following; fish migrating and spawning in tributaries with larvae returning to the lake immediately after being spawned; fish spawning nearshore; and delayed spawning (i.e., spring and summer) in offshore areas. Similarly, Izzo et al. (2023) found that Walleye (*Sander vitreus*) within Green Bay of Lake Michigan had multiple spawning strategies, including tributary and open-water spawning. Fish in both examples were known to mix throughout the year, which can have implications for management. For example, in Green Bay, the Walleye in the north is managed by Michigan, whereas Wisconsin manages the southern Walleye (Izzo et al. 2023). The boundaries have no biological significance, and little to no mixing is assumed with Walleye spawning either in north or south tributaries; however, Izzo et al. (2023) identified that not only were fish mixing throughout the year, but

Walleye were also spawning offshore. Given the potential uniqueness of some fish to spawn offshore, these results have apparent implications for harvest management. This example highlights that firstly, stocks should be defined prior to the implementation of management strategies and secondly that caution must be taken in these strategies to avoid overexploiting sub-populations within larger stock complexes.

Further complicating reproductive biology in most species is inherent annual variability in reproductive output due to biotic and abiotic factors (Fogarty and O'Brien 2016). These factors can be natural (e.g., drastic temperature change during a key period) or anthropogenic (e.g., overexploitation). For example, Lackman et al. (2023) identified a Bigmouth Buffalo population in southeast Saskatchewan, Canada, with individuals up to 127 years old when it was initially assumed a max-age of 20 years (Lackman et al. 2023). The authors also suggested that less than 20 spawning events have occurred since 1894, with the episodic spawning likely due to the population requiring water-level fluctuations to be within a narrow range for spawning, hatching, and rearing (Lackman et al. 2023). A more detailed understanding of this population's reproductive biology will pave the way for better management, knowing the longevity of individuals and the natural conditions that drive variability in reproductive output.

Top-down processes such as increased predator abundance or overexploitation can also cause variability in reproductive output. Such examples in Canadian waters include Atlantic Cod (Hutchings 2005) and Snow Crab (Mullowney and Baker 2021). Between the 1980s and 2000s, including the early 1990s collapse, the length at maturity of cod declined by up to 20%, presumably due to over-exploitation (Hutchings 2005). Recovery of the species since has been potentially dampened by the reduced size at maturity due to a reduced population growth rate (Hutchings

2005). More recently, an unprecedented downward shift in size-at-maturity has been observed in the Newfoundland and Labrador Snow Crab fishery (Mullowney and Baker 2021). The authors suggest that size-selective overexploitation of large male Snow Crabs coupled with cold ocean conditions caused the reduction in male size-at-maturity. As in most cases of size-at-maturity reductions, the downward shift in size would be expected to reduce yields (male-only fishery) and potentially reduce stock growth potential (Mullowney and Baker 2021). Going further, the impacts of depleted densities of large males in some areas of the NL Snow Crab stock range has been suggested to have limited sperm availability for female Snow Crab, potentially dampening population growth (Baker et al. 2022). The NL Snow Crab fishery is a recent example of anthropogenic activities causing variability in reproductive output, given that sperm is limited for females due to the overexploitation of large males, but such examples of biological ramifications from overfishing have long since and broadly been raised in many global fisheries, such as Blue Crab (*Callinectes sapidus*) in Chesapeake Bay (Hines et al. 2003) .

Management strategies should incorporate species' reproductive biology, which is often the case. However, strategies should be adaptive and change based on new information (e.g., Green Bay Walleye; Saskatchewan Bigmouth Buffalo). Understanding reproductive biology will help influence management decisions on season timing, spatial closures, size limits, and harvest limits. Species that grow fast and die young with many offspring can rebound quickly, whereas long-lived species can take generations to recover. Knowledge of habitat requirements can also enhance fish populations through habitat conservation and remediation programs. A blanket approach to fisheries management is no longer suitable, and species-specific management strategies should be guided by reproductive biology and potential if the data exists.

4.0 MATRIX PROJECTION MODELS

MSE is a process that combines fishery dependent and independent data, population models, and HCRs through a closed-loop system to simulate the effects of alternative management strategies prescribed in a fishery (Punt et al. 2016). The MSE process can present the trade-offs of alternative strategies during decision-making for managers to better understand the potential risks and consequences of various HCRs (Punt et al. 2016). A variety of population models can be used during the MSE to simulate the effects of exploitation on a fishery, including simplistic matrix projection models (Power 2007).

4.1 Overview

A classic approach to population simulation is matrix projection models, which, in its simplest form, only require two parameters: survival and fecundity (Caswell 2001; Figure 3). Survival and fecundity values are assigned to individuals through age-classes (i.e., age-based model) or stages (i.e., stage-based matrix model; Power 2007). Stages are typically based on length (e.g., 100 to 150 mm) and are used when there is a paucity of age-length data, there are differences in age- or length-at-maturity between sexes, or when age is generally a poor predictor of life history (Power 2007). For each age or stage, survival (proportion surviving to the next age) and fecundity (if that age is mature), estimates are derived and input into the matrix (Figure 3).

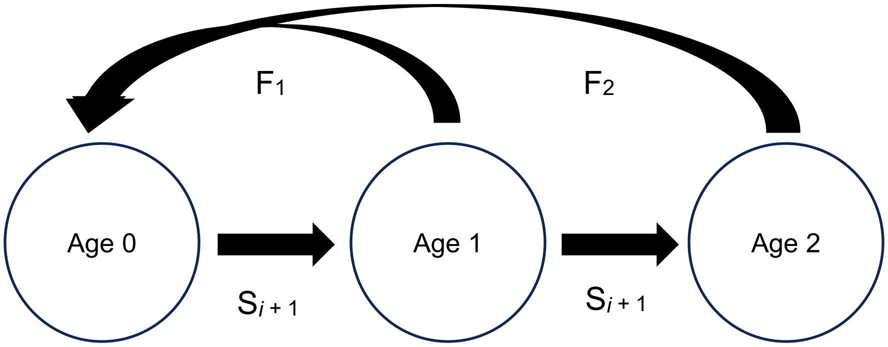

Figure 3 **Typical life cycle used in population matrix models where Si + 1 is the estimated survival of an individual progressing from age$_i$ to age$_{i+1}$ and F is fecundity of that specific age.**

$$M = \begin{bmatrix} F_0 & F_1 & \dots & F_{k-1} & F_k \\ S_0 & 0 & \dots & 0 & 0 \\ 0 & S_1 & \dots & 0 & 0 \\ 0 & 0 & \dots & S_{k-1} & S_k \end{bmatrix}$$

Figure 4 **General form of a Leslie matrix model, where S is survival from age$_i$ to age$_{i+1}$, F_i is fecundity at age i, and k is the maximum age.**

After estimates are derived through known age or stage-structured data or from literature, the model can progress with an initial population abundance estimate, which is in a single-column matrix with abundance values for each age or stage. The life history and abundance matrices are then multiplied, and the abundance by age or stage for the following year (*t + 1)* is derived. The population can be simulated again by multiplying the new *t + 1* abundance column by the original life history matrix.

4.2 Benefits of Matrix Projection Models

Matrix models have numerous benefits over other population models, such as exponential and logistic growth models, individual-based models (IBM) and stock-recruitment models (Perry et al.

2002a; Sable and Rose 2008). Many other models treat all individuals of a stock the same, with identical mortality and fecundity rates; however, this isn't typically true in nature, and matrix models can use age- or stage-specific survival and fecundity rates to account for variability (Perry et al. 2002a). This allows users to examine the outcome of alternative management strategies on specific age or stage classes, which is essential when only a portion of the fishery is affected by harvest (Caswell 2001). Projection matrices are also easy to construct and provide valuable information about a stock, including population growth rates and elasticity (Caswell 2001; Sable and Rose 2008).

4.3 Assumptions and Limitations

Matrix models for fisheries assessments and MSE will have several key assumptions. Typically, the time step is a single year, which is appropriate for iteroparous spawners; however, other time steps or stages can be used, such as stages for populations lacking age data (van der Lee and Koops 2020). Each progression of the matrix is a single birth-pulse event with the number of offspring entering age (or stage) 0. Once again, this is appropriate for iteroparous spawners and can be adjusted based on the study species, such as monthly or other stage intervals. The sex ratio also needs to be determined. Past studies have used an all-female model (van der Lee and Koops 2020), whereas others use a variety of ratios with fecundity proportioned to the ratio (Daskin and Tilman 2022). For example, Daskin and Tilman (2022) halved fecundity to account for a 50:50 male to female ratio. Finally, the population is typically assumed to be closed, with no immigration or emigration (Power 2007).

4.4 Use in Fisheries Science

Beyond basic stock assessment, matrix models, be it age- or stage-structured, have been used for a variety of uses , such as fisheries MSE (Smart et al. 2017), assessing the potential implications of climate change on a stock (Durant and Hjermann 2017) and species-at-risk recovery (Daskin and Tilman 2022). The following summaries describe examples of studies utilizing matrix projection models for various objectives in fisheries science.

Smart et al. (2017) assessed management strategies for two species of sharks, the Silvertip (*Carcharhinus albimarginatus*) and Common Blacktip (*Carcharhinus limbatus*) in the Indo-Pacific using age-structured matrix projection models. Both species are targeted in subsistence and artisanal fisheries, and captures are often length-selected, with larger fish having an increased fishing mortality. Using a variety of studies to infer survival and fecundity estimates, Leslie matrix models were used to simulate the stock through various management strategies, including situations when juveniles are not harvested and when the breeding stock is protected and only juveniles are protected. The study concluded that having a maximum size limit to protect younger fish was the most conservative approach in addition to limiting fishing mortality on mature fish.

Durant and Hjerman (2017) examined the effects of climate change and fishing pressure on recruitment in five stocks using matrix projection models. The stocks included Cod, Haddock (*Melanogrammus aeglefinus*), Capelin *(Mallotus villosus*), Polar Cod (*Boreogadus saida*), and Herring (*Clupea harengus*). Parameter values were derived from stock assessments for each species and fed into age-structured projection matrices. The authors assessed the effects of fishing pressure and climate on each stock by examining the elasticity of reproductive rates to change.

Reproductive rates and population growth were dampened by sustained changes in age structure due to fishing pressure while warming ocean conditions increased the elasticity of reproductive rates.

Daskin and Tilman (2022) used an age-based matrix projection model to assess alternative supplemental stocking strategies as a recovery tool for the endangered Amur Sturgeon. Four populations were simulated for thirty years under three distinct scenarios: maintaining harvest with no supplemental stocking, supplemental stocking with age-0 fish, and supplemental stocking with age-1 fish. The main difference between the age-0 and age-1 fish is the increased survival rate at age-1. The study predicted that supplemental stocking with age-1 fish would be most effective in growing the population, rather than no stocking or stocking age-0 fish with reduced survival.

5.0 ORIGINAL RESEARCH

My original research included individual MSEs for five distinct hypothetical resources with varying reproductive strategies. Each resource was simulated under three scenarios based on stock health, economic pressure, and ecosystem health. Three management strategies—PA, co-management, and EBFM—were evaluated for each resource and scenario. The study aimed to compare simulation outcomes of selected performance metrics and examine the differences between management and reproductive strategies. The following sections describe my research's methods, results, and discussion.

5.1 Methods

The MSE generally followed the best management practices presented by Punt et al. (2016). A flow chart depicting the MSE is provided in Figure 5.

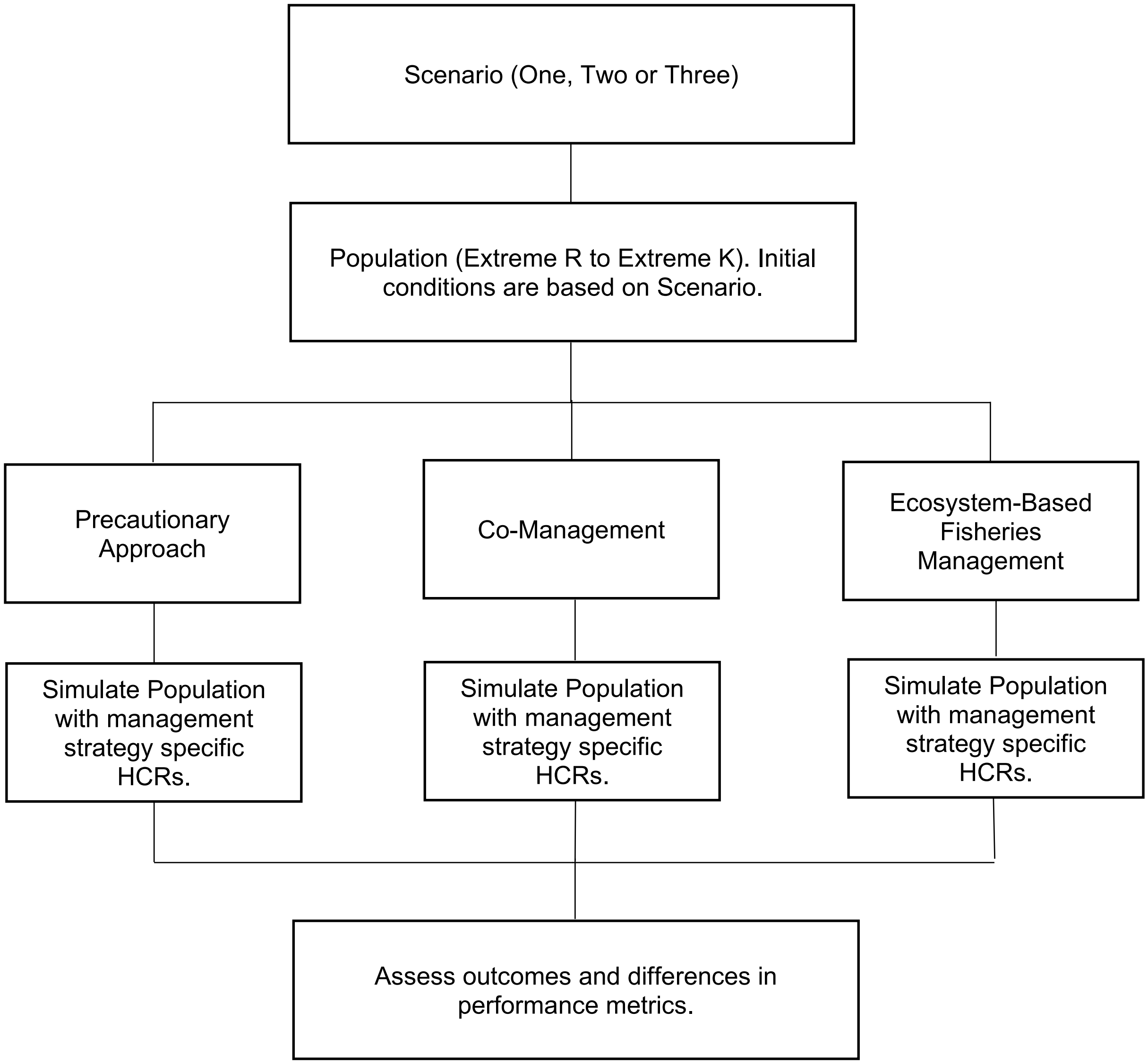

Figure 5 Management strategy evaluation flow chart.

Before the MSE, I first defined my resources and scenarios. I selected five reproductive strategies based on the general r- and K-selection theory to define my resources. The strategies scaled from extreme r (RE) (i.e., short lifespan, early age-at-maturity, abundant offspring) to extreme K (KE) (i.e., long-lived, late age-at-maturity, few offspring), with three other populations (*r* [R], mid [M], and *K* [K]) between the extremes. Though most fish are *r*-selected, the goal was to assess a wide range of reproductive strategies, given that some species have *K*-selected traits, such as sturgeon with long life spans (Daskin and Tilman 2022).

I used three scenarios to drive fisheries management decisions and HCRs for each simulation. The scenarios (Table 1) were based on hypothetical stock, economic pressure to harvest, and ecosystem health, and each scenario was randomly selected using an online random selection generator (RAND function in Microsoft Excel).

Table 1 **Scenarios for simulating populations under different management strategies.**

Metric	Scenario One	Scenario Two	Scenario Three
Stock Health	Healthy	Moderate	Poor
Economic Pressure	High	Low	Moderate
Ecosystem Health	Moderate	Healthy	Poor

The health metrics were used to inform fisheries management decisions and HCRs per fisheries management strategy. I used three categories to define stock, economic pressure, and ecosystem health for the simulation: Healthy/High, Moderate, and Poor/Low (Table 2). These metrics would be used to set the stage for each simulation (e.g., initial abundance), determine HCRs where applicable (e.g., co-management), and act as a measuring stick following simulation (i.e., compare health in year 1 to year 5). The PA only considered stock health, whereas co-management relied on the status of all three metrics. EBFM HCRs considered stock and ecosystem health.

Table 2 **Metrics used to define stock, economic, and ecosystem health for scenarios.**

Metric	Healthy/High	Moderate	Poor/Low
Stock	≥ 50 % of K	15 to 50% of K	≤ 15% of K
Economic Pressure	Increased harvest	Moderate harvest	Low harvest
Ecosystem	Index > 4.0	Index 2.0 to 4.0	Index < 2.0

Stock health was relative to total egg production (number of age-0), where 50% of carrying capacity was “healthy,” 35% was “moderate,” and 20% was “poor.” This metric determined the simulation's initial population size (i.e., each age class starting at 20%, 35%, or 50% of carrying capacity). Stock health was dynamic throughout the simulation, with HCRs changing with increasing or decreasing stock size.

Economic pressure was based on the pressure on the fishery to produce landings. High pressure inferred that increased harvest was required, even if that meant overexploiting the stock in the short term. In contrast, low economic pressure reduced the need to harvest, and management could be more cautious.

Ecosystem health was derived using an index based on the age-0 abundance of the five populations combined. The index was aligned with the TEP percent of MSY for each resource. An index of 1.0 signified that the stock TEP was 100% of MSY, whereas 0.0 meant 0% of MSY. The index for the four species not being simulated was determined by the ecosystem health metric at the start of the simulation, with healthy being greater than 4.0, moderate 2.0 to 4.0, and poor less than 2.0. The fifth stock, the one being simulated, was based on the stock health at the start of the simulation.

Finally, to assess outcomes, I compared the following performance metrics:

- spawning stock biomass (SSB) – total biomass of mature fish each year.
- total egg production (TEP) – abundance of age-0 fish each year.
- exploitable biomass – total biomass of fish exploitable for harvest each year.
- landings – total weight of fish harvested each year.
- cumulative landings – total weight of all fish harvested during the simulation.

- fishing mortality – percent of the population harvested each year.

Identification of Management Objectives

A key component of MSE is determining management objectives and performance metrics (Punt et al. 2016). Given that there are three scenarios, each with specific objectives (Table 3), similar to the metrics, the PA only considers the stock health, EBFM considers stock and ecosystem health, and the co-management strategy considers all three objectives. The stock and ecosystem objectives are similar: maintaining stock and ecosystem health when healthy, balancing maintaining the population with some growth when moderate, and growing the stock/ecosystem when in poor health. The economic objectives are to produce high landings when economic pressure is high, maintain sustainable landings when faced with moderate pressure, and be cautious when economic pressure is low.

Table 3 Management objectives by scenario and metric.

Scenario	Stock Objective	Economic Objective	Ecosystem Objective
One	Maintain	Produce high landings	Maintain/grow
Two	Maintain/grow	Cautious landings	Maintain
Three	Grow	Maintain landings	Grow

Development of Operating Model

I used Leslie matrix projection models to simulate each population and management strategy under all three scenarios. A matrix model for each population was constructed, using estimates of age-at-maturity, survival, and fecundity, described in further detail below. I simulated each population for 5 to 15 years (depending on age-at-maturity), with HCRs (i.e., level of fishing mortality) being implemented based on the management strategy and appropriate stock references and indicators. Harvest control rules (i.e., level of fishing mortality) were prescribed based on the management strategy and scenario being simulated.

Each population was assumed to have a 1:1 ratio of males to females; however, the matrix projection models act as all individuals producing offspring. To counter this, estimated fecundity was reduced to 50%. Therefore, it appears all fish produce offspring even though technically, offspring are only made by 50% of the population. Each time interval was a single birth pulse (i.e., all fecundity values summed to age 0), and harvest was post-breeding (i.e., all fish have produced eggs before harvest). No sex limitations were assumed in any populations, and it was assumed that male and female catchability and harvest were equal.

Selection of Parameters

For each population and associated matrix model, I estimated age-at-maturity, survival by age, fecundity by age, weight-at-age, and inferred density dependency, structured as a Beverton-Holt relationship.

Age-at-maturity and Survival

Age-at-maturity was age-1 for the RE population and was subsequently older for each population, with a max of 10 for KE (Appendix 1, Table 7). Survival estimates were purely natural mortality, with fishing mortality being subtracted from the survival estimate following management strategy implementation and HCRs based on population size and scenario. Age recruited into the fishery (i.e., age of legal size for harvest) was set at one year after maturity for the M, K, and KE stocks, while the RE and R stock's age-at-maturity and legal harvest were the same year (Appendix 1, Table 7). Annual survival estimates ranged from very low (RE) to high (KE)(Appendix 1, Figure 9) and generally aligned with Type 1 (RE, R), Type II (M), and Type III (K, KE) survivorship curves after year one (Figure 2).

Weight-at-age

The weight-at-age was structured as a logistic relationship, with fish experiencing rapid weight gain to maturity (Appendix 1, Figure 10). Weights were selected based on the assumed small size of *r*-select species and heavier *K*-select species. In addition, weights were determined by examining the initial population size at MSY and the five populations with the idea of having the RE population with the greatest biomass overall and the KE having the least. Therefore, this relationship also followed TEP, SSB, and exploitable biomass, which were determined by weight-at-age and calculated for each simulation year.

Though some of these assumptions are not typical of natural populations (e.g., natural mortality independent from fishing mortality or lack of immigration/emigration), this simulation was meant to be simplistic. Adding these factors would only make the simulation more complex and potentially add uncertainty during parameterization.

Fecundity

Fecundity (i.e., the number of eggs produced by each individual) was estimated following asymptotic growth for extreme *K*- and *K*-strategy, logistic growth for extreme *r*-strategy, *r*-strategy, and mid (Appendix 1, Figure 11). Total fecundity per female was halved for a presumed 50:50 male to female ratio (Daskin and Tilman 2022). The fecundity estimate was derived from the maximum number of developed eggs (halved) multiplied by the estimated survival rate of the eggs to age-0 (not specified in the matrix). Egg-to-age-0 survival is rarely known, and the estimate was set to the lowest level that still ensured population stability (Perry et al. 2002).

Population Growth Rate

Each population was simulated for 50 years using the selected survival and fecundity estimates to determine the population growth rate. The survival and fecundity estimates were then adjusted slightly to achieve the preferred population growth rate (Table 4). Population growth rates were set to relatively high growth (RE) or slow growth (KE). This initial simulation also produced the stable stage distribution, which was used to determine the initial population sizes, carrying capacity, and density dependency.

Table 4 Initial population growth rates for each population.

Population	Initial Population Growth Rate
RE	1.20
R	1.18
M	1.13
K	1.10
KE	1.05

Density Dependency

Density dependency was included in each population model and acted on all age classes independently. Density dependence was structured as a Beverton-Holt relationship:

$$D_A = \frac{r}{1 + \frac{b}{K} + N_{t-1}}$$

Where D_A is the density-dependency term to be multiplied by the survival rate in a given year, r is the maximum intrinsic growth rate for the population (Table 4), K is the carrying capacity for each age, b is a coefficient parameter, and N_{t-1} is the abundance for that cohort in the previous year. The coefficient for each population was solved as the value where the population growth rate reaches one (i.e., no further growth above K).

Carrying capacity was estimated using the stable stage distribution and intrinsic population growth rate. At the preferred population growth rate, the stable stage distribution was used to determine the population's age proportions. Initial population sizes were structured using this distribution, assuming that total biomass would be highest for the RE population and the lowest for the KE population. Carrying capacity was then estimated as the values multiplied by two and remained constant throughout the simulation. Figures demonstrating population growth and density-dependency using the Beverton-Holt relationship are presented in Appendix 1, Figure 12.

Identification of Management Strategies

Modern fisheries management in Canada can be divided into three broad strategies: PA, co-management, and EBFM. Given their prominence in modern management, I selected these three strategies for the MSE. Though they were examined independently for this exercise, there is often overlap in real-world fisheries.

The following sections describe each strategy's fisheries management decision-making process and HCRs. The general method is the same for each: a metric is used as a performance indicator for the resource, and HCRs are set for the level of that indicator.

Precautionary Approach

The PA aims to counter a lack of scientific information or uncertainty in the data with cautious harvest rates based on biological reference points of a metric related to resource health. In this simulation, the reference points were based on total age-0 fish, represented by TEP. Surveying the pre-recruit stages of fish and using this information as an indicator is valuable, as one can attempt to project these results forward and estimate the exploitable biomass that may be available in future years (Lo et al. 2016). As per DFO guidance under the fishery decision-making framework incorporating the PA and guidance for the selection of LRPs (DFO 2009, 2023), I established LRPs, USRs, and resulting healthy, cautious, and critical zones for each population (Appendix 2) that aligned with the example in Figure 1. These boundaries set the removal rate, which was equal to or less than MSY following DFO guidance (DFO 2009). MSY was assumed to be *K*/2 for this study, and TEP was used as the reference metric. When the stock was in the healthy zone (≥ 80% TEP_{MSY}), the HCR was set so the fishing mortality rate did not lead to stock decline over a productive peiod (i.e., healthy scenario). When the stock was in the cautious zone, the target fishing mortality rate declined with TEP until 40% of TEP_{MSY} was reached when the stock was in the critical zone. The decline in fishing mortality loosely followed DFO guidance (DFO 2009, 2013) and was generally linear between 40 and 80% of MSY. If the stock was in the critical zone, or below 40% TEP_{MSY} there was no fishing mortality to prevent serious harm to the stock (DFO 2023).

A key component of the PA principle is collecting scientific evidence to inform management decisions. Though hypothetical, the simulation assumes that the estimate of TEP in a given year is derived using a scientifically rigorous survey design and that scientific advice is communicated and used to make management decisions.

Co-management

The co-management strategy is a collaborative effort to determine the best action for HCRs. The benefits of co-management, which are that all stakeholders can participate in fisheries management, are not explicitly apparent in this simulation; however, to best simulate co-management, I had HCRs consider both stock and ecosystem health in addition to economic pressure toward making them realistic to “real-world” decisions bases (Appendix 2 Table 8).

HCRs were based on stock and ecosystem health and economic pressure. As previously described, stock health was healthy (> 50% of K), moderate (20 to 50% of k), or poor (< 20% of k) and was dynamic throughout the simulation. Economic pressure and ecosystem health remained static. The HCR was initially set with healthy stock, moderate economic pressure, and moderate ecosystem health, with a harvest rate similar to the healthy PA fishing mortality. If economic pressure increased, the harvest rate increased; if ecosystem health decreased, the harvest rate decreased; and vice-versa. The same strategy was used with moderate stock and ecosystem health, and economic pressure harvest rates being similar to 60% of MSY in the PA strategy, and low, moderate, and moderate being approximately 41%. Fishing mortality rates were based on directionality, rather than specific values. For example, if during a period of poor economic health with a healthy stock, prescribed fishing mortality was increased above the mortality rate set as part of the PA simulation, given that economic values outweigh biological values in the co-management approach when economic health is poor.

Ecosystem-Based Fisheries Management

The EBFM strategy can incorporate the stock health of all populations rather than single-species management. Making management decisions based on the health of the ecosystem can lead to more resilient populations and harvest levels that are not detrimental to both the stock and the ecosystem.

In this MSE, the EBFM strategy was prescribed using an index approach. For each stock, TEP was given a score, where 0.0 inferred that TEP was at 0% of MSY and a score of 1.0 means TEP was at 100% of MSY. The stock index could be greater than 1.0; for example, a stock with TEP for a given year of 120% would be given a score of 1.2. The scores for all five species were then combined for a total ecosystem index. Though a simplified version, other proposed EBFM frameworks have included similar criteria and scoring indices (Fletcher et al. 2010).

HCRs were set for the population being simulated based on the ecosystem index. The prescribed fishing mortality rate and ecosystem index have a similar relationship to that presented in the PA (Figure 1, Appendix 2 Figure 18). The percent of TEP for the simulated stock was based on the stock health (i.e., 100%, 70%, or 30% of MSY), whereas the other four stocks were set on the ecosystem health metric. In the healthy ecosystem scenario (Scenario Two), the four stocks were set to 80% MSY, 50% MSY for the moderate scenario (Scenario One), and finally, 20% for the poor health scenario (Scenario Three). Other than the stock being simulated, the remaining four stocks were kept static for the simulation.

Simulation

Each population was projected through time under each management strategy for the three scenarios. Simulation time met or exceeded the lifespan of an individual (Table 5). The simulation length allowed for multiple age classes to reach maturity under each management strategy, as the benefits or repercussions of management actions on longer-lived populations may not be observed in less than five years. Simulation and analysis were completed through a combination of Excel (Microsoft) and R Open-Source Programming (R Core Team 2021).

Table 5 **Simulation length (years) by reproductive strategy.**

Reproductive Strategy	Lifespan (years)	Age-at-Maturity (years)	Simulation Length (years)
RE	3	1	5
R	5	3	5
M	8	5	8
K	12	8	12
KE	15	12	15

Results

The results of the simulation were examined and compared, including TEP, SSB, exploitable biomass, annual fishing mortality, annual landings, and cumulative landings. Ultimately, the best management strategy was the one that most aligned with the fishery objectives.

Identification of Uncertainties

With the benefit of using hypothetical populations and parameters, my simulations had limited uncertainty in life history parameters and population size. The simulation did however have the following assumptions:

- In the healthy stock scenario (Scenario One), the fishery is assumed to have past harvest.
- The moderate and low stocks assume harvest contributed to their current state.
- The sex ratio for all populations is 50:50, males to females.
- Both sexes are harvested equally.
- All fish are of legal size at first year of harvest.

- There is equal fishing pressure among harvested age groups.
- Natural mortality is independent of fishing mortality.
- There is no immigration or emigration.

In addition, no environmental conditions or variability were incorporated into the model. Environmental conditions can drive fluctuations in recruitment, such as flow in lotic freshwater systems (Paul 2013), water levels in reservoirs (Radigan et al. 2024), or temperature (Bopp et al. 2023) among others.

5.2 Results

Scenario One

Scenario One was defined by a healthy stock, high economic pressure to fish, and moderate ecosystem health. The main objectives include maintaining a healthy stock, producing high landings, and maintaining/growing ecosystem health. A summary figure is presented in Figure 6 with individual results for TEP (Figure 19), SSB (Figure 20), exploitable biomass (Figure 21), annual landings (Figure 22), cumulative landings (Figure 23), and fishing mortality (Figure 24) presented in Appendix 2.

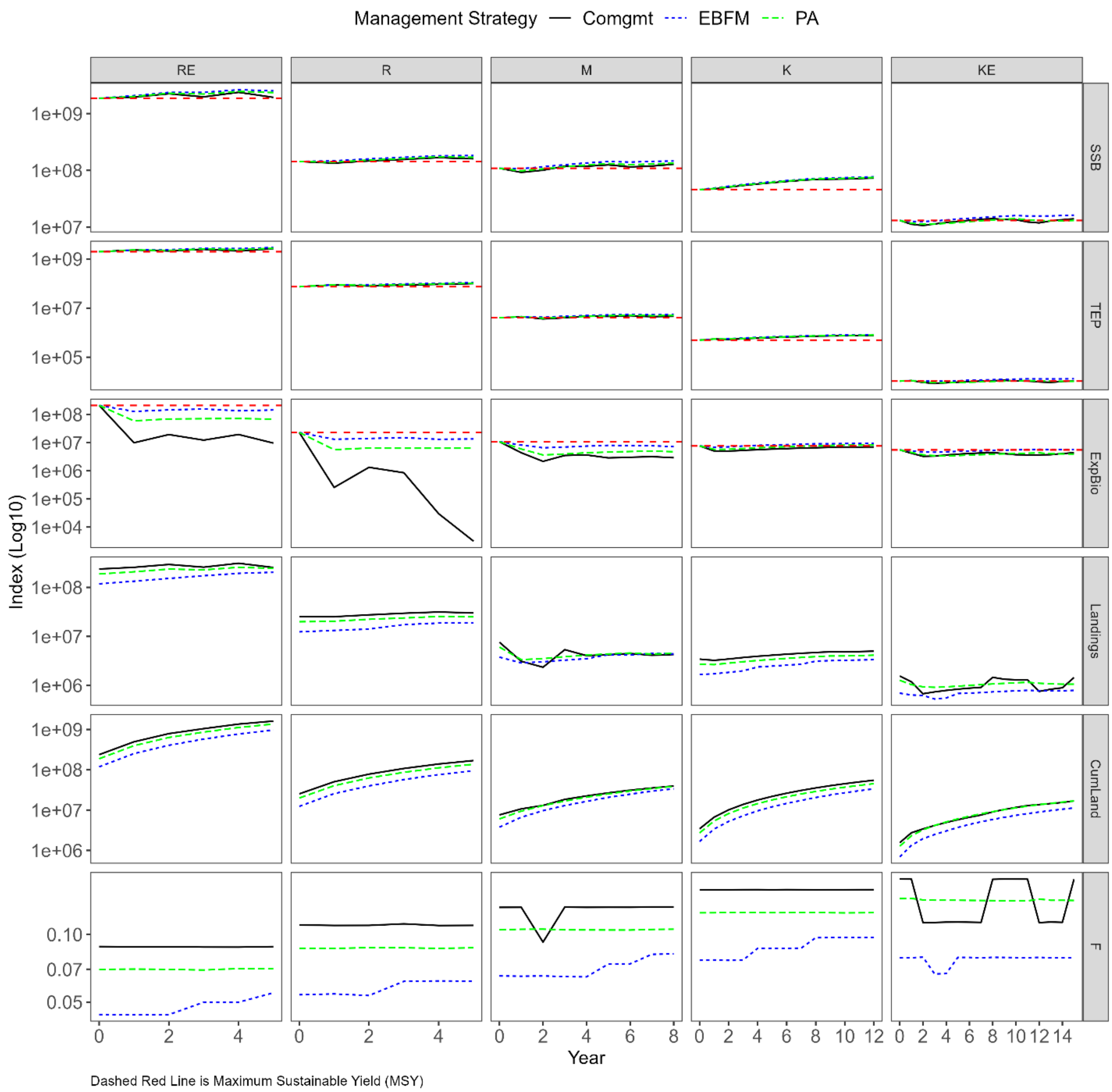

Figure 6 **Simulation results for Scenario 1. Columns represent populations RE to KE. Rows represent performance metrics including spawning stock biomass (SSB), total egg production (TEP), exploitable biomass (ExpBio), landings, cumulative landings (CumLand), and fishing mortality (F). Blue, black and green lines are individual management strategy (co-management, ecosystem-based fisheries management, and precautionary approach) results and the red dashed line is maximum sustainable yield for SSB, TEP, and ExpBio.**

TEP was the metric for determining stock health and HCRs for each population and management strategy. Following the simulation for Scenario One, TEP was consistently highest for stocks managed by the EBFM strategy and lowest for the co-management strategy, apart from the KE population, where co-management and PA were similar. All populations and management strategies finished at a level greater than MSY, apart from the KE PA (97% of MSY) simulation. This was slightly lower than the co-management (101% of MSY) and EBFM (124% of MSY). Populations with *r*-selected traits (RE and R) finished well above MSY (> 130% of MSY); however, total mortality averaged 98% each year. All stocks experienced an increase in TEP in year 1 and a decline in year 2 of the simulation.

SSB predictably followed a similar trend, with EBFM finishing with the greatest SSB remaining in all five simulations. The PA followed in all stocks except KE, where it was slightly lower than the co-management simulation. Again, SSB finished at or greater than MSY levels in all simulations.

Exploitable biomass was greatest for EBFM, followed by the PA strategy and co-management, again except for the KE simulation where co-management was slightly higher than the PA simulation. Apart from the KE, K-EBFM and the K-PA simulations, exploitable biomass in all populations was below MSY levels. The high level of harvest was most problematic in the *r*-select strategies, where the exploitable biomass consisted of mostly first-year recruits (Table 6). In the R population, ages 4 and 5 disappeared entirely (Figure 5).

Table 6 **Abundance by age for the RE population in year 5 of the simulation.**

	PA	Co-management	EBFM
Age 0	2,719,985,998	2,637,523,558	2,918,263,125
Age 1	22,766,877	19,222,048	23,684,608
Age 2	453,836	64,103	953,824
Age 3	2,229	91	7,982

Annual landings and fishing mortality were highest for the co-management strategy in most years. The exceptions were year 2 in the M simulation and years 2 to 7 and 12 to 14 in the KE simulation, where in both simulations, the PA strategy had a greater fishing mortality. In these years, early harvest drove the population into the "moderate" zone and prescribing a reduced fishing mortality rate and landings. Cumulative landings for the M and KE populations were similar for the PA and co-management strategies, whereas for the remaining populations, cumulative landings were greatest for simulations under the co-management strategy.

In a healthy stock situation, there was a bump in TEP and thus more landings in year 1 as the population was at 100% MSY and all mature fish spawned prior to harvest. The effect was exacerbated in the KE population simulation, with an increase of approximately 1,000 age-0 fish in year 1, which declines in subsequent years as mature fish are removed. The model used for the simulation is a post-spawn harvest, which resulted in elevated landings in the initial year of the simulation (year 0) due to a high stock abundance. Consequently, landings dropped in subsequent years and eventually levelled to a stable state.

Scenario Two

Scenario Two was defined by a moderately healthy stock, low economic pressure to fish, and a healthy ecosystem. The main objectives include maintaining/growing the stock, cautious landings, and maintaining ecosystem healthy. A summary figure is presented in Figure 7 with individual results for TEP (Figure 25), SSB (Figure 26), exploitable biomass (Figure 27), annual landings (Figure 28), cumulative landings (Figure 29), and fishing mortality (Figure 30) presented in Appendix 2.

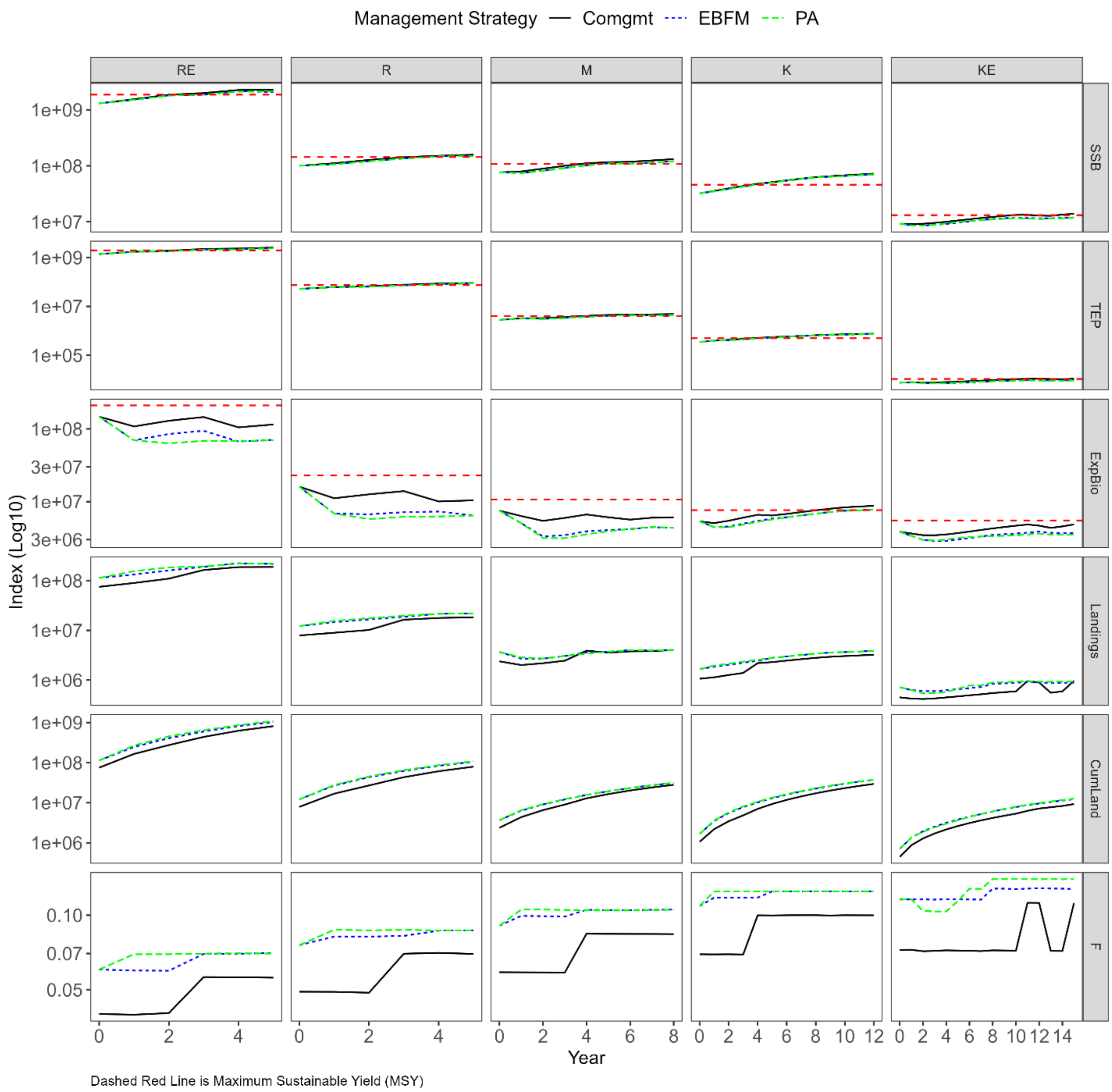

Figure 7 **Simulation results for Scenario 2. Columns represent populations RE to KE. Rows represent performance metrics including spawning stock biomass (SSB), total egg production (TEP), exploitable biomass (ExpBio), landings, cumulative landings (CumLand), and fishing mortality (F). Blue, black and green lines are individual management strategy (co-management, ecosystem-based fisheries management, and precautionary approach) results and the red dashed line is maximum sustainable yield for SSB, TEP, and ExpBio.**

Following the simulation for Scenario Two, TEP was consistently highest for stocks managed by the co-management strategy, while PA and EBFM were relatively similar. At the moderate health stock level, all populations began at 70% MSY. All resources apart from the KE stock reached MSY after a few years (three to five). Similar to Scenario One, there was an increase in TEP in year one, followed by slow growth in subsequent years (RE, R) or a short decline (M, KE) followed by growth. The KE population took the longest to reach MSY levels, and in the case of the PA and EBFM simulations, reaching 86% and 88% of MSY respectively.

SSB followed a similar trend, reaching MSY levels after a few years with the exception again of the KE population. TEP and SSB growth slowed in the PA simulation after the population entered the "healthy" stage in year 8, increasing the prescribed harvest rate. Similarly, in the EBFM simulation, the ecosystem index reached 4.0 in year 8, prescribing an elevated harvest rate. SSB for the KE-PA and EBFM simulations then stabilized at year 11.

Exploitable biomass was predictably greatest for co-management, though still below MSY levels apart from the K population. Like other metrics, the PA and EBFM simulations were relatively similar. In the RE, R, and M simulations, exploitable biomass rapidly decreased in year one before falling into stable levels for the remainder of the simulation for all three management strategies. In the *K*-select populations (K and KE), there was a sharp decrease, followed by sustained growth.

Annual landings and fishing mortality were highest for the PA and EBFM simulations in most years. However, co-management landings and fishing mortality exceeded the other two strategies in some years, when the stock moved from the cautious to healthy zones in years 3 to 4 (and 10 in the KE simulations). The increase was sustained in four populations, with the exception of the KE simulation, where the population fell back to the cautious level for two years before finishing in a healthy state.

Scenario Three

Scenario Three was defined by poor stock and ecosystem health and moderate economic pressure to fish. The main objectives included growing the stock, moderate landings, and promoting ecosystem health. A summary figure is presented in Figure 8 with individual results for TEP (Figure 31), SSB (Figure 32), exploitable biomass (Figure 33), annual landings (Figure 34), cumulative landings (Figure 35), and fishing mortality (Figure 36) presented in Appendix 2.

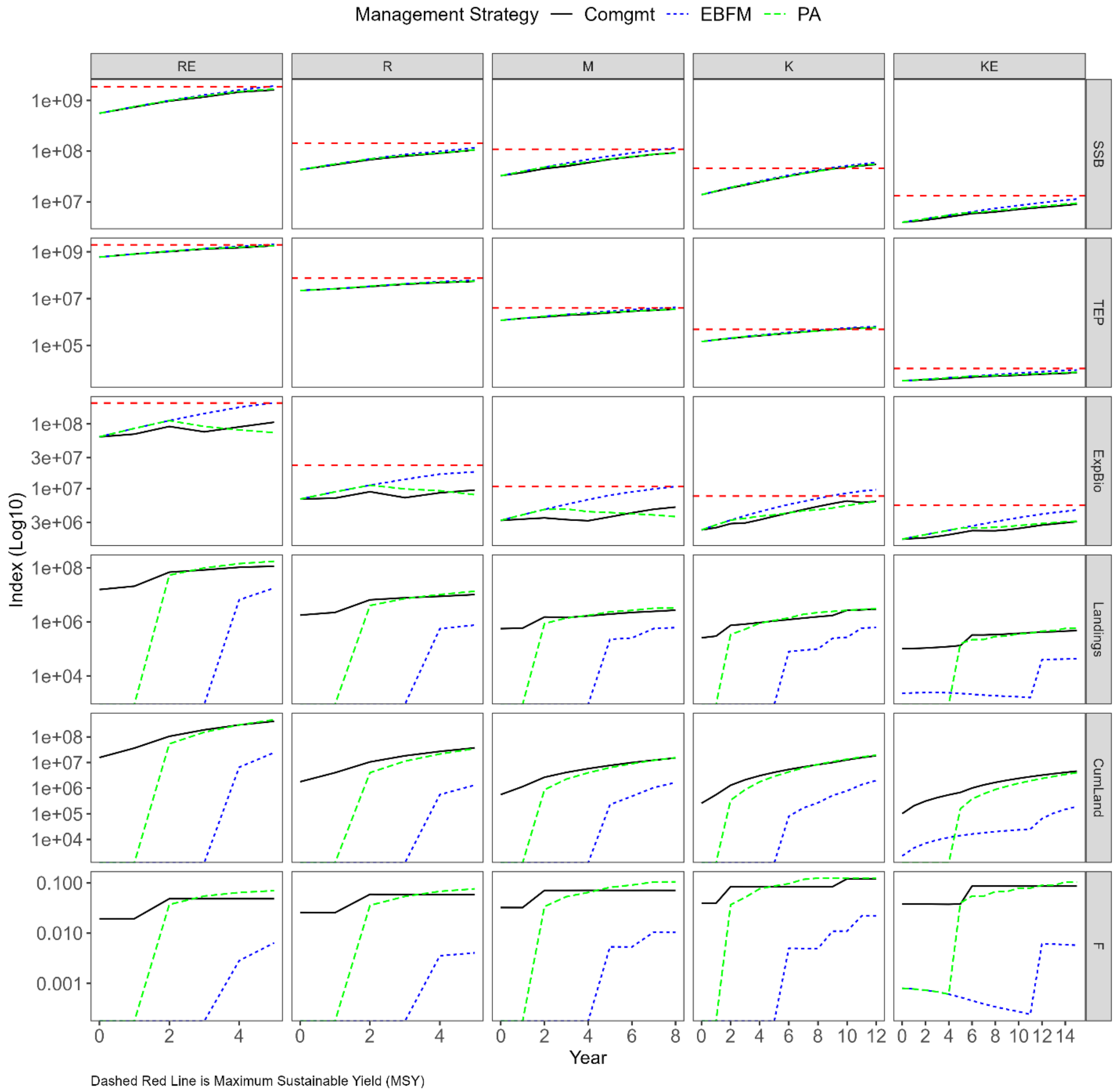

Figure 8 **Simulation results for Scenario 2. Columns represent populations RE to KE. Rows represent performance metrics including spawning stock biomass (SSB), total egg production (TEP), exploitable biomass (ExpBio), landings, cumulative landings (CumLand), and fishing mortality (F). Blue, black and green lines are individual management strategy (co-management, ecosystem-based fisheries management, and precautionary approach) results and the red dashed line is maximum sustainable yield for SSB, TEP, and ExpBio.**

Following the simulation for Scenario Three, the EBFM consistently had the highest TEP and SSB remaining. PA and co-management simulations were relatively similar, with the PA strategy ending slightly higher in each of the five populations. TEP and SSB increased annually across all five populations, with growth slowing under the PA and co-management strategies once the population entered cautious (PA) or moderate (co-management) zones. The RE population rebounded the fastest, reaching the healthy zone under the PA strategy by the end of the simulation. Surprisingly, the K population surpassed MSY by year 10 of the simulation for all three management strategies, with EBFM in year 9 and PA and co-management in year 10. Other populations were on track to reach MSY and likely would have done so with increased simulation time.

Exploitable biomass in the EBFM simulations reached or nearly exceeded MSY levels in all five populations. Under the PA strategy, there was a short increase in exploitable biomass for 2 to 5 years until the population entered the cautious zone and even the healthy zone for the RE simulation. Exploitable biomass for the co-management simulations was routinely below the other two management strategies, except for the end of the RE, R and M simulations, where it exceeded the PA strategy. This was largely due to increased removal rates prescribed by the PA strategy as the populations were thought to withstand elevated harvest.

The increased removal rates in the PA strategy resulted in the highest annual landings and fishing mortality rates by the end of the simulation for all five populations. Co-management sustained increased landings and removal rates at the start of the simulation; however, it was eventually passed by the PA strategy in all five populations. Fishing at the start of the simulation under the co-management strategy, while the PA simulations were in a closed fishery, resulted in an early lead in landings. Though the PA simulations gained ground towards the end of the simulation, cumulative landings were highest for the PA strategy in only three of the populations (RE, M, and K). Co-management cumulative landings were greatest in the R and KE populations. EBFM was regularly the lowest in landings and fishing mortality rate; however, the strategy had the greatest effect on ecosystem health, with an average index increase of 0.72 following simulations.

5.3 Discussion

An MSE of varying reproductive and management strategies demonstrated that fisheries management tools and objectives need to align with the reproductive strategy of the stock being managed. Populations with *r*-selected traits can expand quickly, rebounding from low stock status in a relatively low number of years; however, harvest can be entirely dependent on one or two age classes, reducing the resiliency of the fishery to environmental change and perturbations. Populations that have *K*-selected traits, such as a later age-at-maturity, low fecundity, and long-life spans rebound much slower; however, harvest can be distributed across several age classes, increasing fishery resiliency.

Scenario One

Overall, the EBFM strategy was the most cautious in Scenario One, leaving the stock in a state well above MSY in all populations at the cost of reduced landings. The PA followed in most simulations, and the co-management strategy left the stock in the worst state, albeit with the highest landings. Given the economic status of Scenario One, there was pressure to produce high landings, which was achieved by the co-management strategy.

Predictably, with the high economic pressure to fish, the co-management approach had the highest landings and fishing mortality. Assuming some stakeholders in the collaborative decision-making process would successfully drive for increased harvest rates, the objective of producing high landings was achieved. However, this did come at a cost, particularly in the RE and R populations where age-at-maturity and age of first legal harvest were the same age. The result was a diminished abundance of older age classes and a fishery entirely dependent on the cohort recruited each year. There are potentially catastrophic consequences with this situation, especially given the sporadic recruitment sometimes observed in fast-growing populations. Any perturbation, such as an environmental event which drives decreased survival in age-0 fish, could potentially reduce a population to low levels, especially when coupled with the elevated harvest of exploitable biomass. Overexploitation of an age class and the reduction in older, larger individuals could also reduce reproductive output, potentially causing unintended biological consequences (Baker et al. 2022; Griffeth et al. 2023).

Under Scenario One, the PA arguably was the best strategy for balancing management objectives of maintaining stock health, producing high landings, and maintaining and growing ecosystem health. Though the co-management approach produced the highest landings, the strategy also led to high exploitation rates and produced precarious situations in the *r*-select populations (RE and R), which were susceptible to collapse. Conversely, the EBFM strategy left the stock in the greatest shape; however, landings were substantially lower than the other two strategies. Ultimately, it could be argued that the PA and EBFM underfished as TEP and SSB were well above MSY levels. However, caution should be exercised so as not to fall into a similar situation as the co-management strategy, where much of the harvest in the RE simulation was dependent on incoming recruits.

Scenario Two

Overall, the co-management strategy was the most cautious in Scenario Two, leaving the stock in the best state of the three management strategies. With a lack of economic pressure to fish, the idea was to limit harvest to continue to grow the stock, which was successful. Though, like Scenario One, co-management, and in some cases the PA and EBFM strategies, may have been underfishing once the stock recovered to a healthy status. Of the three scenarios, EBFM harvest levels were highest in Scenario Two, given the moderate stock health and healthy ecosystem.

In all five populations, under the PA approach, the stock eventually entered the healthy zone and prescribed elevated harvest rates. Predictably, due to *K*-selected traits of slow growth and later age-at-maturity, the KE population was the slowest growing. This was further dampened by reaching the healthy status in both the PA and EBFM simulations, resulting in elevated removal rates. The growth observed here did, however, fit the overall objective of growing and maintaining the stock.

Scenario Three

Overall, the EBFM strategy was the most cautious in Scenario Three, leaving the stock in the best state of the three management strategies but at the trade-off of the lowest landings. Ultimately, this did benefit ecosystem health in the context of our metrics; however, landings were substantially lower in a period where economic pressure to harvest fish was moderate.

The PA was arguably the most successful management strategy for balancing stock, economic, and ecological objectives. TEP and SSB were relatively similar between PA and co-management, and exploitable biomass was slightly lower for the PA strategy; however, the trade-off was elevated landings towards the end of the simulation. Co-management HCRs routinely prescribed increased harvest rates at the start of the simulations to sustain some level of landings, which slowed the growth of exploitable biomass. The benefit of a single closed fishing season had obvious benefits for the PA simulations, as populations were able to get a jump on the growth of the stock leading to higher harvest rates later in the simulation. This was potentially problematic in the slower-growing KE population, which had no fishing for four years before harvest was prescribed. Though annual landings eventually were higher at the end of the simulation for the PA strategy, cumulative landings for the entire simulation were ultimately higher for the co-management strategy.

Comparison of Management Strategies

The EBFM strategy was overall the most cautious, apart from Scenario Two, where there was low economic pressure to fish. Though potentially underfishing, the strategy did have benefits, with the ecosystem index increasing the greatest across all simulations and stock health rebounding the fastest in Scenario Three. Unfortunately, the lack of landings may have dire social and economic consequences, even though the stock health has increased greatly.

Under the EBFM strategy, it was assumed that the other four resources remained at the initial level of MSY (i.e., poor, moderate, healthy). Another, and potentially more realistic scenario, would prescribe the EBFM strategy for the five resources simultaneously. Due to the model structure and operating time, this scenario was not undertaken. Additionally, increased ecosystem health and index would result in higher removal rates over time. This benefit is not immediately obvious in this simulation and could lead to more positive social and economic outcomes under the EBFM strategy. Species interactions such as predator-prey, fishery-related (e.g., by-catch), or environmental challenges such as habitat could have also been included in this analysis, though at the consequence of being overly complex.

Overall, simulations under the PA strategy were arguably the most successful at balancing objectives and accepting trade-offs in stock, economic, and ecosystem health. In Scenario One, simulations under the PA strategy resulted in slightly fewer landings than the co-management approach; however, the stock was left in better shape for sustainable fishing. The key example is the populations with *r*-select traits and a lack of reliance on a single cohort, which was observed in the co-management strategy.

One of the limitations of the PA is a short-term loss in yield due to reduced harvest as a mitigation for uncertainty in stock health (Mildenberger et al. 2022). This effect can be exacerbated in short-lived, high reproductive output species (Mildenberger et al. 2022) and was observed in the simulation, particularly in Scenario One, with a healthy stock and high pressure to fish. In simulations of RE and R populations, the co-management strategy exceeded PA landings, though both populations were technically at a state greater than MSY. As previously discussed, this led to a potential issue in the co-management simulation, with harvest being entirely driven by recruitment. In contrast, harvest under the PA was more balanced, with over seven times as many age-2 fish and over 24 times as many age-3 fish.

The co-management strategy had varying results, from overexploitation in scenario one, with high economic pressure to fish, to under-exploitation in Scenario Two, where there was low economic pressure. The key benefit, and also the challenge, of the co-management strategy is that all stakeholders are at the same table in the decision-making process (Linke and Bruckmeier 2015).

Obviously, that is difficult to add to a simulation analysis; however, competing objectives such as stock and economic health could lead to similar scenarios.

The simulation only evaluated the outcome of a single-acting management strategy. In reality, many fisheries are managed using a combination of strategies and there are often linkages between them (Cucuzza et al. 2021). In Canada, recent amendments to the federal *Fisheries Act* now require concomitantly bringing in concepts of all three management strategies examined in this study to the decision-making process.

Comparison of Reproductive Strategies

Populations with *r*-select traits are fast growth, have high mortality rates, high fecundity, and early age-at-maturity. In the management of these fisheries, HCRs and decision-making must react quickly. For example, in the RE simulation in Scenario One under the co-management strategy, the fishery was entirely dependent on a single age class contributing the majority of the landings. In a true co-management situation, differences between stakeholders in how the fishery should be managed could lead to a slower response and sustained overexploitation (Jentoft and McCay, 1995). The delay could harm the fishery, as an environmental event such as warming ocean temperatures reducing age-0 survival coupled with exploitation of age-1 fish, could collapse the fishery. Such populations with extreme *r*-select traits can be highly variable due to environmental conditions typically driving recruitment (Sanchirico and Essington 2021), which was not incorporated into this analysis.

Populations with *K*-selected traits exhibit slow growth, have low mortality rates, low fecundity, and a later age-at-maturity. In scenarios with low stock health, it can take years, or even decades to recover the fishery. For example, in Scenario Three with poor initial stock health, the KE population had a fishing closure for four years before TEP was greater than 40% of MSY. In the other populations, this level was reached after one to two years. Though the co-management strategy was able to sustain minimal harvest during this period, the PA strategy ultimately would have been better in the long-term, exceeding co-management in landings once individuals surviving the early closure years reached maturity. Unfortunately, this delay can be detrimental for social and economic objectives.

Study Limitations

Though the idea of *r*- and *K*-selection is useful for basic analyses, some species carry traits from both overarching groups (e.g., sturgeon, Daskin and Tilman 2022) and there is a wealth of species-specific information that can be used during the decision-making process (Reznick et al. 2002). Though most fishes tend to have *r*-select traits related to high fecundity and lack of parental care, there are many species that are long-lived, with slow growth and later age-at-maturity. It is obvious that simulating the KE population and, to a lesser extent, the K and M populations can affect recovery efforts and time. It can take a generation to see the effect of a management change, which can take nearly a decade for species with *K*-select traits versus fast-growing species that can potentially recover within a short period of time. Ultimately, the decision-making must consider the species' biology and life history when developing HCRs for a fishery.

Natural and fishing mortality were considered independent for this simulation, resulting in a fishing mortality rate that in some cases was not high enough to achieve stable fishing at MSY. For example, in the RE simulation, with fishing mortality maxed out and the exploitable biomass completely removed annually, the population still grew 1 to 2% per year. Since the model removes fish post-spawning, fish in their first year of maturity were still able to spawn and produce the next generation prior to harvest.

A multi-indicator approach could potentially identify signals in the fishery that pose future problems, such as a fishery entirely dependent on incoming recruits. Rather than relying on pre-harvest metrics, such as this simulation with TEP, some measure of exploitable biomass from survey data could be beneficial. This study used a simple, single indicator for applying the PA framework. Though this strategy is widely accepted and applied today, there are inherent problems (Mullowney and Baker 2023). SSB is a popular indicator for the PA; however, only using biomass could lead to mismanagement. There are many facets to a stock assessment, and determining HCRs for a single metric, such as biomass, could blind managers to other potential indicators such as low recruitment (Mullowney and Baker 2023). The multi-indicator system could bring other metrics (e.g., size, predator abundance) that factor into the decision rule (Mullowney and Baker 2023). This strategy could be taken a step further with the development of indicator-specific HCRs (Mullowney and Baker 2023).

Finally, the closed-loop simulation was static, with no variation in economic pressure, no natural variability of recruitment, and known population sizes annually. In reality, social pressures and environmental stochasticity will lead to variation in socioeconomic objectives and biological parameters, such as natural mortality. Future examination would benefit from the inclusion of such variation to strengthen model predictions and to assess outcomes.

Conclusion

Fisheries management must balance the well-being of biological and socioeconomic aspects of exploiting fisheries resources. Aligning reproductive and fisheries management strategies to address these needs can be highly complex. The benefit of MSE is the closed-loop simulation of stocks under various fisheries management strategies and resulting HCRs to assess trade-offs, regardless of reproductive strategy. Even with the best scientific and harvester knowledge of a fishery, uncertainty still exists for stocks ranging from *r*- to *K*-selected traits. Uncertainty in what effect a bad spawning year may have on an overexploited stock with *r*-selected traits, how long it may take to recover a population with *K*-select traits and if the economy can wait for the population to recover. Based on this simulation and the level of uncertainty that exists in fisheries management, I purport that PA principles should be incorporated into the decision-making process, including conservative HCRs considering scientific advice and survey data. Canadian fisheries management strives to incorporate all three strategies studied here, and all have their own benefits if used correctly; however, the PA principle may be the key to successful fisheries management in best balancing biological and socio-economic well-being in prosecuting fisheries on varying stocks with varying reproductive strategies.

6.0 LINKAGES TO REAL-WORLD FISHERIES

6.1 Precautionary Approach

An interesting example of the successful implementation of the PA to fisheries management is the British Columbian Green Sea Urchin *(Strongylocentrotus droebachiensis)* fishery (Perry et al. 2002b). The urchin reaches maturity in 2 to 4 years, at approximately 25 to 45 mm in test diameter (TD; Perry et al. 2002b). Individuals greater than 55 mm TD can be legally harvested by divers, who handpick urchins in suitable substrates (Perry et al. 2002b). The fishery rapidly expanded in the late-1980s, early 1990s leading to a near collapse by 1995 (Perry et al. 1998). In the early 1990s, a PA framework was implemented and featured numerous scientific studies, including the first stock assessment in 1995, and management actions based on the results of these studies (Perry et al. 1998; Perry et al. 2002b). Such management actions included adjusting the season, spatial management tools, and adjusting biological reference points (Perry et al. 2002b). Following the original scientific studies and adjusting management tools in response to scientific advice, the fishery stabilized between 1996 and 2000, with CPUE increasing annually (Perry et al. 2002b). Since Perry et al.'s (2002b) work, urchin abundance and CPUE increased rapidly, with both metrics being nearly double of 2000 values in 2019 (DFO 2021). The sea urchin would align most closely with the RE and R populations in the simulation and the results are similar between this report's findings and Perry et al.'s (2022b) work. Both the simulation and case study highlight that species with *r*-select traits, specifically an early age-at-maturity, do have the potential to recover quicky. That being said, the same species can also be quickly overexploited if not managed correctly.

In the poor-health stock scenario under the PA strategy, some stocks went years without any harvest to grow the stock into at least the critical zone before any removals are prescribed. This reaction can result in short-term negative socioeconomic consequences with a lack of harvest, but as seen in the simulation, cumulative landings over a longer period can exceed those of other strategies that continued fishing in addition to the resource being healthier. This scenario of short-term reduced landings for potentially long-term benefits undoubtedly creates a challenging situation for resource management. Similar to scientific surveys signifying poor resource health, uncertainty in data, or lack thereof, can also drive reduced harvest in PA management systems

(Calderwood and Ulmer 2023). For example, arguably at the largest possible social-spatial scale possible, under the Agreement to Prevent Unregulated High Seas Fisheries in the Central Arctic Ocean, multiple countries and the European Union have agreed not to expand commercial fishing into the north until better data are available to inform decision-making and to install the PA strategy into future exploitation of the Arctic (Calderwood and Ulmer 2023). The idea is to reverse past mistakes of rapid fishery expansion with little scientific knowledge by collecting biotic and abiotic data relevant to these northern fisheries to inform management decisions before any fish are exploited (Calderwood and Ulmer 2023). Though different than this simulation, where overexploitation creates poor stock health, the benefits of halting harvest for a short period should align with the simulation results of more fish and more landings.

6.2 Co-management

Though not a full co-management approach (i.e., DFO maintains final decision-making power), fisheries management in Canada does maintain a high level of collaboration and stakeholder engagement. Management decisions occur after the dissemination of various information such as biological data, stakeholder interests, socio-economic outlook, and scientific advice through data collected during scientific assessments (Hamelin et al. 2023). Following scientific assessments, stakeholders in the fishery are engaged to facilitate recommendations for fisheries decisions (Soomai 2017). The strategy has been broadly in place since the collapse of Atlantic groundfish stocks in the early 1990s (Linke and Bruckmeier 2014; DFO 2019; Mullowney et al. 2020) and continues to adapt, most recently with the modernized Fisheries Act (R.S.C., 1985, c, F-14).

Ultimately, it is difficult to install the positives of co-management into this MSE. For example, Maine's inshore scallop fishery installed a co-management approach between managers, scientists, and harvesters that resulted in a highly successful fishery, but would be difficult to include in the projections (Schick et al. 2005; Cucuzza et al. 2021). Scallop landings were low throughout the mid-2000s, and the Maine Department of Marine Resources sought suggestions on management strategies from a newly formed harvester committee (Cucuzza et al. 2021). This committee collaborated with managers and scientists to determine new management strategies to enhance the fishery (Cucuzza et al. 2021). The harvester committee suggested that three different areas be managed according to their ecological characteristics (e.g., different timing closures, harvest limits), which led to increased scallop abundance and landings (Cucuzza et al. 2021). The co-management approach of integrating local ecological knowledge ultimately resulted in a more successful and sustainable fishery; however, the key benefit of co-management in this example, the inclusion of local ecological knowledge, would be difficult to include in the broad projections.

Newfoundland and Labrador's Snow Crab (*Chionoecetes opilio*) fishery rapidly expanded in the late 1990s following the collapse of groundfish (Mullowney et al. 2020). Early in the history of the fishery, there were two key developments: the fishery expansion outpaced scientific knowledge of the Snow Crab stock, and the fisheries management structure was built on a co-management system that aimed to rectify the problems contributing to the groundfish collapse (Mullowney et al. 2020). Many stakeholders are involved in providing inputs into the decision-making process, including harvesters, managers, and scientists, with the final decision-making left to DFO (Mullowney et al. 2020). A key issue that has been identified in historic management of this fishery is quotas heavily weighted toward CPUE rather than scientific surveys, which can be in part driven by harvesters and managers (Griffeth et al. 2023). Overexploitation and hyperstability in the CPUE index have been identified in the northern range of NL Snow Crab, and in particular, assessment area 2HJ where resources are declining in abundance (Griffeth et al. 2023). Coupled with a previous lack of biological knowledge regarding the species, high exploitation rates of large Snow Crab have ultimately caused negative biological consequences (Mullowney and Baker 2021; Griffeth et al. 2023). As seen in the RE simulation under a co-management approach, fisheries can become overly reliant on one or two age classes, which can result in reproductive consequences. In the case of NL Snow Crab, most of the older males were removed which drove a decline in male size-at-maturity (Mullowney and Baker 2021; Griffeth et al. 2023).

In Scenario 1, socioeconomic objectives were prioritized in the co-management simulation, resulting in diminished stock health. Similarly, in the case of Snow Crab assessment area 2HJ, management changes implemented to improve catch rates ultimately backfired and led to reduced resource abundance and negative reproductive consequences (Griffeth et al., 2023). In contrast, when economic pressure was low or moderate in the simulation, stock health was comparable to the other management strategies. Ultimately, co-management might be best utilized in conjunction with PA or EBFM principles. Maintaining strict reference points and corresponding HCRs that define upper bounds on acceptable harvest rates would be key to maintaining biological health while the fishery is under periods of high economic stress. Conversely, the simulations showed co-managed resources can be under-exploited in cases where economic pressures to fish are low. Overall, it appears the added element of focus on economics in decision-making in co-management systems adds a level of complexity that can affect resource outcomes, and that benefits of co-management may be enabled to most clearly resonate if systems operate within context of defined lower and upper limits on exploitation.

6.3 Ecosystem-Based Fisheries Management

Most of the literature and in fact application of management systems related to EBFM is at the conceptual stage, with a wealth of literature on simulation analysis and conceptual framework models (Fletcher et al. 2010; Gruss et al. 2017; Sanchirico and Essington 2021) but a lack of published literature on the true implementation and success of EBFM (Pitcher et al. 2009). Regardless, incorporating multi-species management and considering interactions between fisheries under an EBFM approach can potentially improve all resources involved. One example is the shrimp and Red Snapper (*Lutjanus campechanus*) fisheries in the Gulf of Mexico.

The Gulf of Mexico shrimp fishery is highly valuable and primarily operates as a trawl fishery (Gallaway et al. 2017). Unfortunately, trawling can produce high by-catch including juvenile Red Snapper, a resource being rebuilt through different harvest management strategies (Gallaway et al. 2017). In the 1990's, early studies on shrimp and Red Snapper fishery dynamics suggested that by-catch mortality of juvenile red snappers was though to contribute to 80% of the total mortality (Gallaway et al. 2017). Rebuilding plans for Red Snapper in the 2000's focused on reducing by-catch mortality and aimed to reduce shrimp trawling effort, which was simplified with high vessel losses in the fishery due to economic pressures (Gallaway et al. 2017). Similar to this simulation analysis, though the resource may be in a healthy or moderate healthy state, prescribed fishing mortality can potentially be a tool in the rebuilding of other stocks. In the Gulf of Mexico case, reduced effort was coupled with increased surveys of Red Snapper in the region, which produced unique results. First, Red Snapper abundance did not immediately rebound or even produce an observable trend and secondly, with a reduced number of vessels, catch rates skyrocketed (Gallaway et al. 2017). Surveys aimed at assessing juvenile Red Snapper survival suggested that natural mortality was higher than originally thought, with by-catch only contributing approximately 4% of total mortality (Gallaway et al. 2017). Though an unintended consequence, examining the management of shrimp and Red Snapper through a multi-species (EBFM) approach improved knowledge of both fisheries. The information can then be used to inform future management and rebuilding strategies for the species.

7.0 CONCLUSION

Aligning management and reproductive strategies is important in developing sustainable fisheries. As demonstrated here, different life history types will have varying responses and response times to management changes. Fast-growing species with *r*-select traits can recover quickly but are susceptible to rapid decline in response to short-term environmental change and overexploitation. Long-lived species aligning with *K*-selection can take years to recover but are strengthened by multiple cohorts producing offspring.

Management strategies evaluated in this report included PA, co-management, and EBFM. Simulation analysis suggested that this paper's interpretation of the PA strategy may be the most ideal for meeting set objectives under Scenarios One, Two, and Three. Ultimately, the success of any management strategy hinges on whether it is implemented correctly. The PA strategy appears great in theory; however, if not prescribed correctly, for example lacking inclusion of scientific advice in harvest decision making, the strategy may fail. Similarly, co-management and EBFM, if implemented correctly, can lead to a sustainable fishery, though both come with inherent challenges, such as a lack of shared objectives in the case of co-management or the incorrect interpretation of complex multi-species models in EBFM. Finally, though described independently here, these three management strategies are often linked, and fisheries are managed using a combination of approaches. In fact, the Canadian *Fisheries Act* now stipulates that all three management strategies be implemented in the management of Canadian fisheries.

In other regions of the world, simply shifting to a PA, co-managed, or ecosystem-friendly style of fisheries management may be the key to improving stocks, particularly for those that are in underdeveloped nations and are data deficient (Hilborn and Ovando 2014). The implementation of any fisheries management system (whichever form) and some level of assessment would likely increase the success of many fisheries globally, as many remain unmanaged (Hilborn and Ovando 2014).

Regardless, the MSE presented here is an excellent example of examining the outcomes of implementing various management strategies on stocks with varying reproductive traits. Though this report leaves many aspects of reproductive biology and harvest management tools unexplored, the findings can still contribute to the global push for sustainable fisheries.

8.0 REFERENCES

Baker, K.D., Mullowney, D.R.J., and Sainte-Marie, B. 2022. Large males matter: Low sperm reserves in female snow crab (*Chionoecetes opilio*) off Newfoundland, Canada. *Fisheries Research* 253: 106385.

Bopp, J., Herbst, S.J., Brenden, T.O., Wehrly, K., and Hessenauer, J.M. 2023. Biotic and abiotic factors that influence Walleye recruitment in stocked lakes in Michigan. *North American Journal of Fisheries Management* 43(6): 1673-1686.

Branch, T. A., Hilborn, R., Fay, A. C., Flynn, L., Griffiths, J., Marshall, K. N., Randall, J. K., Scheurell, J. M., Ward, E. J., & Young, M. 2006. Fleet dynamics and fisherman behavior: Lessons for fisheries managers. *Canadian Journal of Fisheries and Aquatic Sciences*, 63(7), 1647–1668.

Bruder, A., Salis, R.K., Jones, P.E., and Matthaei, C.D. 2017. Biotic interactions modify multiple-stressor effects on juvenile brown trout in an experimental stream food web. *Global Change Biology* 23: 3882-3894.

Boyce, D.G., Fuller, S., Karbowski, C., Schleit, K., and Worm, B. 2021. Leading or lagging: How well are climate considerations being incorporated into Canadian fisheries management? *Canadian Journal of Fisheries and Aquatic Science* 78: 1120-1129.

Calderwood, C. and Ulmer, F.A. 2023. The Central Arctic Ocean fisheries moratorium: a rare example of the precautionary approach principle in fisheries management. *Polar Record* 59: 1-14.

Caswell, H. 2001. Matrix population models: construction, analysis, and interpretation. Sinauer Associates, Sunderland, Mass.

COSEWIC. 2017. COSEWIC assessment and status report on the Atlantic Walrus Odobenus rosmarus rosmarus, High Arctic population, Central-Low Arctic population and Nova Scotia-Newfoundland-Gulf of St. Lawrence populationin Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. xxi + 89 pp.

Cucuzza, M., Stoll, J.S., and Leslie, H.M. 2021. Evaluating the theoretical and practical linkages between ecosystem-based fisheries management and fisheries co-management. *Marine Policy* 126: 104390.

Daskin, J.H. and Tilman, A.R. 2022. Alternative restocking strategy could reverse declines of a critically endangered sturgeon. *Environmental Conservation* 49: 53-58.

Davis, N.A. Evaluating collaborative fisheries management planning: A Canadian case study. *Marine Policy* 32: 867-876.

DFO. 2006. A Harvest Strategy Compliant with the Precautionary Approach. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2006/023.

DFO. 2009. A fishery decision-making framework incorporating the precautionary approach. Available from: https://www.dfo-mpo.gc.ca/reports-rapports/regs/sff-cpd/precaution-eng.htm

DFO. 2019. Snow Crab Integrated Management Plan. http://www.dfo-mpo.gc.ca/fisheries-peches/ifmp-gmp/snow-crab-neige/2019/index-eng.htm.

DFO. 2021. Stock Status Update and Harvest Options for the Green Sea Urchin *(Strongylocentrotus droebachiensis)* fishery in British Columbia, 2021-2024. DFO Can. Sci. Advis. Sec. Sci. Resp. 2021/036.

DFO. 2023. Science Advice on Guidance for Limit Reference Points under the Fish Stock Provision. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2023/009.

Durant, J.M. and Hjermann, D.O. 2017. Age-structure, harvesting and climate effects on population growth of Arcto-boreal fish stocks. *Marine Ecology Progress Series* 577: 177-188.

FAO. 1995. Precautionary Approach to Fisheries, part 1: guidelines on the precautionary approach to capture fisheries and species introductions. Food and Agriculture Organization of the United Nations, Fisheries Technical Paper 350, part 1: 52 p.

FAO. 1996. Precautionary approach to capture fisheries and species introductions. Food and Agriculture Organization of the United Nations, Fisheries Technical Paper 2: 54 p.

Fletcher, W.J., Shaw, J., Metcalf, S.J., and Gaughan, D.J. 2010. An Ecosystem Based Fisheries Management framework: the efficient, regional-level planning tool for management agencies. *Marine Policy* 34: 1226-1238.

Fogarty, M. and O'Brian L. 2016. Chapter 5: Egg, Larval, and Juvenile Surveys. Fish Reproductive Biology: Implications for Assessment and Management Second Edition. Edited by Jakobsen, T., Fogarty, M.J., Megrey, B.A., and Moksness, E.

Frid, A., Wilson, K.L., Walkus, J., Forrest, R.E., and Reid, M. 2023. Re-imagining the precautionary approach to make collaborative fisheries management inclusion of Indigenous Knowledge Systems. *Fish and Fisheries* 24: 940-958.

Gallaway, B.J., Gazey, W.J. and Cole, J.G. 2017. An Updated Description of the Benefits and Consequences of Red Snapper Shrimp Trawl Bycatch Management Actions in the Gulf of Mexico. *North American Journal of Fisheries Management* 2: 414-419.

Geiger, H.J., Wang, I., Malecha, P., Herbert, K., Smoker, W.W., and Gharrett, A.J. 2007. What causes variability in pink salmon family size. *Transactions of the American Fisheries Society* 136: 1688- 1698.

Griffeth, S.P., Baker, K.D., and Mullowney, D.R.J. 2023. Short-term fishery gains mask long-term resource pains: Spatial fisheries management changes promote hyperstable CPUE in Labrador snow crab Chionoecetes opilio during a period of heavy exploitation. *North American Journal of Fisheries Management* 43: 1359 – 1371.

Gruss, A., Rose, K.A., Simons, J., Ainsworth, C.H., Babcock, E.A., Chagaris, D.D., Mutsert, K.D., Froeschke, J., Himchak, P., Kaplan, I.C., O'Farrell, H., and Rejon, M.J.Z. 2017. Recommendations on the Use of Ecosystem Modeling for Informing Ecosystem Based Fisheries Management and Restoration Outcomes in the Gulf of Mexico. *Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science* 9: 281-295.

Guidetti, P. and Claudet, J. 2009. Comanagement practices enhance fisheries in marine protected areas. *Conservation Biology* 24(1): 312-318.

Hamelin, K.M., Hutchings, J.A., and Bailey, M. 2023. Look who's talking: contributions to evidence-based decision making for commercial fisheries in Atlantic Canada. *Canadian Journal of Fisheries and Aquatic Sciences* 80(2): 211-228.

Harley, S. J., Myers, R. A., & Dunn, A. (2001). Is catch--per--unit--effort proportional to abundance? *Canadian Journal of Fisheries and Aquatic Sciences*, 58(9), 1760–1772.

Hilborn, R. and Ovando, D. 2014. Food for Thought: Reflections on the success of traditional fisheries management. *ICES Journal of Marine Science* 71(5): 1040-1046.

Hines, A.H., Jivoff, P.R., Bushmann, P.J., van Montfrans, J., Reed, S.A., Wolcott, D.L., and Wolcott, T.G. 2003. Evidence for sperm limitation in the blue crab, *Callinectes sapidus*. *Bulletin of Marine Science*, 72(2): 287-310.

Howell, D., Schueller, A.M., Bentley, J.W., Buchheister, A., Chagaris, D., Cieri, M., Drew, K., Lundy, M.G., Pedreschi, D., Reid, D.G., and Townsend, H. 2021. Combining ecosystem and single-species modeling to provide ecosystem-based fisheries management advice within current management systems. *Frontiers in Marine Science* 7: 607831.

Hutchings, J.A. 2005. Life history consequences of overexploitation to population recovery in Northwest Atlantic cod (*Gadus morhua*). *Canadian Journal of Fisheries and Aquatic Science* 62: 824-832.

Izzo, L.K., Dembkowski, D., Hayden, T., Binder, T., Vandergroot, C., Hogler, S., Donofrio, M., Zorn, T., Krueger, C.C., and Isermann, D. 2023. Spawning Locations, Movements, and Potential for Stock Mixing of Walleye in Green Bay, Lake Michigan. *North American Journal of Fisheries Management* 43: 695-714.

Jude, D.J., Wang, Y., Hensler, S.R., and Janssen, J. 2013. Burbot Early Life History Strategies in the Great Lakes. *Transactions of the American Fisheries Society* 142: 1733-1745.

Karp, M.A., Brodie, S., Smith, J.A., Richerson, K., Selden, R.L., Liu, O.R., Muhling, B.A., Samhouri, J.F., Barnett, L.A.K., Hazen, E.L., Ovando, D., Fiechter, J., Jacox, M.G., and Pozo Buil, M. 2023. Projecting species distributions using fishery-dependent data. *Fish and Fisheries* 24: 71-92.

Lackman, A.R., Sereda, J., Pollock, M., Bryshun, R., Chupik, M., McCallum, K., Villeneuve, J., Bielak-Lackmann, E.S., and Clark, M.E. 2023. Bet-hedging bigmouth buffalo (*Ictiobus cyprinellus*) recruit episodically over a 127-year timeframe in Saskatchewan. *Canadian Journal of Fisheries and Aquatic Science* 80: 313-329.

Lo, N.C.H., Smith, P.E., and Takahashi, M. 2016. Chapter 5: Egg, Larval, and Juvenile Surveys. Fish Reproductive Biology: Implications for Assessment and Management Second Edition. Edited by Jakobsen, T., Fogarty, M.J., Megrey, B.A., and Moksness, E.

Link, J.S. and Browman, H.I. 2014. Integrating what? Levels of marine ecosystem-based assessment and management. *ICES Journal of Marine Science* 71(5): 1170-1173.

Link, J.S. and Marshak, A.R. 2021. Ecosystem-Based Fisheries Management: Progress, Importance, and Impacts in the United States. Oxford University Press.

Linke, S. and Bruckmeier, K. 2015. Co-management in fisheries – Experiences and changing approaches in Europe. *Ocean and Coastal Management* 104: 180-181.

Little, L.R., Punt, A.E., Dichmont, C.M., Dowling, N., Smith, D.C., Fulton, E.A., Sporcic, M., and Gorton, R.J. 2016. Decision trade-offs for cost-constrained fisheries management. *ICES Journal of Marine Science* 73(2): 494-502.

McMillan, J.R., Pess, G.R., McHenry, M.L., Moses, R., and Quinn, T.P. 2014. Documentation of Unusual, Fall Spawning by Coastal Cutthroat Trout in the Elwha River System, Washington. *Transactions of the American Fisheries Society* 143: 1605-1611.

Mildenberger, T.K., Berg, C.W., Kokkalis, A., Hordyk, A.R., Wetzel, C., Jacobsen, N.S., Punt, A.E., and Nielsen, J.S. 2022. Implementing the precautionary approach into fisheries management: Biomass reference points and uncertainty buffers. *Fish and Fisheries* 23:73-92.

Morgan, M.J. 2008. Integrating Reproductive Biology into Scientific Advice for Fisheries Management. *Journal of Northwest Atlantic Fishery Science* 41: 37-51.

Mullowney, D.R.J., Morris, C., Dawe, E., Zagorsky, I., and Goryanina, S. 2018. Dynamics of snow crab (Chionoecetes opilio) movement and migration along the Newfoundland and Labrador and Eastern Barents Sea continental shelves. *Reviews in Fish Biology and Fisheries* 28: 435-459.

Mullowney, D.R.J., Baker, K.D., Zabihi-Seissan, S., and Morris, C. 2020. Biological perspectives on complexities of fisheries co-management: A case study of Newfoundland and Labrador snow crab. *Fisheries Research* 232: 105728.

Mullowney, D.R.J. and Baker, K.D., 2021. Size-at-maturity shift in a male-only fishery: factors affecting molt-type outcomes in Newfoundland and Labrado snow crab (*Chionoecetes opilio*). ICES *Journal of Marine Science* 78: 516-533.

Mullowney, D.R.J. and Baker, K. 2023. Multi-indicator precautionary approach frameworks for crustacean fisheries. *Canadian Journal of Fisheries and Aquatic Sciences* 00: 1-14.

Nahass, T. and Arrington, D. 2023. Survivorship Curve Definition, Types, and Examples. CLEP Biology: Study Guide and Test Prep Course Chapter 15.

Oceana Canada. 2023. 7th Annual Fishery Audit. November 2023.

Paul, A. 2013. Environmental flows and recruitment of walleye (Sander vitreus) in the Peace-Athabasca Delta. *Canadian Journal of Fisheries and Aquatic Sciences 7* (2): 307-315

Perry, R.I., Waddell, B.J., Campbell, A., Hobbs, K., 1998. Review of fishery-dependent data and quota recommendations for 1995/96 for the green sea urchin fishery in British Columbia. In: Waddell, B.J., Gillespie, G.E., Walthers, L.C. (Eds.), Invertebrate Working Papers Reviewed by the Pacific Stock Assessment Review Committee (PSARC) in 1995. Part 2. Echinoderms. Can. Tech. Rep. Fish. Aquat. Sci. No. 2215, pp. 111–145.

Perry, E., Seegert, G., Vondruska, J., Lohner, T., and Lewis, R. 2002a. Modeling Possible Cooling-Water Intake Systems Impacts on Ohio River Fish Populations. *The Scientific World Journal* 2 (S1) 58-80.

Perry, R.I., Zhang, Z. and Harbo, R. 2002b. Development of the green urchin *(Strongylocentrotus droebachiensis)* Fishery in British Columbia – back from the brink using a precautionary framework. *Fisheries Research* 55: 253-266.

Pianka, E.R. 1970. On r- and K-selection. *American Naturalist* 104: 592-597.

Pinkerton, E. and John, L. 2008. Creating local management legitimacy. *Marine Policy* 32: 680-691.

Pitcher, T.J., Kalikoski, D., Short, K., Varkey, D., Pramod, G. 2009. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. *Marine Policy* 33: 223-232.

Power, M. 2007. Fish Population Bioassessment. Pages 561-624 in C.S. Guy and M.L. Brown, editors. *Analysis and Interpretation of Freshwater Fisheries Data*. American Fisheries Society, Bethesda, Maryland.

Puebla O., Sévigny J.-M., Sainte-Marie B., Brêthes J.-C., Burmeister A. D., Dawe E. G., Moriyasu M. 2008. Population genetic structure of the snow crab (Chionoecetes opilio) at the Northwest Atlantic scale. *Canadian Journal of Fisheries and Aquatic Sciences*, 65: 425–436.

Punt, A.E., Butterworth, D.S., de Moor, C.L., De Oliveira, J.A.A., and Haddon, M. 2016. Management Strategy Evaluation: best practices. *Fish and Fisheries* 17: 303-334.

R Core Team. 2021. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Radigan, W., Chvalam P., Longhenry, C., and Pegg, M. 2024. Factors affecting walleye and sauger recruitment in Lewis and Clark Lake, South Dakota 2001-2022. *Fisheries Management and Ecology* 31: e12692.

Reznick, D., Bryant, M.J., and Bashey, F. 2002. r- and K-selection revisited: the role of population regulation in life-history evolution. *Ecology* 83(6): 1509-1520.

Rose, G.A. and Kulka, D.W. 1999. Hyperaggregation of fish and fisheries: how catch-per-unit-effort increased as the northern cod (Gadus morhua) declined. *Canadian Journal of Fisheries and Aquatic Science* 56 (supplement 1): 118-127.

Roseman, E.F., Kennedy, G.W., Boase, J., Manny, B.A., Todd, T.N., and Stott, W. 2007. Evidence of Lake Whitefish Spawning in the Detroit River: Implications for Habitat and Population Recovery. *Journal of Great Lakes Research* 33: 397-406.

Sable, S.E. and Rose, K. A. 2008. A comparison of individual-based and matrix projection models for simulating yellow perch population dynamics in Onedia Lake, New York, USA. *Ecological Modelling* 215: 105-121.

Sanchirico, J.N. and Essington, T.E. 2021. Direct and ancillary benefits of ecosystem-based fisheries management in forage fish fisheries. *Ecological Applications* 31 (7): e02421.

Schick, D.F., and Scott, C.F. 2005. Marine Scallop Fishery: Monitoring and Enhancement. Final Report to the Northeast Consortium, Maine Department of Marine Resources, 2005.

Smart, J.J., Chin, A., Tobin, A.J., White, W.T., Kumasi, B., and Simpfendorfer, C.A. 2017. Stochastic demographic analyses for the silvertip shark (*Carcharhinus albimarginatus*) and the common blacktip shark (Carcharhinus limbatus) from the Indo-Pacific. *Fisheries Research* 191: 95-107.

Sinnatamby, R.N., Pinto, M.C., Johnston, F.D., Paul, A.J., Mushens, C.J., Stelfox, J.D., Ward, H.G.M., and Post, J.R. 2018. Seasonal timing of reproductive migrations in adfluvial bull trout: an assessment of sex, spawning experience, population density, and environmental factors. *Canadian Journal of Fisheries and Aquatic Science* 75: 2172-2183.

Solomon, F.N. and Ramnarine, I.W. 2007. Reproductive biology of white mullet, *Mugil curma* (Valenciennes) in the Southern Caribbean. *Fisheries Research* 88: 133-138.

Soomai, S.S. 2017. The science-policy interface in fisheries management: Insights about the influence of organizational structure and culture on information pathways. Marine Policy 81: 53-63.

Stephenson, R.L., Paul, S., Wiber, M., Angel, E., Benson, A.J., Charles, A., Chouinard, O., Clemens, M., Edwards, D., Foley, P., Jennings, L., Jones, O., Lane, D., McIsaac, J., Mussells, C., Neis, B., Nordstrom, B., Parlee, C., Pinkerton, E., Saunders, M., Squires, K., and Sumaila, U.R. 2018. Evaluating and implementing social-ecological systems: A comprehensive approach to sustainable fisheries. *Fish and Fisheries* 19: 853-873.

Swerdfager, T. and Armitage, D. 2023. Co-management at a crossroads in Canada: issues, opportunities, and emerging challenges in fisheries and marine contexts. FACETS 8: 1-10.

Szuwalski, C.S., Aydin, K., Fedewa, E.J., Garber-Yonts, B., and Litzow, M.A. 2023. The collapse of eastern Bearing Sea snow crab. *Science* 382: 306-310.

van der Lee, A.S. and Koops, M.A. 2020. Recovery Potential Modelling of Westslope Cutthroat Trout (*Oncorhynchus clarkii lewisi*) in Canada (Saskatchewan-Nelson River populations). DFO Can. Sci. Advis. Sec. Res. Doc. 2020/046 v + 26 p.

Wilson, D.C., Nielson, J.R., and Degnbol, R. (Eds.). 2003. The Fisheries Co-Management Experience. Kluwer, Dordrecht, Netherlands.

Winter, A.M. and Hutchings, J.A. 2020. Impediments to fisheries recovery in Canada: Policy and institutional constraints on developing management practices compliant with the precautionary approach. *Marine Policy* 121: 104161.

Worm, B., Hilborn, R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennings, S., Jensen, O.P., Lotze, H.K., Mace, P.M., McClanahan, T.R., Minto, C., Palumbi, S.R., Parma, A.M., Ricard, D., Rosenberg, A.A., Watson, R., and Zeller, D. 2009. Rebuilding Global Fisheries. *Science* 325: 578-585.

Yin, J., Xu, J., Xue, Y., Xu, B., Zhang, C., Li, Y., and Ren, Y. 2021. Evaluating the impacts of El Niño events on a marine bay ecosystem based on selected ecological network indicators. *Science of the Total Environment* 763: 144205.

Appendix 1. Life History Parameters and Population Structure

Table 7 **Survival rate by age. Bold indicates the first year of maturity and <u>underline</u> indicates the first year of legal harvest.**

Age	RE	R	M	K	KE
0	0.010	0.125	0.325	0.660	0.800
1	**<u>0.100</u>**	0.300	0.525	0.640	0.790
2	0.020	0.200	0.450	0.620	0.780
3	0.000	**<u>0.125</u>**	0.375	0.590	0.770
4	-	0.050	0.300	0.560	0.760
5	-	0.000	**0.225**	0.530	0.750
6	-	-	<u>0.150</u>	0.500	0.740
7	-	-	0.075	**0.470**	0.730
8	-	-	0.000	<u>0.430</u>	0.720
9	-	-	-	0.350	0.710
10	-	-	-	0.270	**0.700**
11	-	-	-	0.150	<u>0.675</u>
12	-	-	-	0.000	0.650
13	-	-	-	-	0.600
14	-	-	-	-	0.400
15	-	-	-	-	0.000

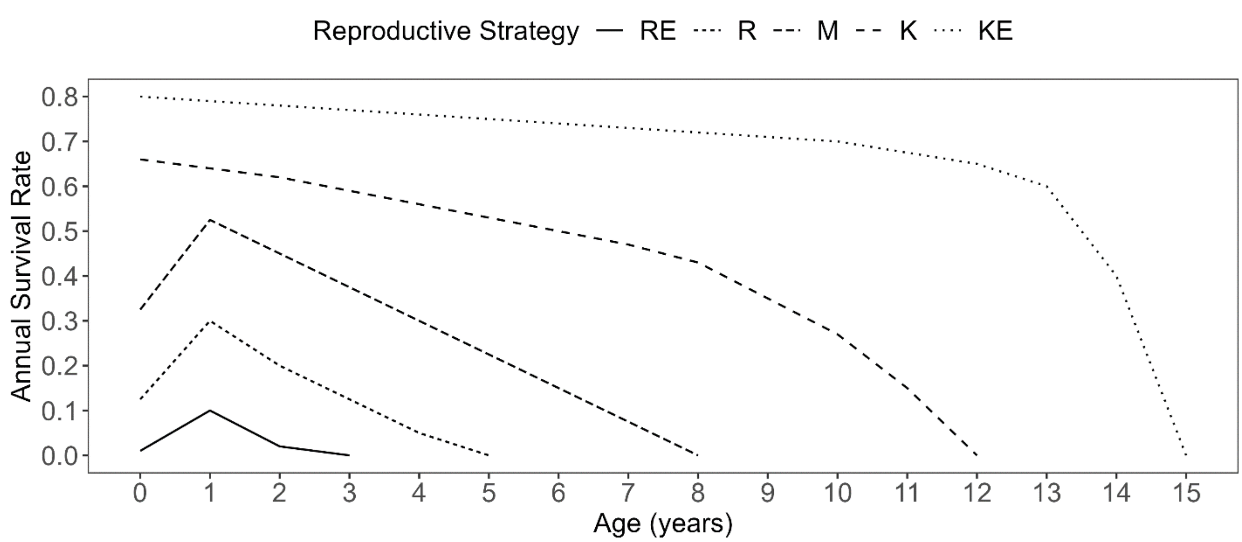

Figure 9 **Annual survival rate for each population. Align with standard Type 1, 2, and 3 survivorship curves after year 1.**

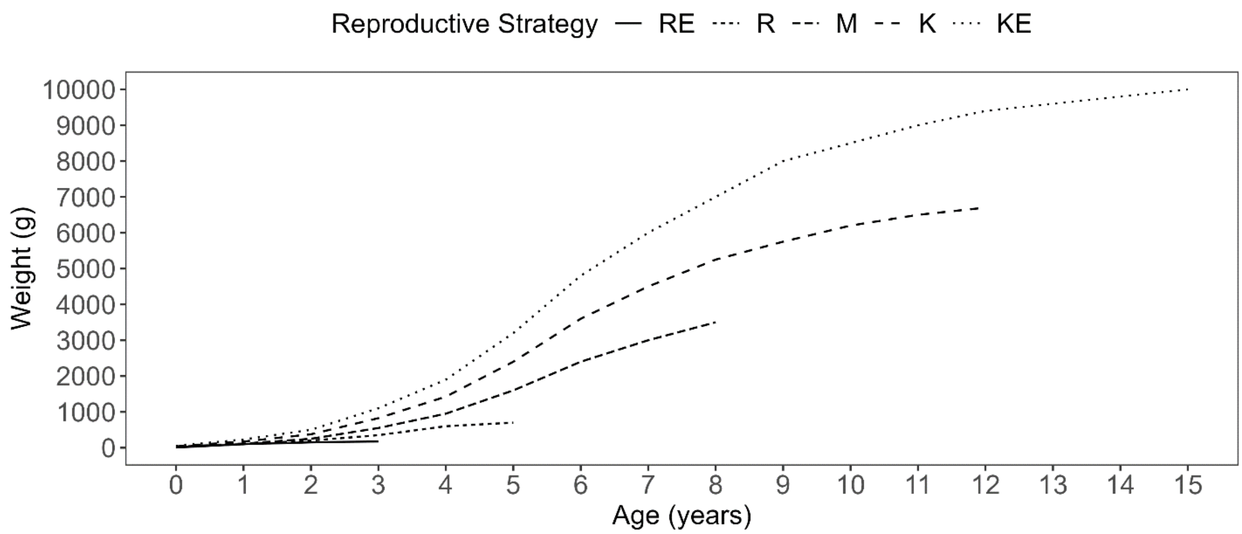

Figure 10 Weight-at-age relationship for each population.

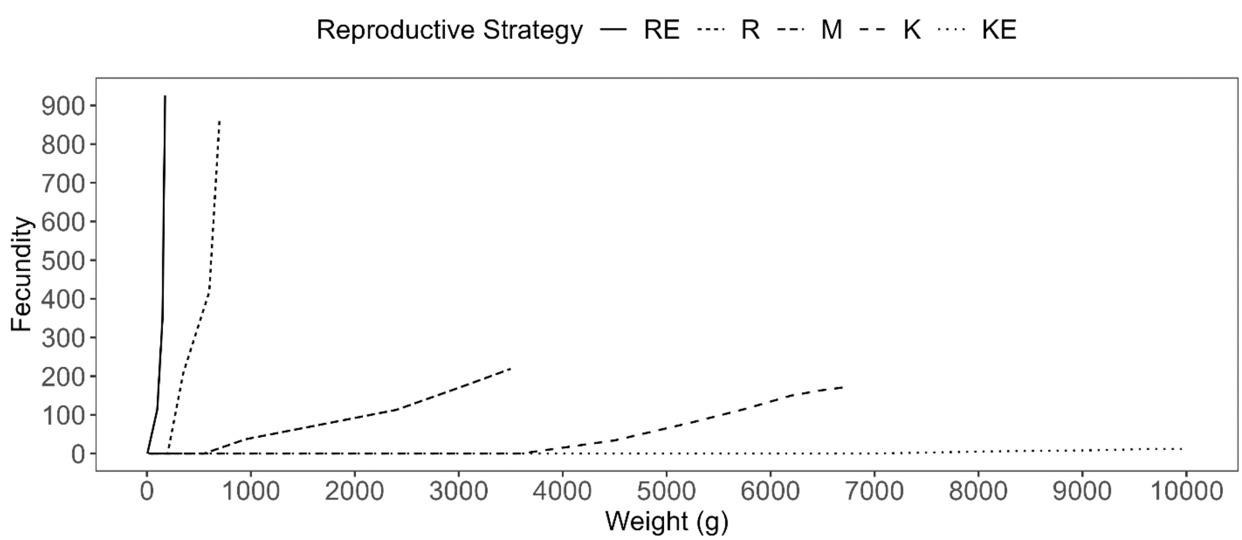

Figure 11 Fecundity-at-weight relationship for each population.

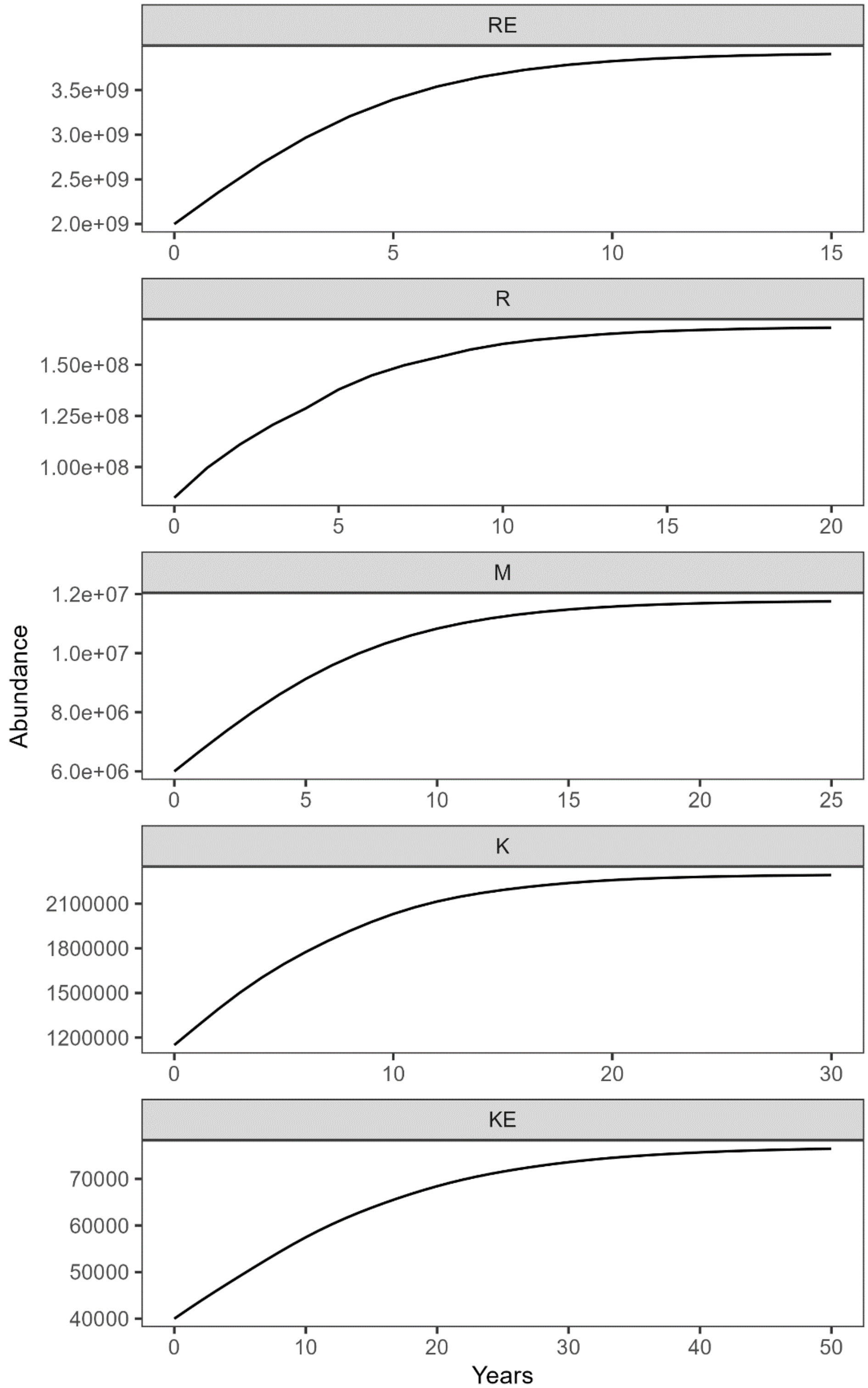

Figure 12 **Population projection from 50% of K with no harvest by population. Structured as a Beverton-Holt relationship.**

Appendix 2. Harvest Control Rules by Management Strategy

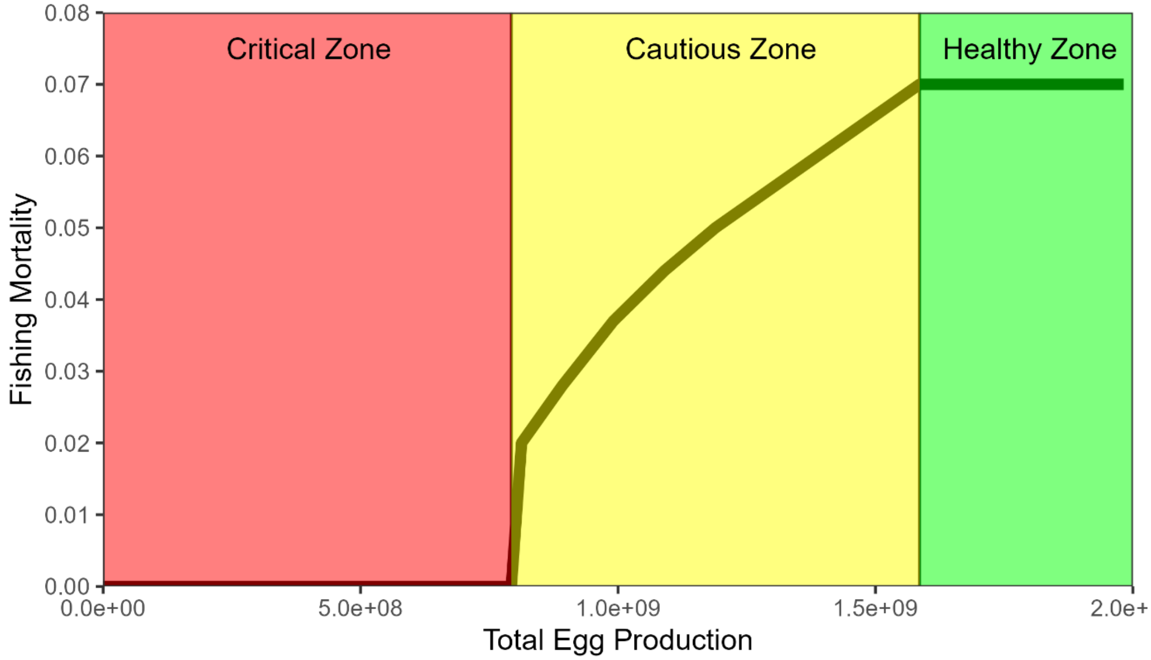

Figure 13 **Harvest control rules (fishing mortality) for the RE population under the precautionary approach strategy.**

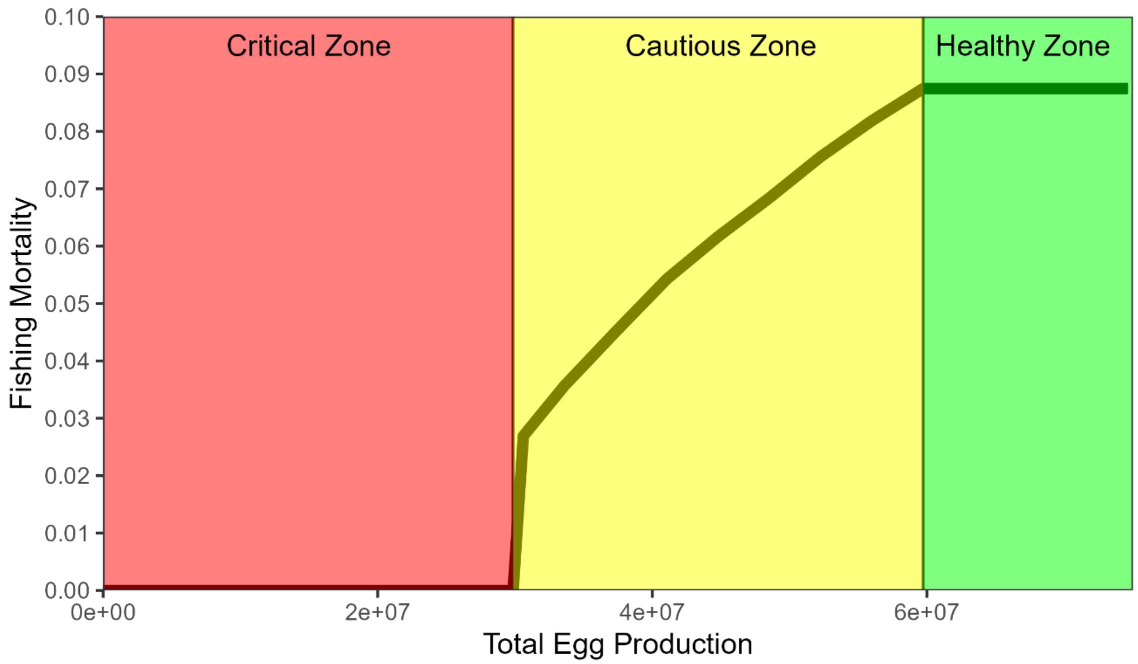

Figure 14 **Harvest control rules (fishing mortality) for the R population under the precautionary approach strategy.**

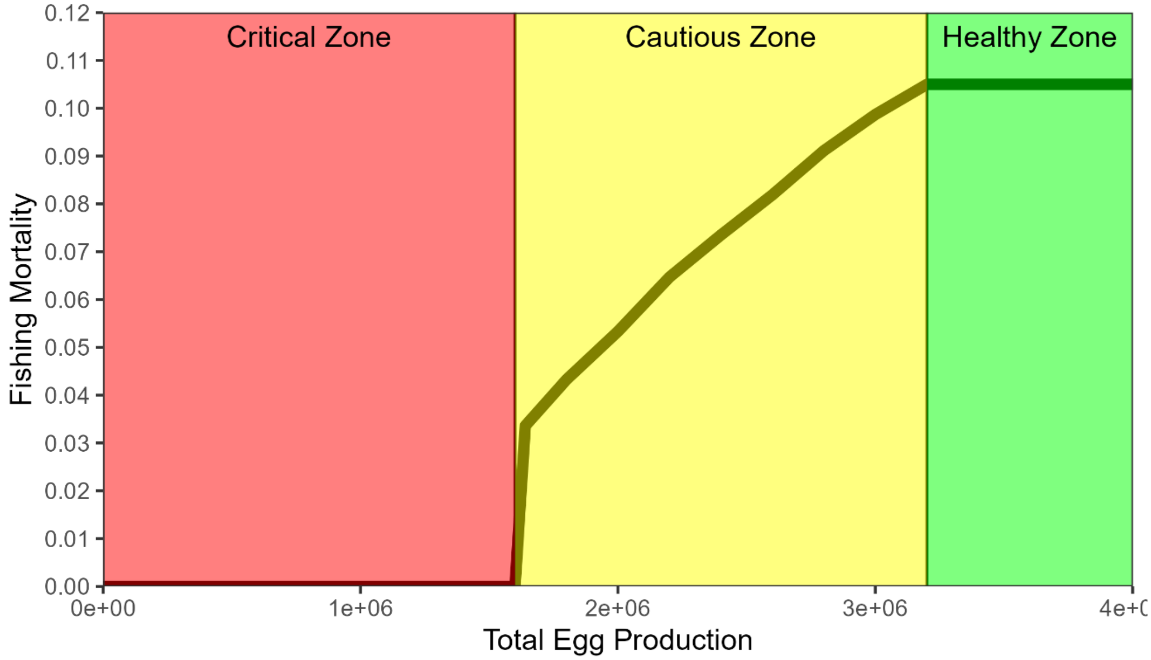

Figure 15 **Harvest control rules (fishing mortality) for the M population under the precautionary approach strategy.**

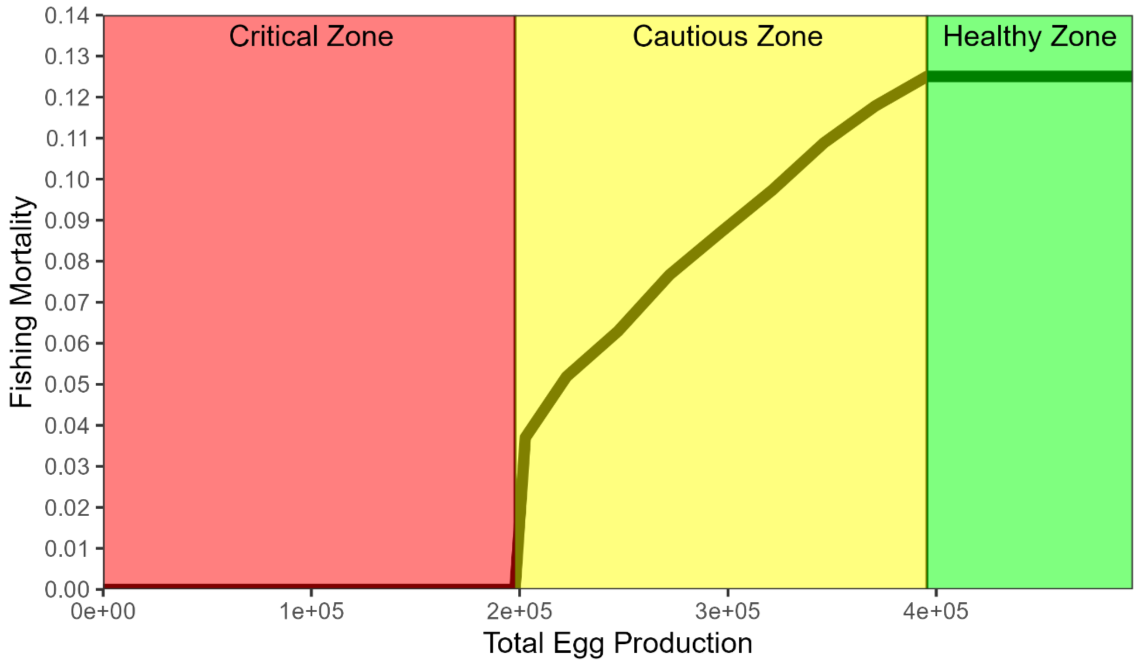

Figure 16 **Harvest control rules (fishing mortality) for the K population under the precautionary approach strategy.**

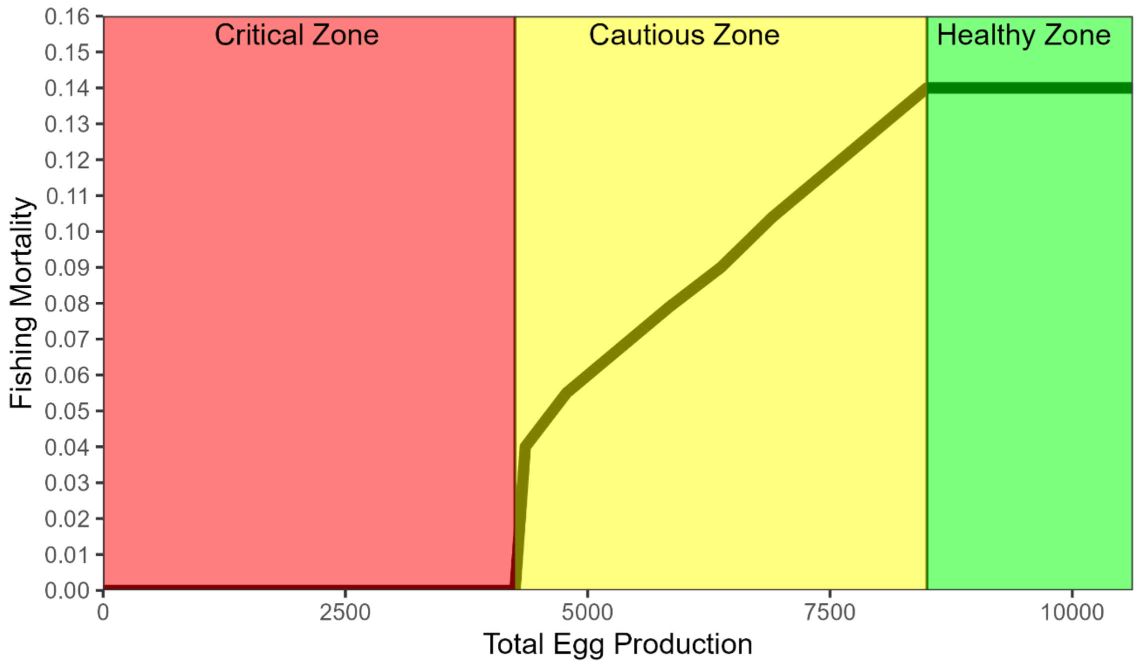

Figure 17 **Harvest control rules (fishing mortality) for the KE population under the precautionary approach strategy.**

Table 8 **Harvest control rules (fishing mortality) under the co-management strategy and various stock health, economic pressure, and ecosystem health scenarios.**

Stock	Economic	Ecosystem	Extreme R	R	Mid	K	Extreme K
Poor	High	Poor	0.0240	0.0323	0.0403	0.0492	0.0480
Poor	High	Moderate	0.0252	0.0339	0.0423	0.0517	0.0504
Poor	High	Healthy	0.0264	0.0355	0.0444	0.0541	0.0528
Poor	Moderate	Poor	0.0194	0.0261	0.0326	0.0398	0.0388
Poor	Moderate	Moderate	0.0200	0.0269	0.0336	0.0410	0.0400
Poor	Moderate	Healthy	0.0206	0.0277	0.0346	0.0422	0.0412
Poor	Low	Poor	0.0000	0.0000	0.0000	0.0000	0.0000
Poor	Low	Moderate	0.0000	0.0000	0.0000	0.0000	0.0000
Poor	Low	Healthy	0.0000	0.0000	0.0000	0.0000	0.0000
Moderate	High	Poor	0.0600	0.0740	0.0882	0.1045	0.1080
Moderate	High	Moderate	0.0630	0.0777	0.0926	0.1097	0.1134
Moderate	High	Healthy	0.0660	0.0814	0.0970	0.1150	0.1188
Moderate	Moderate	Poor	0.0485	0.0598	0.0713	0.0845	0.0873
Moderate	Moderate	Moderate	0.0500	0.0617	0.0735	0.0871	0.0900
Moderate	Moderate	Healthy	0.0515	0.0636	0.0757	0.0897	0.0927
Moderate	Low	Poor	0.0300	0.0370	0.0441	0.0523	0.0540
Moderate	Low	Moderate	0.0350	0.0432	0.0515	0.0610	0.0630
Moderate	Low	Healthy	0.0400	0.0494	0.0588	0.0697	0.0720
Healthy	High	Poor	0.0840	0.1050	0.1260	0.1500	0.1680
Healthy	High	Moderate	0.0882	0.1103	0.1323	0.1575	0.1764
Healthy	High	Healthy	0.0924	0.1155	0.1386	0.1650	0.1848
Healthy	Moderate	Poor	0.0679	0.0849	0.1019	0.1213	0.1358
Healthy	Moderate	Moderate	0.0700	0.0875	0.1050	0.1250	0.1400
Healthy	Moderate	Healthy	0.0721	0.0901	0.1082	0.1288	0.1442
Healthy	Low	Poor	0.0420	0.0525	0.0630	0.0750	0.0840
Healthy	Low	Moderate	0.0490	0.0613	0.0735	0.0875	0.0980
Healthy	Low	Healthy	0.0560	0.0700	0.0840	0.1000	0.1120

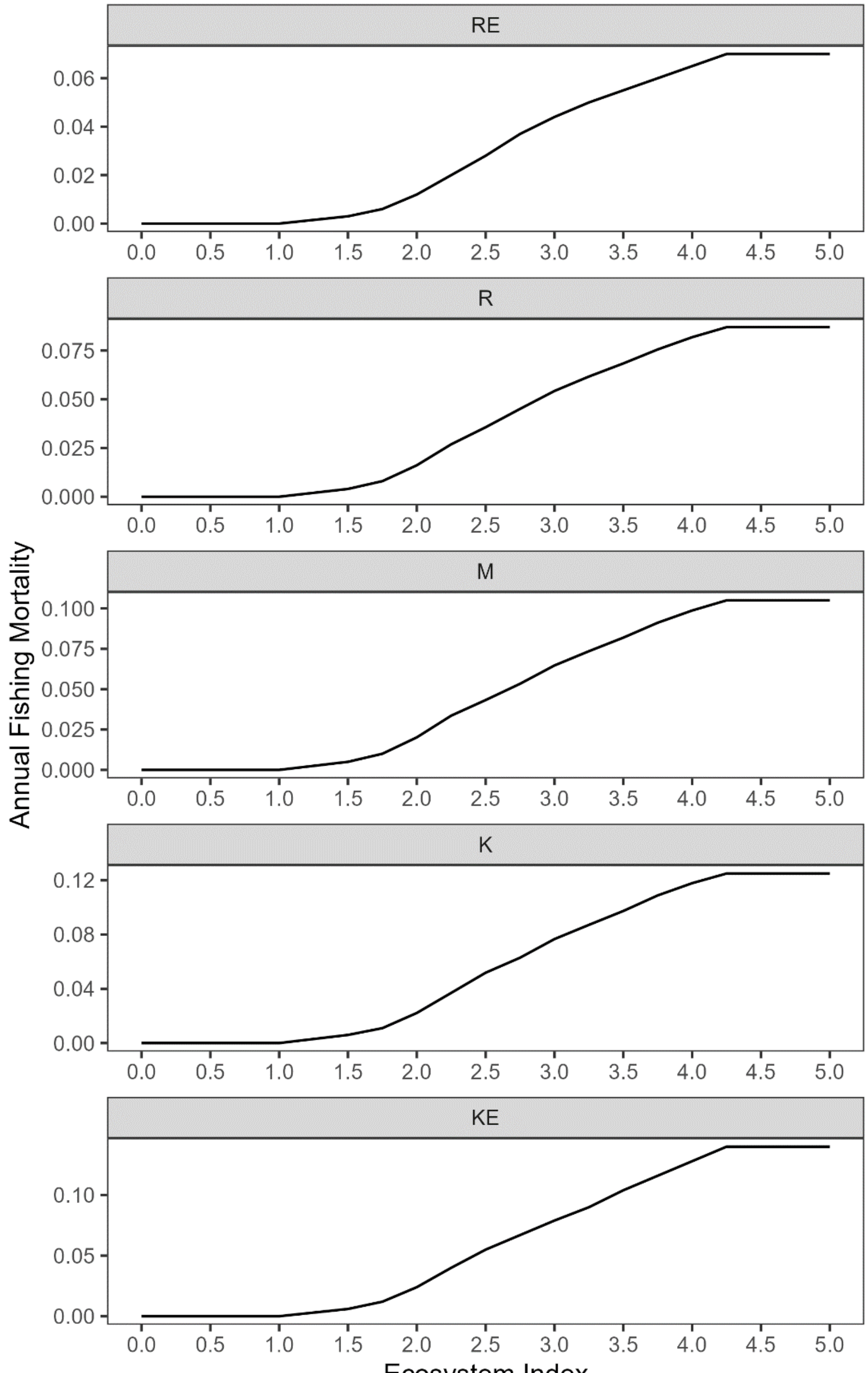

Figure 18 **Harvest control rules (fishing mortality) under the ecosystem-based fisheries management strategy.**

Appendix 3. Management Strategy Evaluation Results

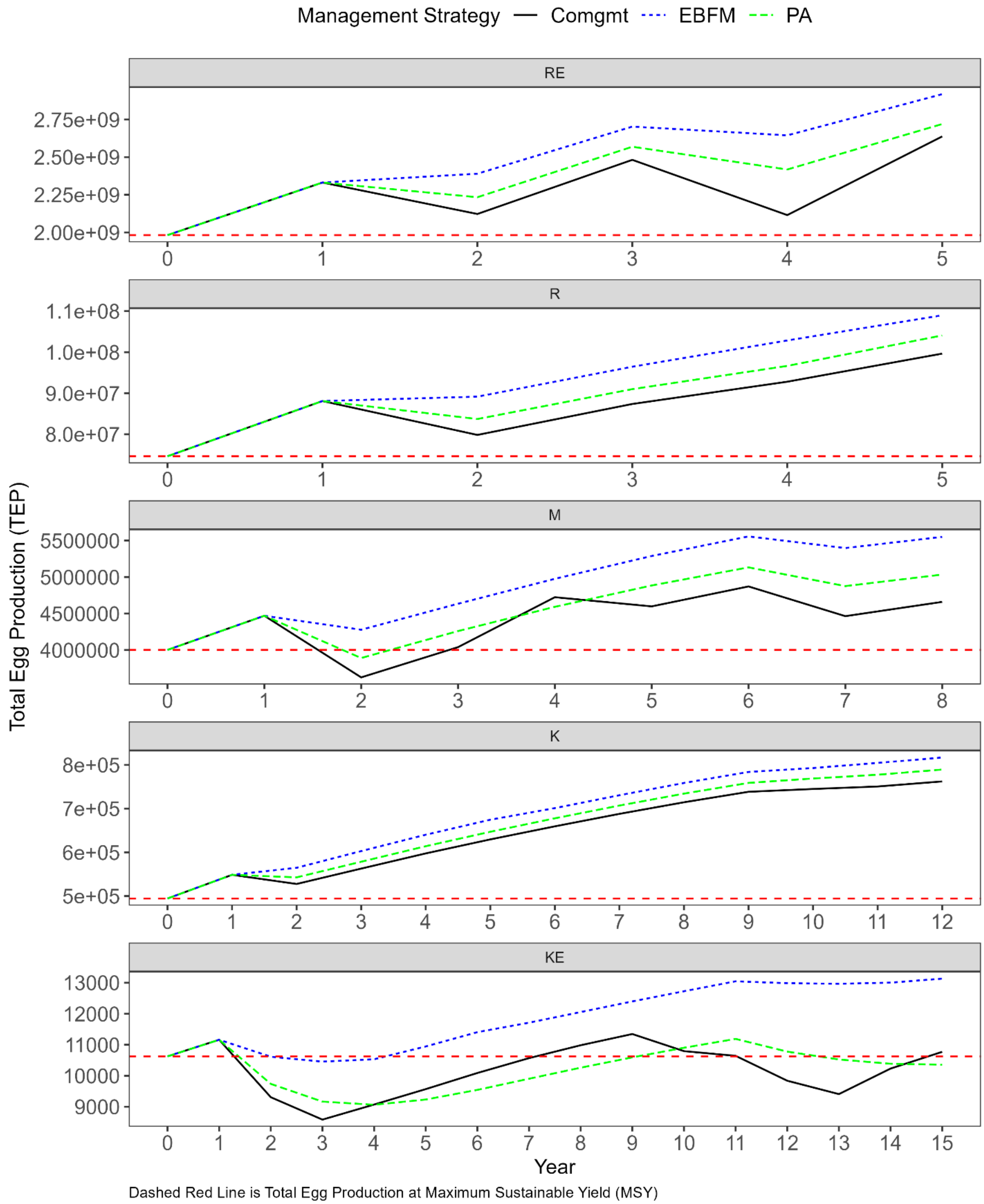

Figure 19 **Annual total egg production by population and management strategy for Scenario One.**

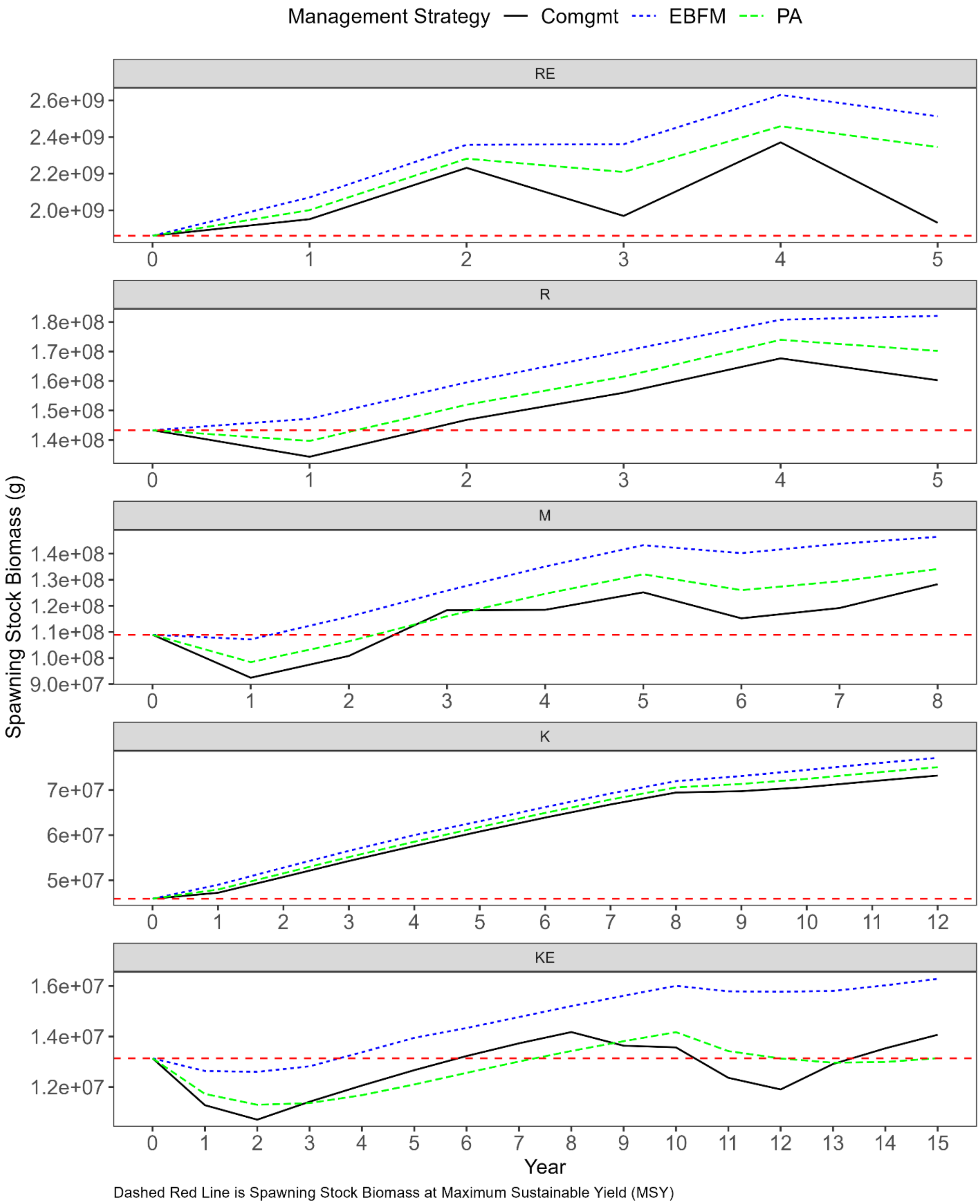

Figure 20 Annual spawning stock biomass by population and management strategy for Scenario One.

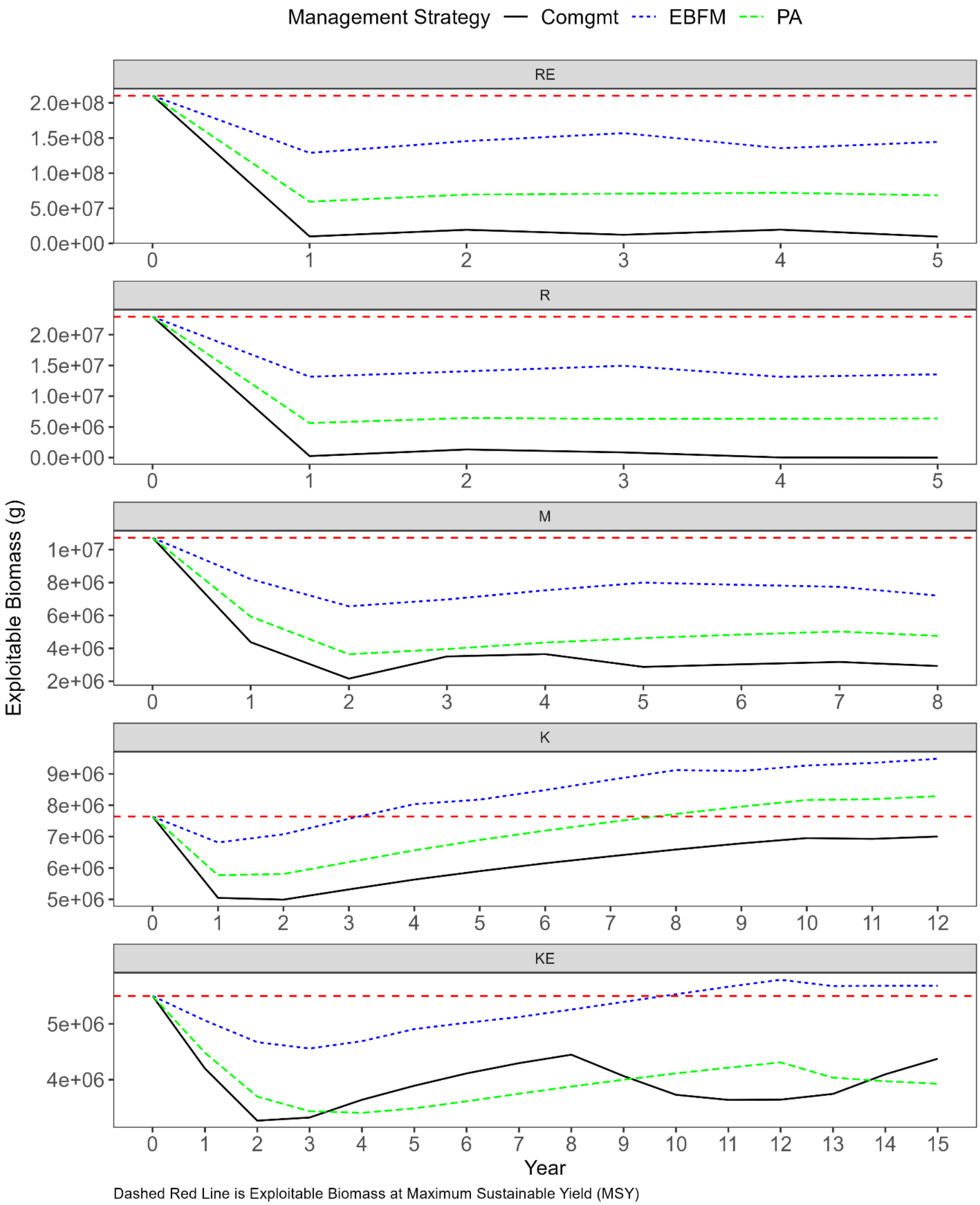

Figure 21 Annual exploitable biomass by population and management strategy for Scenario One.

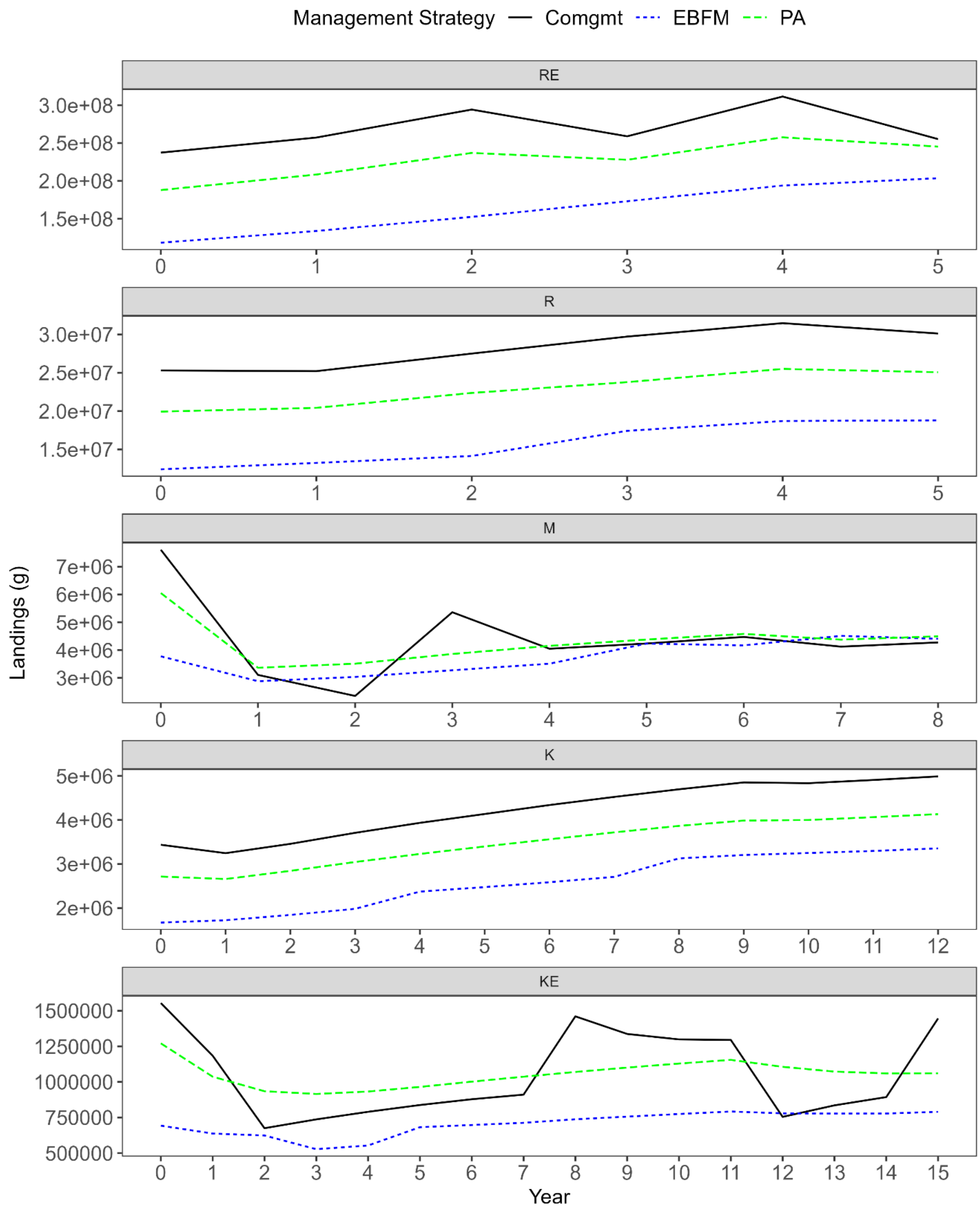

Figure 22 Annual landings by population and management strategy for Scenario One.

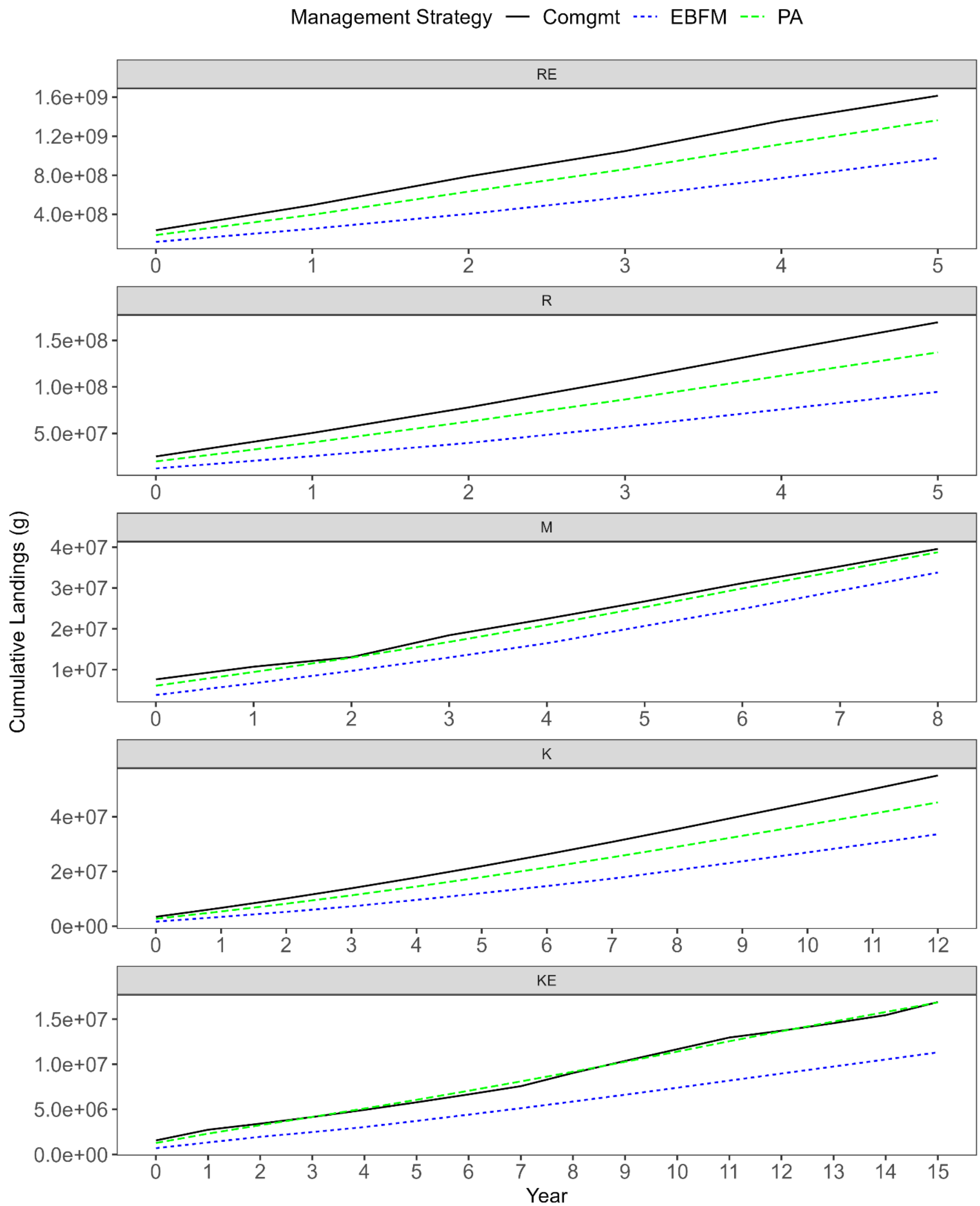

Figure 23 **Cumulative landings by population and management strategy for Scenario One.**

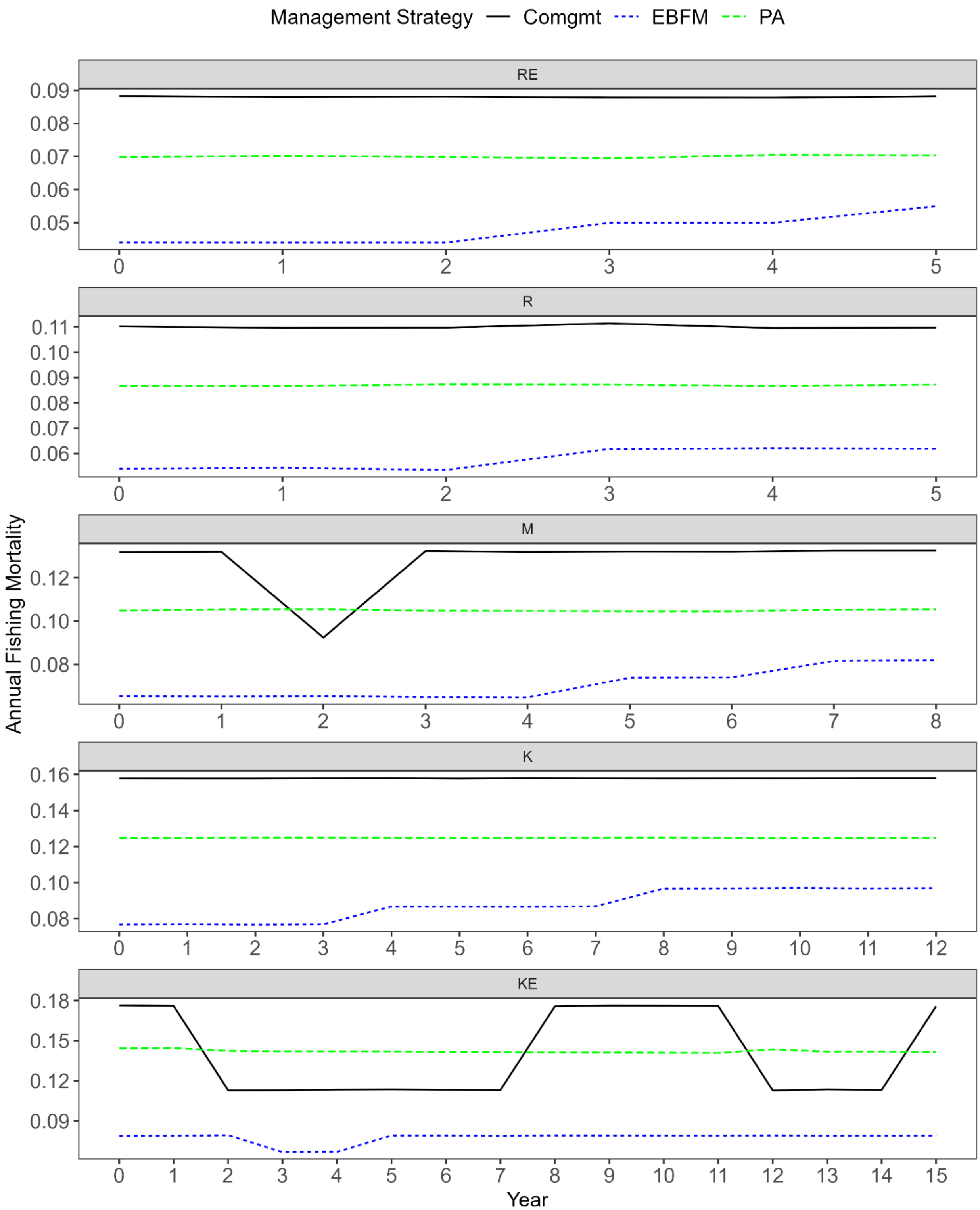

Figure 24 **Annual fishing mortality by population and management strategy for Scenario One.**

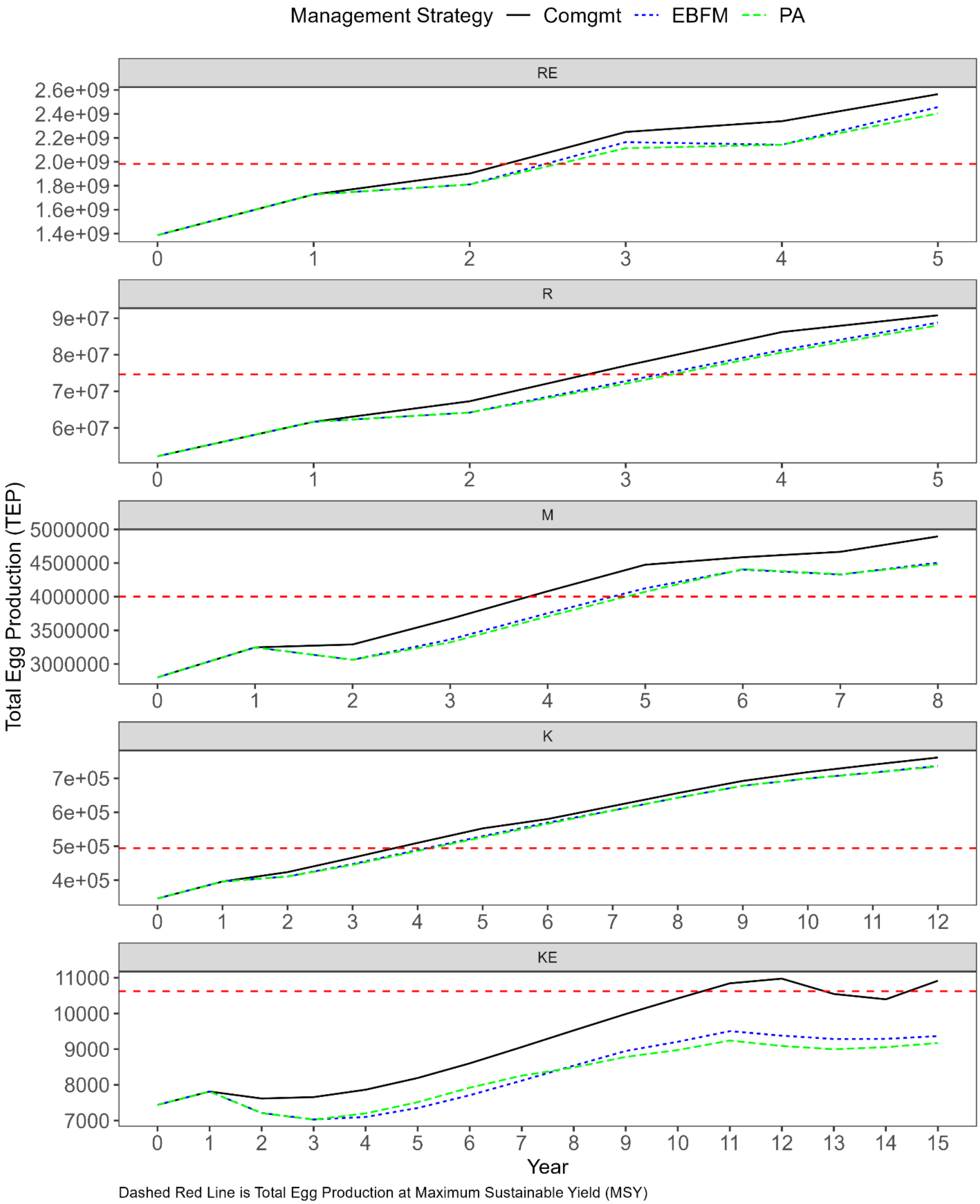

Figure 25 Annual total egg production by population and management strategy for Scenario Two.

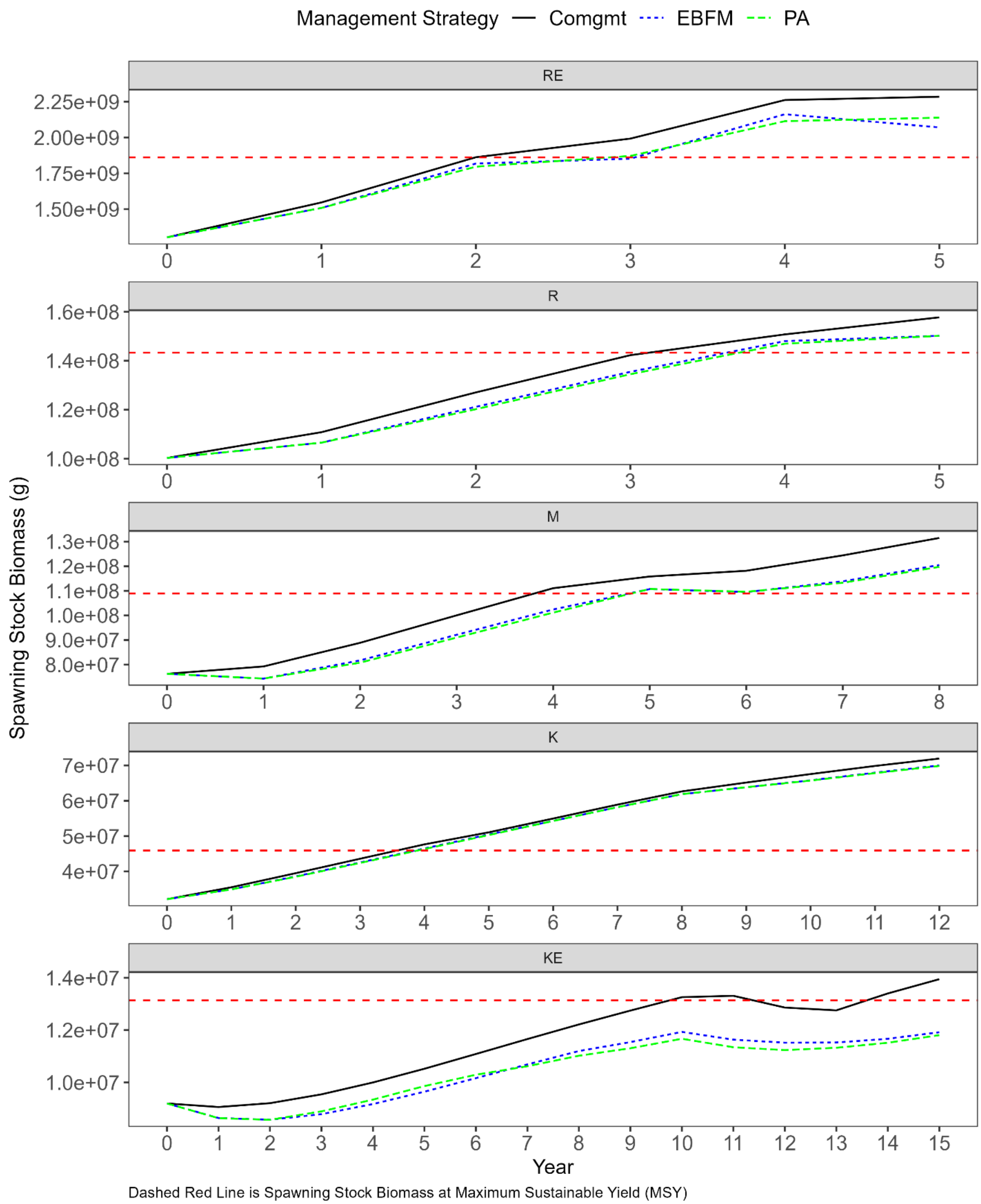

Figure 26 **Annual spawning stock biomass by population and management strategy for Scenario Two.**

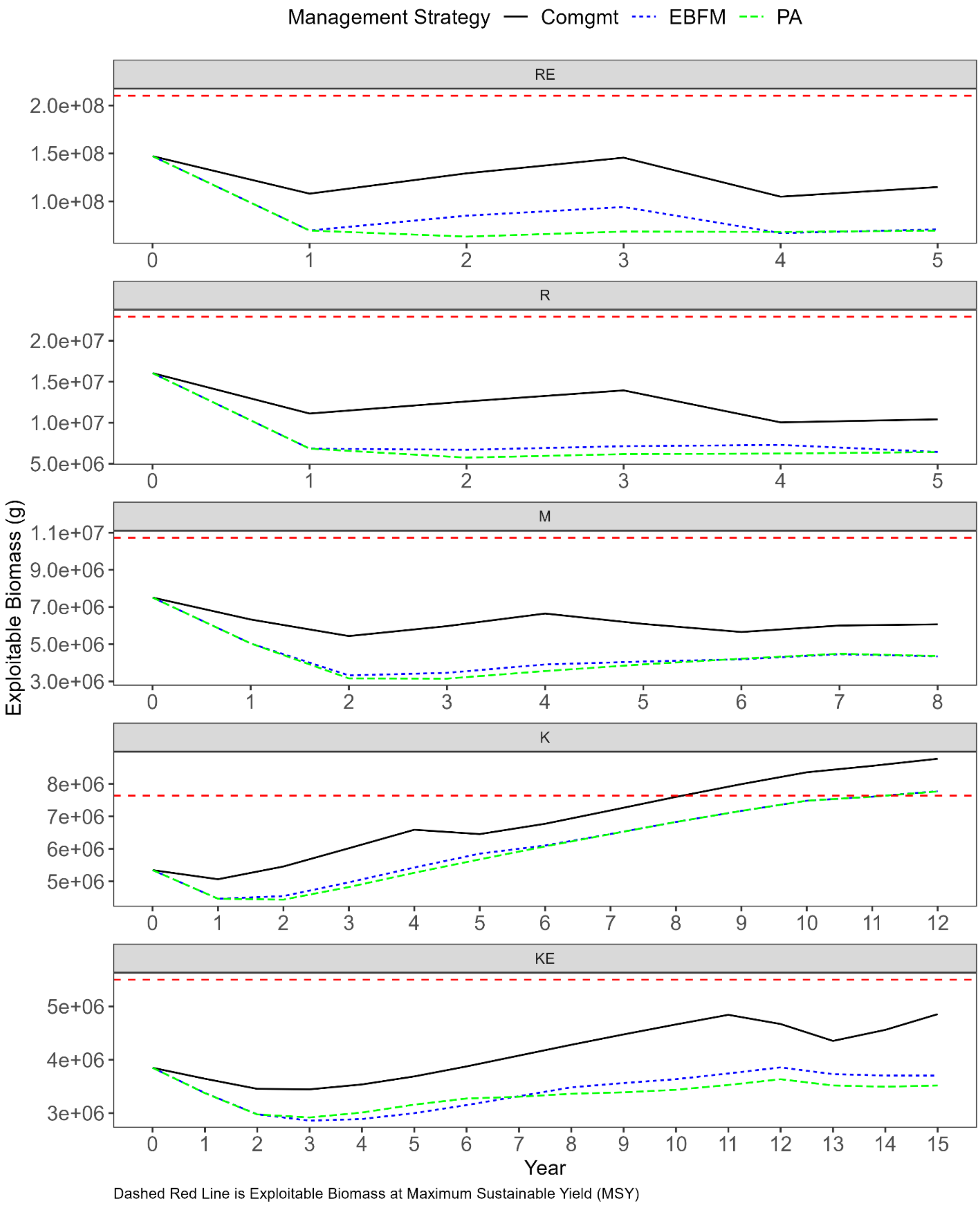

Figure 27 Annual exploitable biomass by population and management strategy for Scenario Two.

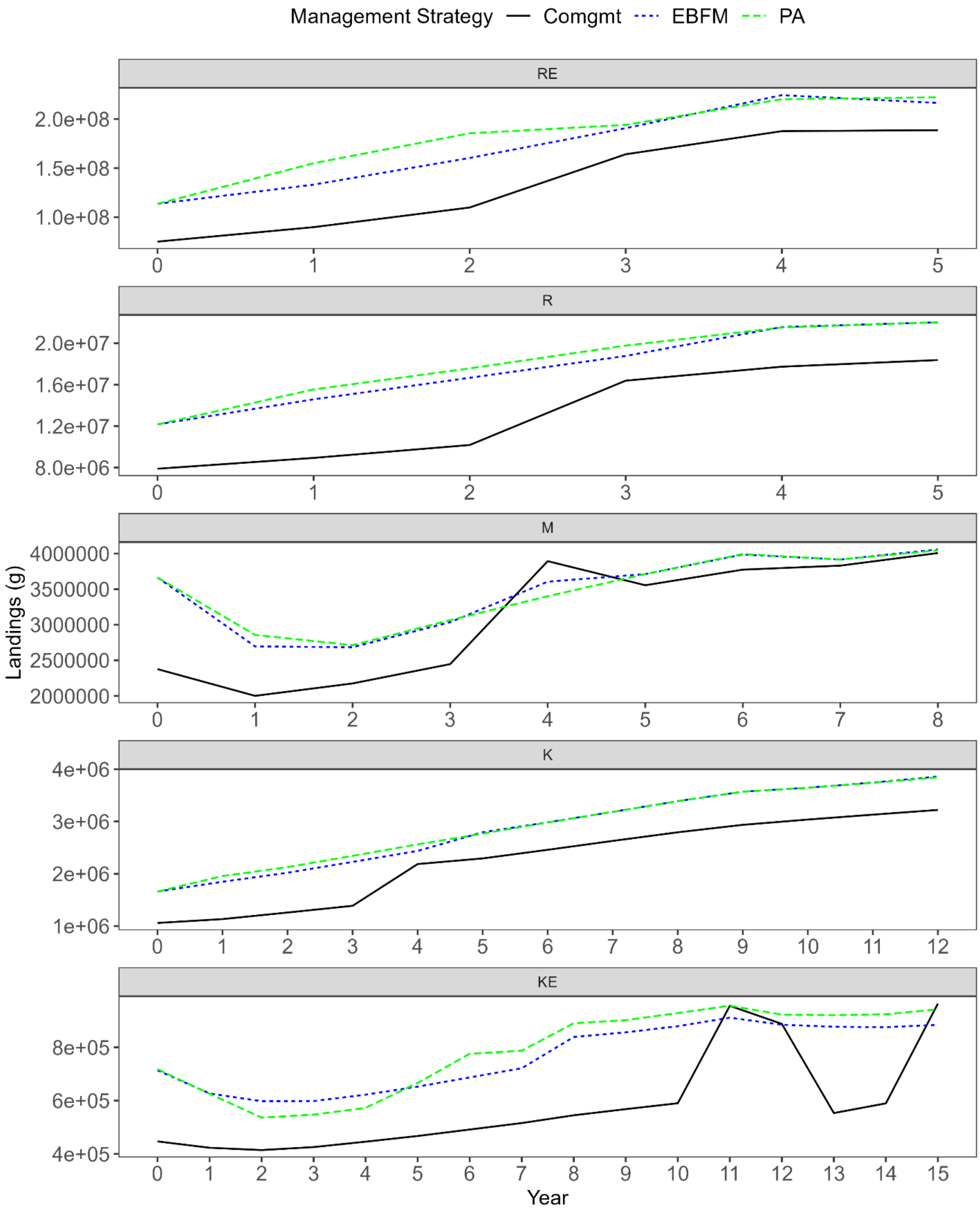

Figure 28 Annual landings by population and management strategy for Scenario Two.

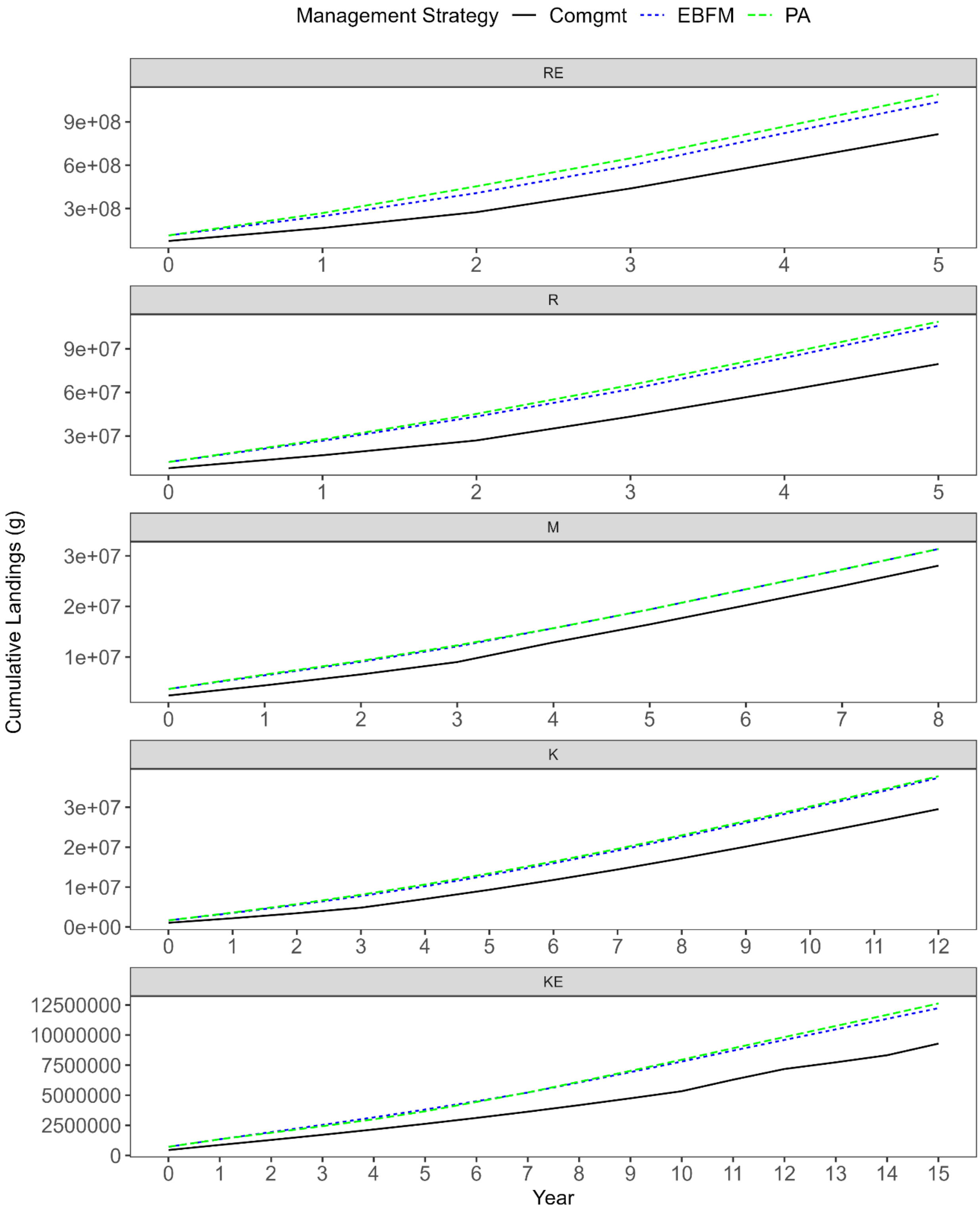

Figure 29 **Cumulative landings by population and management strategy for Scenario Three.**

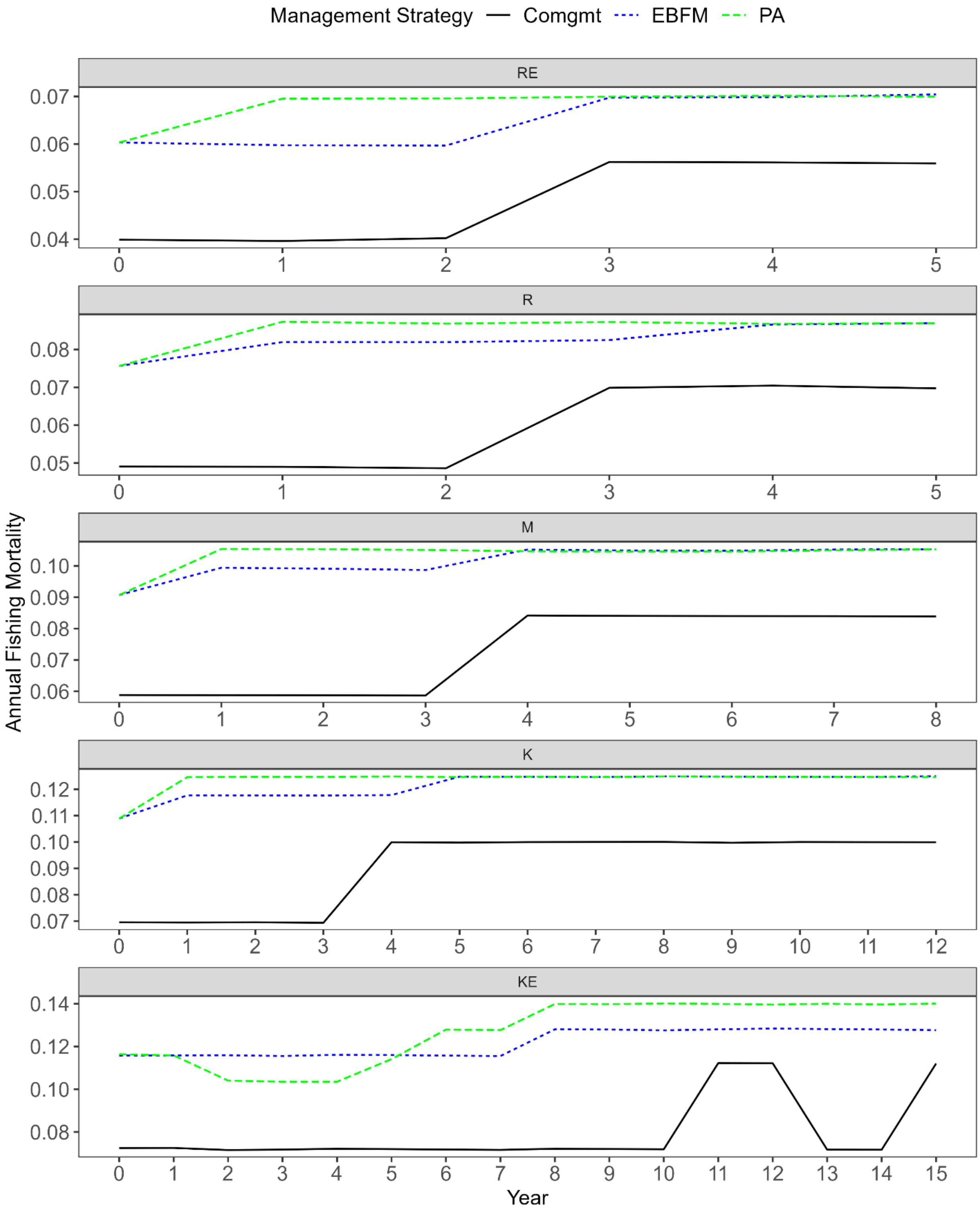

Figure 30 **Annual fishing mortality by population and management strategy for Scenario Two.**

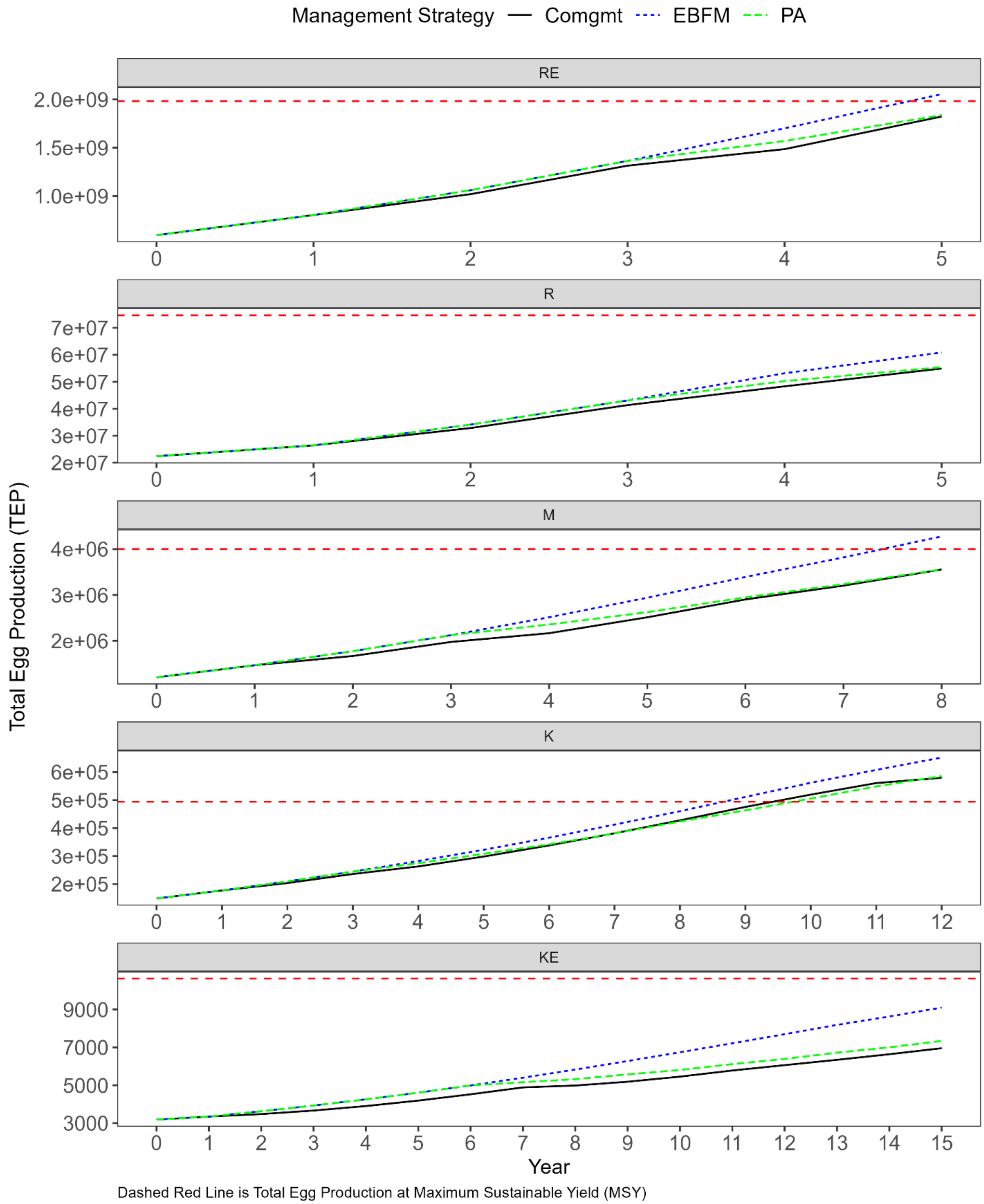

Figure 31 **Annual total egg production by population and management strategy for Scenario Three.**

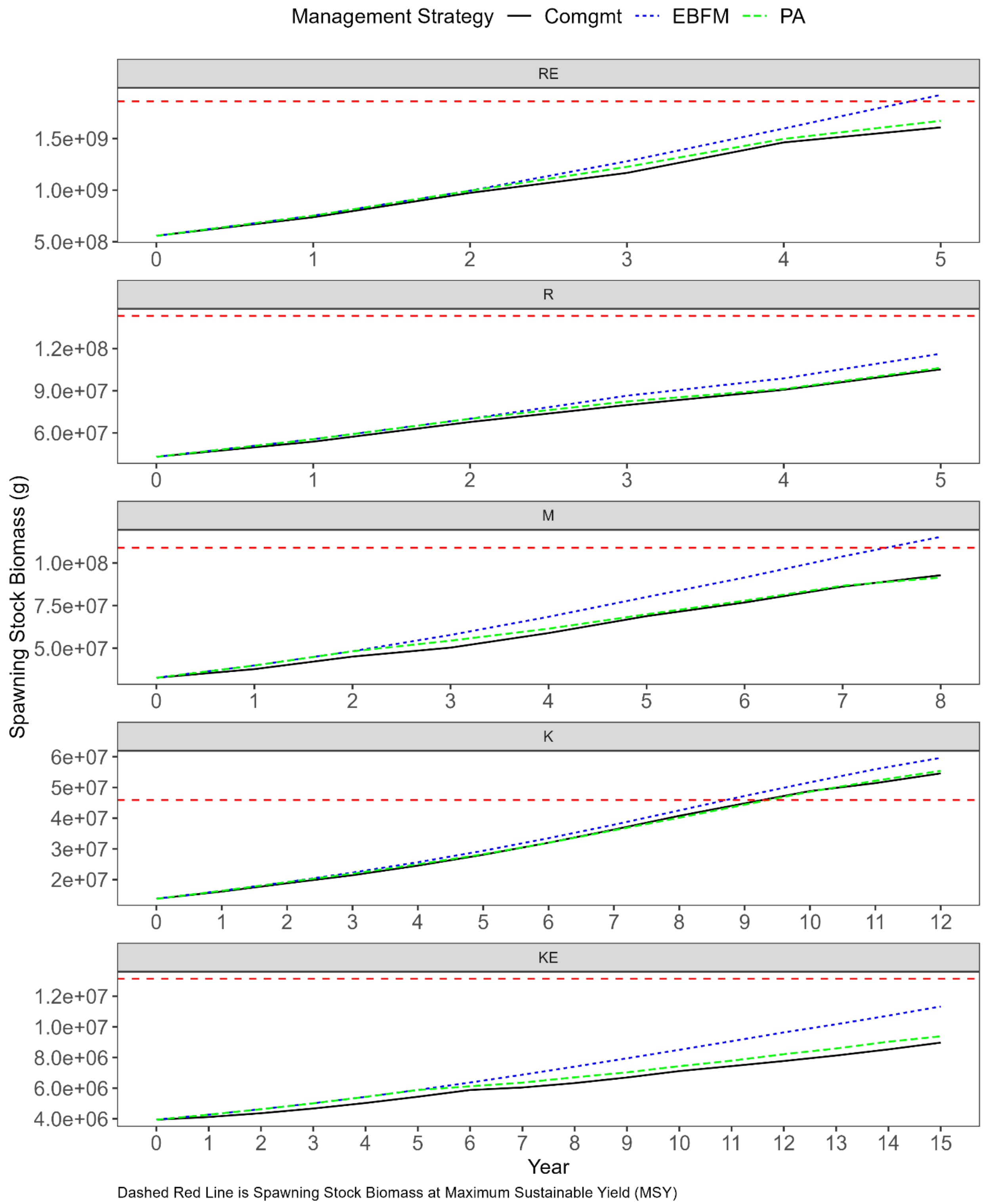

Figure 32 Annual spawning stock biomass by population and management strategy for Scenario Three.

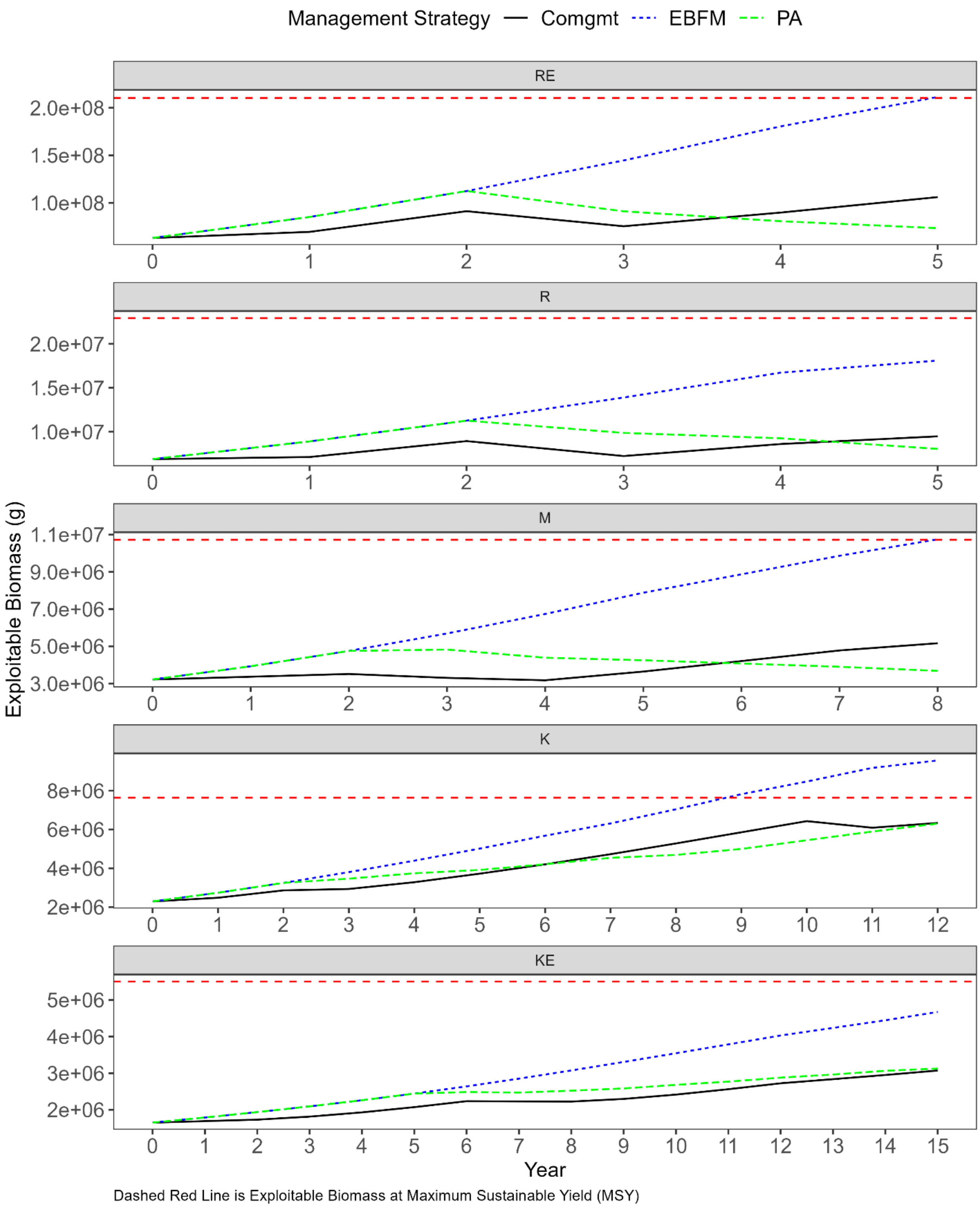

Figure 33 Annual exploitable biomass by population and management strategy for Scenario Three.

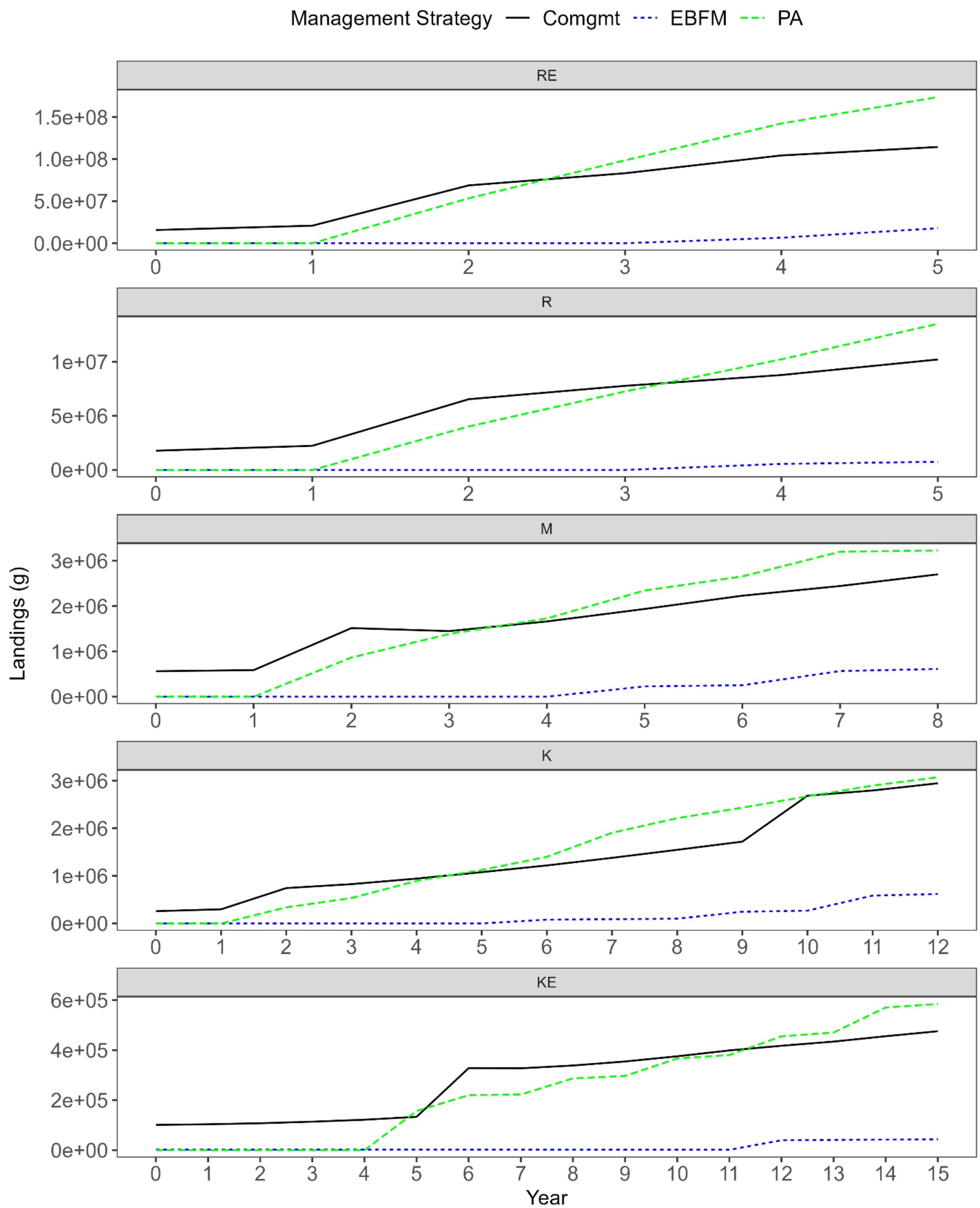

Figure 34 Annual landings by population and management strategy for Scenario Three.

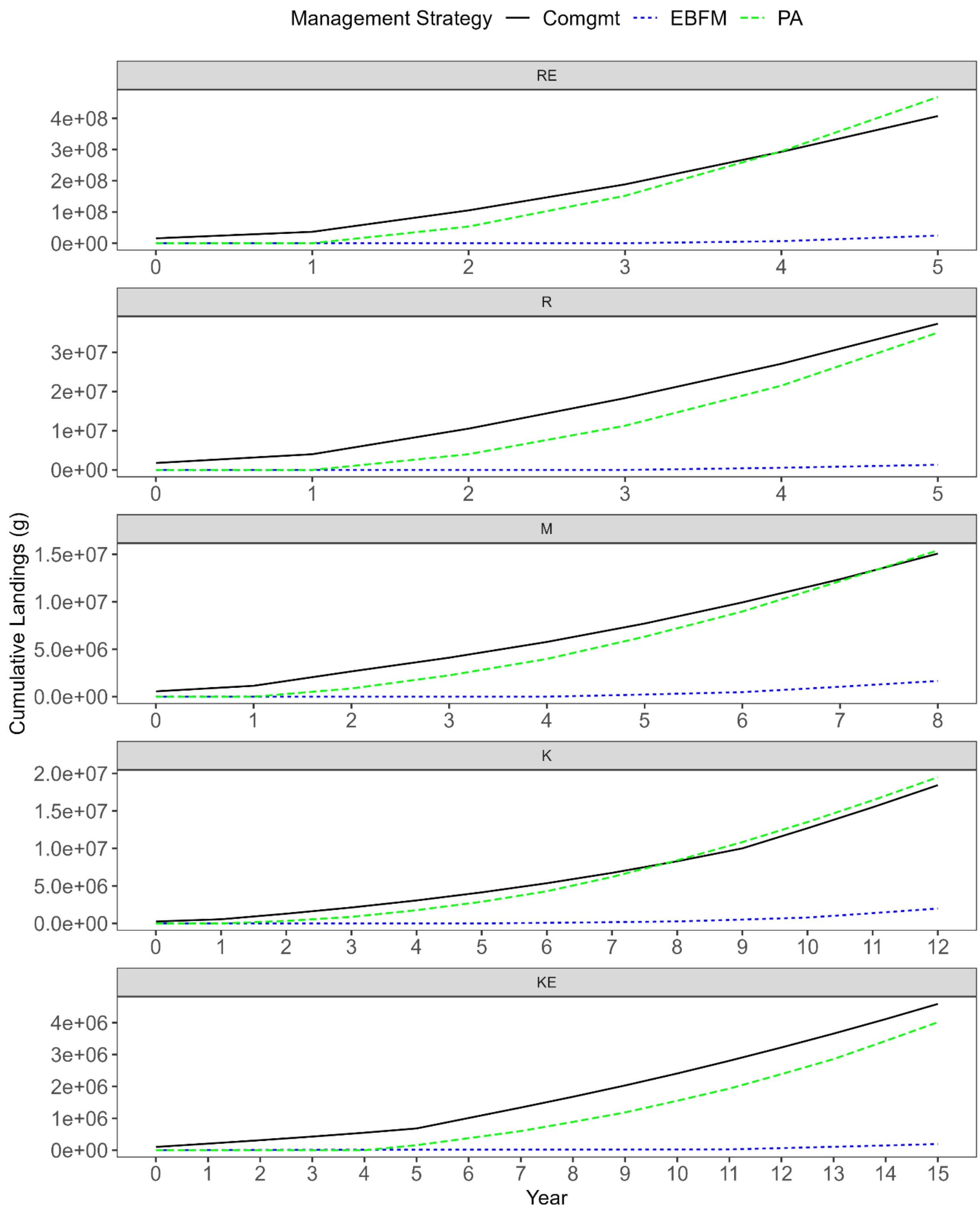

Figure 35 Cumulative landings by population and management strategy for Scenario Three.

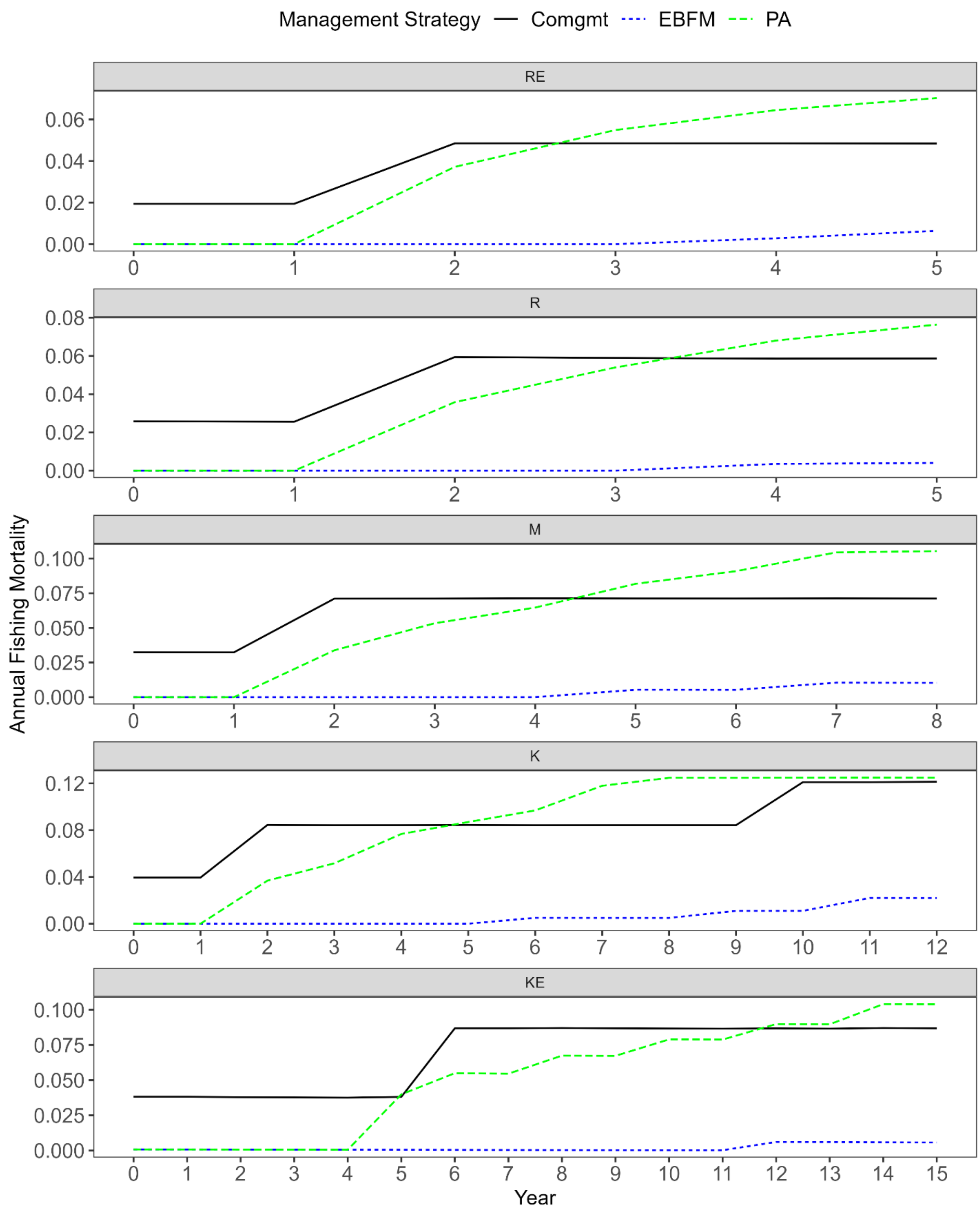

Figure 36 **Annual fishing mortality by population and management strategy for Scenario Three.**