Evaluating the Effectiveness of Fish Stock Rebuilding Plans in the United States

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Evaluating the Effectiveness of Fish Stock Rebuilding Plans in the United States

Committee on Evaluating the Effectiveness of Stock Rebuilding Plans of the 2006 Fishery Conservation and Management Reauthorization Act

Ocean Studies Board

Division on Earth and Life Studies

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This report has been reviewed in draft form by individuals chosen for their diverse perspectives and technical expertise, in accordance with procedures approved by the NRC's Report Review Committee. The purpose of this independent review is to provide candid and critical comments that will assist the institution in making its published report as sound as possible and to ensure that the report meets institutional standards for objectivity, evidence, and responsiveness to the study charge. The review comments and draft manuscript remain confidential to protect the integrity of the deliberative process. We wish to thank the following individuals for their participation in their review of this report:

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Although the reviewers listed above have provided many constructive comments and suggestions, they were not asked to endorse the conclusions or recommendations nor did they see the final draft of the report before its release. The review of this report was overseen by **Andrew Solow**, Woods Hole Oceanographic Institution, appointed by the Divison on Earth and Life Studies, and **Bonnie McCay**, Rutgers, The State University of New Jersey, appointed by the Report Review Committee, who were responsible for making certain that an independent examination of this report was carried out in accordance with institutional procedures and that all review comments were carefully considered. Responsibility for the final content of this report rests entirely with the authoring committee and the institution.

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SUMMARY

In the United States (U.S.), the Fishery Conservation and Management Act of 1976, now known as the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA), was the first major legislation to regulate federal fisheries in the U.S. Fishery Conservation Zone (later designated as the U.S. exclusive economic zone). Although the MSFCMA contained language to "prevent overfishing", the emphasis was on developing the domestic fishery. Major declines in the productivity of several important fisheries led Congress to amend the MSFCMA in 1996, with the Sustainable Fisheries Act, which more clearly defined overfishing and required rebuilding of overfished stocks within a specified time limit. The re-authorization of the MSFCMA passed by Congress in 2006 included additional mandates for conserving and rebuilding fish stocks and strengthening the role of scientific advice in fisheries management.

The depleted status of many fish stocks continues to be a challenge for fishery managers and the fisheries that depend on these stocks. Approximately 20% of the fisheries that have been assessed are considered overfished according to the September 2012 stock status Report to Congress prepared by the U.S. National Oceanic and Atmospheric Administration (NOAA). Overfished refers to a stock that is below the minimum stock size threshold, commonly set to half the stock size at which maximum sustainable yield (MSY) is achieved. Under the provisions of the MSFCMA, rebuilding plans for overfished stocks, covering both commercial and recreational fisheries, should take no more than 10 years, except when certain provisions apply. To meet these provisions, rebuilding plans have required substantial reductions in catch and effort for many fisheries, raising concerns about the consequent social and economic impacts to the fishing communities and the industry. Fishing restrictions have not only affected stocks under rebuilding plans, but have also impacted the utilization of stocks that are not overfished but are part of mixed-stock fisheries. In 2010, U.S. Senator Olympia Snowe and U.S. Representative Barney Frank requested that the NOAA Administrator fund a study by the National Academy of Sciences' National Research Council (NRC) regarding the MSFCMA's rebuilding requirements.

The committee reviewed the technical specifications that underlie the current set of federally-implemented rebuilding plans, the outcomes of those plans in terms of trends in fishing mortality and stock size, and changes in stock status with respect to fishery management reference points.

A total of 85 stocks or stock complexes were declared overfished under the provisions of the MSFCMA. Rebuilding plans were implemented for 79 stocks, of which 25 were classified as rebuilt[1] and 5 more stocks rebuilt before a plan was implemented. Based on the review of information for a subset of stocks that are assessed by analytical methods, the committee found that fishing mortality of stocks placed under rebuilding plans has generally been reduced and stock biomass has generally increased following reductions in fishing mortality. Although some stocks have rebuilt, others are still below rebuilding targets, and some continue to experience overfishing. Given the inherent uncertainties in both specifying a threshold for rebuilding and in determining whether a stock has dropped below that threshold, the current policy dependence on thresholds results in discontinuities in management when there is a change in stock status associated with updated stock assessments. While the Committee attributes some of the variable or mixed performance of rebuilding plans to scientific uncertainty, this should not be interpreted as a criticism of the science. It often reflects a mismatch between policy makers' expectations for scientific precision and the inherent limits of science because of data limitations and the complex dynamics of ecosystems.

The mixed outcomes of rebuilding plans have added to concerns about the significant social and economic costs associated with the implementation of time-constrained rebuilding plans. To address these rebuilding challenges, the committee highlights the following key findings for consideration by scientists, managers, and policy makers:

1) Harvest control rules that promptly, but gradually reduce fishing mortality as estimated stock size falls below B_{MSY} could result in a lower likelihood of a stock becoming overfished and provide an approach for rebuilding if necessary;
2) Fishing mortality reference points seem to be more robust to uncertainty than biomass reference points both in the context of rebuilding and more generally;
3) Rebuilding plans that focus more on meeting selected fishing mortality targets than on exact schedules for attaining biomass targets may be more robust to assessment uncertainties, natural variability and ecosystem considerations, and have lower social and economic impact.
   a. The rate at which a fish stock rebuilds depends on ecological and other environmental conditions such as climate change, in addition to the fishing-induced mortality,
   b. A rebuilding strategy that maintains reduced fishing mortality for an extended period (e.g., longer than the mean generation time) would rebuild the stock’s age structure and be less dependent on environmental conditions than one that requires rebuilding to pre-specified biomass targets, and
   c. When rebuilding is slower than expected, keeping fishing mortality at a constant level below F_{MSY} may forgo less yield and have fewer social and economic impacts than a rule that requires ever more severe controls to meet a predetermined schedule for reaching a biomass target.
4) In the case of data-poor stocks for which analytical assessments are not available and catch limits are therefore difficult to establish, empirical rebuilding strategies that rely on input controls to reduce fishing mortality may be more effective and defensible than strategies based on annual catch limits and B_{MSY} targets.
5) Retrospective reviews of the socioeconomic impacts of rebuilding plans are rare, in part due to data availability. Such reviews would help in refining rebuilding plans and objectives and ameliorating for the consequences of such actions.

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[1] As of 30 September 2012.

These key findings are described in more detail below. The remainder of this summary is organized around the seven tasks that the committee was directed to address.

TASK 1

Evaluate methods and criteria used (1) to set target fishing mortality and biomass levels for rebuilding overfished stocks, and (2) to determine the probability that a particular stock will rebuild by a certain date. Consider the quantity and quality of information available for defining maximum sustainable yield (MSY)-based reference points or proxies for such reference points. Compare these methods and criteria to those used in major fishery management settings outside the U.S.

The committee reviewed the evolution of the MSFCMA from its origins in 1976, its subsequent amendments that introduced rebuilding requirements and accountability measures, and the guidelines for rebuilding U.S. fish stocks. Fisheries management has evolved substantially since 1977 when the U.S. extended its jurisdiction to 200 miles, in the direction of being more prescriptive and precautionary in terms of preventing overfishing and rebuilding overfished fisheries. This evolution has been positive in making clear the objectives, resulting in fewer fisheries are currently subject to overfishing. However, the trade-offs between precaution, ecosystem impacts, and net benefits from fisheries have not been fully evaluated.

One of the central tenets of fisheries management is the concept of MSY, which represents the maximum, sustainable, long-term average yield that can be taken from a fish stock. The MSFMCA bases the success or failure of fisheries management on the MSY concept and its associated population biomass (B_{MSY}) and fishing mortality rate (F_{MSY}), which are used as reference levels against which to compare how stock status and harvest rate change over time.

MSY is not fixed but may be influenced by a variety of factors encompassing fishing practices, ecological interactions and environmental conditions. In addition, management reference points based on MSY have a level of uncertainty that depends on the amount and quality of information available. Estimates of B_{MSY} may be imprecise even for stocks that are relatively "data-rich," because of the complex and dynamic nature of ecosystems. The MSFCMA is virtually silent on the implications of uncertainty and variability of MSY. Guidelines for implementing the Act are primarily oriented to situations in which estimates of MSY reference points are reasonably precise and stable. Although the MSY approach has been successful for some fisheries, in other situations, management based on MSY falls short in addressing ecosystem complexity and variability, and in accounting for uncertainty in the estimates of stock size and reference points.

The requirement to end overfishing for all stocks in mixed-stock fisheries has protected less productive species but has led to loss of yield for healthy stocks in the same complex. The "Mixed-Stock Exception" in the MSFCMA provides an option for reducing the impact of rebuilding on the harvest of healthy stocks. However, it has not been invoked in these cases, in part due to the narrow range of situations to which it applies under the MSFCMA and also because of the complexity of the issue it is meant to address. The operational feasibility of the mixed-stock exception could be modified to expand the range of situations to which it can be applied, subject to assurances that the less productive species are not driven to unacceptably low abundance.

Rebuilding Plans are designed using quantitative models to project likely future trends in stock size in response to alternative harvest control rules. This approach works best for data- and knowledge-rich fisheries, which are generally those stocks with a long history of exploitation and high economic value, and which contribute the bulk of the U.S. landings. The main focus of this review was on the stocks for which quantitative assessments and estimates of MSY reference points are available. For many stocks, however, data and understanding are so limited that stock projections cannot be conducted, and stock-by-stock application of MSY-based control rules is unrealistic. NOAA reports to Congress indicate that over half of the stocks or stock complexes identified have either not been assessed or their status as overfished or experiencing overfishing is unknown.

In general, fishing mortality reference points appear more robust to scientific uncertainty than biomass reference points. Fishing mortality reference points are often more reliably estimated at lower stock sizes than biomass reference points, whose estimates rely more strongly on density-dependent processes that generally manifest only at higher stock sizes. Furthermore, proxy values for fishing mortality reference points can often be derived from other information sources, such as life history parameters of growth and natural mortality, which do not require estimates of future recruitment levels.

When data and understanding are too limited to design a rebuilding plan with a predetermined time limit for rebuilding, it may be practical to implement harvest control measures (either by adjusting catch limits or effort controls) that at a minimum would be expected to increase stock size. In the case of data-poor stocks for which analytical assessments are not available, and therefore catch limits are difficult to establish, empirical rebuilding strategies that rely on input controls to reduce fishing mortality may be more effective and defensible than strategies based on annual catch limits and B_{MSY} targets as prescribed by the National Standard 1 Guidelines (NS1G).

TASK 2

Assess the effects of uncertainty in current stock abundance, population dynamics, and variability in recruitment in setting rebuilding targets. Identify criteria for adjusting rebuilding targets and schedules based on new information and updated stock assessments.

Scientific management advice is subject to several sources of uncertainty, including variability and bias in the data, sensitivity to model assumptions, implementation uncertainty (reflecting management effectiveness and fisher responses), and unpredictable natural events. These sources act simultaneously, resulting in substantial uncertainty surrounding reference points, the determination of stock status, and projected outcomes of management regulations**.** As required by law, rebuilding plans have target years for recovery to B_{MSY}, but the rate at which stocks rebuild is probabilistic such that some stocks will rebuild before the target year while others will rebuild after the target year or not rebuild until environmental conditions improve, even if the rebuilding plan is implemented as intended, fishing mortalities are close to the targets, and targets are based on robust stock assessments.

The MSFCMA requires review of progress of rebuilding plans at least every second year. However, reviews do not always include updated, quantitative stock assessments. The frequency of assessments varies widely, both within and among regions, from stocks that have never been assessed to stocks that are assessed annually. More frequent assessments might lead to more frequent, but less extreme, changes in rebuilding plans and closer adherence to fishing mortality targets.

Due to the uncertainty in stock assessments, the perceived status of fish stocks in any particular year can change substantially as more data become available and as assessment methods change over time. According to the most recent assessments available, there is a substantial probability of (i) classifying stocks as overfished and requiring rebuilding plans when later assessments indicate that the stocks were not below the minimum stock-size threshold, and (ii) classifying stocks as rebuilt when the updated assessments indicate that the stocks were never overfished. By inference, the inverse may also occur so that overfished stocks may be misclassified as not overfished. How many and which stocks these are cannot be determined from the data available.

The MSFMCA, as operationalized by the NS1G, requires an end to overfishing and provides minimum standards for stock rebuilding, namely that stocks designated as overfished must rebuild to B_{MSY} within a maximum time period. Although effective in increasing the probability that rebuilding occurs quickly once a stock has fallen below the minimum stock-size threshold, preventative management actions taken prior to falling below the threshold could obviate the need for a rebuilding plan. Harvest control rules that promptly, but gradually, reduce fishing mortality as estimated stock size falls below B_{MSY} could result in a lower likelihood of a stock becoming overfished as well as providing an approach for rebuilding if necessary.

Such rules may reduce the need for more stringent reductions that would be required if the stock fell below the minimum stock-size threshold. Delaying reductions in fishing mortality until the stock falls below the threshold creates a discontinuity: – managers are then required to make immediate and substantial decreases in fishing mortality based on what may be only small changes in estimates of stock size from a previous assessment. Furthermore, the mandate that rebuilding targets be met with a certain minimum probability, along with the requirement to utilize the most current stock assessments, may lead to marked changes in rebuilding plans based on new data or models as they become available. These adjustments can also create economic and social impacts, potentially either positive (e.g., increases in allowable catch due to rapid rebuilding) or negative (e.g., decreases in allowable catch when rebuilding is slower than expected). Although these adjustments may reflect the best available science, the perceived credibility of the science among stakeholders may be reduced when rebuilding plans are changed markedly.

Population projections used in rebuilding analyses have much higher uncertainties than historical estimates of population sizes. Because of the uncertainty surrounding projections, the emphasis placed on achieving a biomass threshold in a defined time frame may require severe reductions in target fishing mortality (well below F_{MSY}) when rebuilding is slower than expected. In situations where recruitment is below expectations (e.g., due to unfavorable environmental conditions), a control rule aimed at maintaining fishing mortality at some constant level below F_{MSY} may forgo less yield, especially in mixed-stock situations, and have fewer social and economic impacts than one that forces ever more severe controls to try to keep rebuilding on schedule.

The standard approach used in most regions for adjusting catch limits involves the use of a single "best" estimate of current or projected stock size. Often, several alternative models or configurations of a standard stock-assessment model are first applied and the "best" of these is selected using formal criteria or expert judgment. An alternative to this best-assessment approach is to describe the consequences of alternative decision rules under each of the models considered plausible. A general framework known as Management Strategy Evaluation (MSE) has been used internationally and by some RFMCs to evaluate alternative harvest control rules that specify in advance how catch limits will be adjusted in response to new data as they become available. Different candidate rules are tested across a broad range of simulated scenarios (e.g., different levels of stock productivity, different environmental regimes), a process that allows decision-makers to select a decision rule based on robust performance under various scenarios.

TASK 3

Provide an overview of the success of rebuilding plans under the MSA and compare to success of approaches used outside the U.S. Using a few representative rebuilding plans, identify factors (such as fishing mortality rate, life histories, uncertainty in stock assessments, and others) that affect the timeframe over which a stock is rebuilt.

The committee reviewed the 85 stocks or stock complexes that were declared to be overfished or approaching an overfished state between 1997 and 2011. Rebuilding plans were implemented for 79 of these 85 stocks, based on target fishing mortalities generally lower than 75% F_{MSY} and substantially lower than this in some regions; rebuilding time frames chosen in those regions are much shorter than the maximum specified by the NS1G.

The committee focused on a subset of 55 stocks assessed using quantitative methods. The most recent assessments indicate that fishing mortality was reduced below F_{MSY} (i.e., overfishing was halted) in 23 of the 36 stocks that were subject to overfishing at the time of overfished designation. According to these assessments, 20 of the 55 stocks analyzed were not overfished, and 10 were actually above B_{MSY} at the time of overfished designation. Of the 35 stocks that were below the minimum stock size threshold:

- 43% of the stocks are no longer overfished; 10 have rebuilt and 5 are rebuilding.
- Of the remaining 20 stocks estimated to still be overfished, 11 had fishing mortalities well below F_{MSY} in the last year included in the assessment and are therefore expected to rebuild if low fishing mortalities are sustained.

Stocks that rebuilt or whose biomass increased appreciably were, in almost all cases, experiencing fishing mortalities below F_{MSY}.

Some stocks (9 of the 35) continue to be subject to overfishing even though fishing targets were set at or below 75% F_{MSY} to allow rebuilding within the maximum time frame. The failure of rebuilding plans to achieve the intended reductions in fishing mortality reflects implementation problems due to ineffective input controls and lack of accountability measures, difficulties in reducing fishing mortality of species caught as bycatch in other fisheries, or errors in the estimates of stock size that led to catch limits that were too high. In particular, retrospective biases in the assessments revealed apparent overestimations of stock size that contributed to continued overfishing.

The U.S. approach to rebuilding overfished stocks is comparable to that used by several developed countries (such as Australia, Canada and New Zealand) and the results are similar (in terms of the fraction of overfished stocks). The European Union has a higher proportion of stocks that are subject to overfishing than the U.S., although the proportion has decreased sharply in recent years.

TASK 4

Consider the effects of climate and environmental conditions, habitat loss and degradation, ecological effects of fishing on the food chain, and ecological interactions among multiple species, and identify ways to adjust rebuilding plans to take these factors into account.

Ecosystem variables related to climate, habitat, and food-web interactions can influence population dynamics, yielding a broader spectrum of possible outcomes than is typically considered in single-species rebuilding projections. Stock biomass forecasts and projections can vary in response to alternative plausible assumptions (models) and parameter values used in simulations, because the underlying population dynamics are nonlinear. Reference points, such as B_{MSY}, that are used throughout fisheries management, are based on single-species production functions that do not generally account for the influences of environmental and ecological interactions. The committee notes that reference points based on single-species assessments are likely to shift over time as a consequence of climate change and the complex and dynamic nature of ecosystems.

Fishing truncates the age structure of a population, especially when fisheries selectively harvest larger fish. Removing the more productive individuals from a population may amplify the effects of environmentally-driven recruitment variability. Rebuilding plans that restore the demographic structure of the overfished population are more likely to improve recruitment and increase the likelihood of success of the rebuilding effort than plans that restore spawning stock biomass without also restoring demographic structure. In nature, growth, maturity, and natural mortality are influenced by interactions with other species that may be competitors, predators, or prey. Fisheries management involves tradeoffs among harvested species that interact, even if these tradeoffs are not explicitly considered in management decisions. Our understanding of how ecosystems function is improving, in some cases enough to contribute to the models used in fisheries management. For example, stock assessments can be linked with multispecies models. Ecosystem considerations, among other reasons, argue for more emphasis on rebuilding plans that maintain reduced fishing mortality for an extended period (e.g., longer than the mean generation time). This strategy rebuilds age structure and is more robust to natural variability than a focus on biomass targets, which may be more or less attainable depending on environmental conditions.

TASK 5

Assess the types of information needed and current understanding of the economic and social impacts of rebuilding programs, particularly on fishing communities. Identify the economic, social, and ecological tradeoffs of rebuilding a fishery associated with shorter or longer *rebuilding times. Evaluate available methods for integrating these social, economic and ecological factors when designing and evaluating rebuilding plans.*

The relationship between economic and social factors and rebuilding programs that extend over multiple years is complex and dynamic, although the state of knowledge and understanding about these interactions is improving. Causal relationships among rebuilding and socio-economic outcomes are difficult to disentangle, due to the general quantity and quality of data and resources available to fishery managers and scientists, behavioral responses of those being impacted by the changes, and the multitude of confounding factors. It can also be difficult to establish counterfactual conditions that capture what the status of a stock might have been in the absence of rebuilding or under alternative rebuilding plans. The estimated impacts of a rebuilding plan are conditional on these (assumed or estimated) counterfactuals. Hence, the ability to predict and measure rigorously the *ex post* economic and social impacts and tradeoffs is limited.

Socioeconomic analyses and research are used to inform the evaluation of alternative rebuilding plans, but the role of the formal analyses in the decision process is less clear, as these decisions are made in a highly charged political setting. Furthermore, compliance with MSFCMA requires that economic and social considerations for rebuilding plans are contingent on biological mandates being met. Rebuilding plans that do not meet these mandates cannot be adopted, even if doing so would improve projected socioeconomic outcomes.

Fish stock rebuilding plans are designed to achieve rapid rebuilding of biomass and spawning stocks consistent with the biological characteristics of the resource. However, the requirement to rebuild within 10 years, if biologically possible, eliminates certain management options from consideration that could lead to greater social and economic benefits while still supporting stock recovery in the long run. Several alternative management strategies that could be considered in this context have been implemented successfully in venues outside the U.S. (e.g., New Zealand).

At the same time, socioeconomic considerations do influence the management of overfished stocks through the public participation process (e.g., public testimony to Councils regarding the magnitude of socioeconomic impacts). Stakeholder participation and concerns regarding the impacts of rebuilding plans can also result in *ad hoc* mitigation measures (e.g., disaster relief assistance) that operate outside of the fishery management process. The implications of these measures on other fisheries, and on the long-run social and economic viability of coastal communities are not fully known.

TASK 6

Summarize how the social, economic and ecological impacts of rebuilding plans are affected by the structure of fisheries management measures, e.g., limited entry, catch shares systems, and closed areas.

In the U.S., many commercial and recreational fisheries are managed by allocating a portion of a species' total allowable catch to different fishing sectors (e.g., defined by gear type, recreational versus commercial, and size of fishing vessel) and linking this allocation with additional controls, for example on fishing locations, seasons, technology, size and sex restrictions, and trip or bag limits. The incentives and constraints created by this (and any other) regulatory strategy affect the economics of fishing, the structure of fishing communities, and the choices available to fishermen. These common regulatory constraints, which are often tightened if stocks become depleted, reduce the ability of fishermen to adapt their fishing behaviors (e.g., changing where, how and for what species they fish) in response to the new harvest limits that accompany rebuilding plans. Although constraints and incentives may vary across regulatory strategies (e.g., catch shares, limited entry, regulated open-access), all approaches limit the capacity of fishermen to adapt practices in some manner. As a result, fishermen are less able to mitigate costs associated with rebuilding plans.

Another factor limiting the adaptive potential of fishermen is the highly specialized fleets that evolved in response to the sector-by-sector allocation process institutionalized by the RFMCs. While specialization can have economic gains, it also reduces the potential for behavioral responses, such as switching fishing gears to improve quality (and obtain higher prices for the fish) or switching between species in response to a rebuilding plan. Specialization of the fishing sector also has ripple effects in the fish processing and fishing-related industries and can result in local communities having less diversity in the local economy to mitigate short-run economic impacts.

In summary, the nature of fisheries management can lead to situations that exacerbate the economic and social impacts of meeting rebuilding targets by institutionalizing the specialization of the fishing industry (including fishing fleets, processing, and related support businesses). These constraints reduce the ability of the fishermen and communities to absorb some of the costs associated with curtailing catches and have potential impacts on the resilience of fishing communities.

TASK 7

Identify the biological, ecological, social and economic knowledge gaps that impede the implementation and effectiveness of rebuilding programs, and determine what additional data and analyses are needed to address those gaps.

Gaps in knowledge exist at many different points due to limitations in data and assessment methods, shortage of human resources and expertise, and analytical capabilities to integrate biological, economic and social data. Some of the knowledge gaps could be filled with additional data collection and analysis. Other knowledge gaps will likely remain unfilled because of finite resources and limits to the predictability of coupled human-natural systems (for example, the influence of climate change on fisheries). This type of gap requires robust strategies for managing with uncertainty, as mentioned below.

When data are insufficient to perform analytical stock assessments and estimate biomass and fishing mortality reference points with sufficient confidence to design and apply MSY-based control rules, alternative paradigms should be considered and evaluated. Strategies that combine spatial controls and habitat-based approaches with empirical rules to adjust harvest measures in response to demographic indicators or other proxies of stock status, as well as ecosystem-level indicators, could be designed to try and ensure that fishing rates are reasonable and precautionary and that rebuilding is progressing.

The success of any formal approach for developing robust control rules requires clearly-specified management objectives, so that quantitative performance measures and tradeoffs (e.g., between risks and yield) can be evaluated. While analyses generally consider uncertainties that affect population or ecosystem projections and future catch rates, most do not consider the full suite of risks in these complex and dynamic systems. Currently, the treatment of uncertainty is not integrated across the ecological, economic, and social dimensions of rebuilding, and the cumulative risk tradeoffs are not well understood. Consequently, it is not clear whether the necessary precaution (or too much precaution) is being applied.

In terms of assessing actual outcomes of rebuilding plans, the Committee focused its review on biological metrics, consistent with current legal mandates. These are available through regular stock assessments conducted for ongoing management. By contrast, information is not readily available to evaluate the broader impacts of rebuilding plans. Retrospective reviews of the socioeconomic impacts of rebuilding plans are rare, at least partially due to data availability. These socioeconomic impacts include changes in the structure of commercial fishing sector, economic returns, recreational values, fish processing industry, and culture of fishing communities. Methods exist and innovations are emerging in economic and social science approaches to characterize the breadth of economic and social impacts of rebuilding plans and factors in a coupled natural-human system that contribute to the success of these plans, although they have not yet been broadly applied, tested and refined to meet these information needs.

CONCLUSIONS

The current implementation of the MSFMCA relies on a prescriptive approach that has resulted in demonstrated successes in identifying and rebuilding overfished stocks. Fishing mortality has generally been reduced, and stock biomass has generally increased, for stocks that were placed under a rebuilding plan. Where they have been estimated, the long-term net economic benefits of rebuilding appear to be generally positive. Stocks that rebuilt or whose biomass increased appreciably were, in almost all cases reviewed, experiencing fishing mortalities below F_{MSY}, and often lower than 75% of F_{MSY}. More extreme reductions in target fishing mortalities have been implemented in situations in which rebuilding progress was slower than anticipated when the rebuilding plan was adopted, or the target year for rebuilding was approaching. In some cases rebuilding plans have failed to reduce fishing mortality as much as intended, either due to overestimation of stock sizes or implementation issues, and rebuilding has been slow or has not occurred.

The legal and prescriptive nature of rebuilding mandates forces difficult decisions to be made, ensures a relatively high level of accountability, and can help prevent protracted debate over whether and how stocks should be rebuilt. Setting rebuilding times is useful for specifying target fishing mortality rates for rebuilding and for avoiding delays in initiating rebuilding plans, which would otherwise require more severe management responses. However, the focus on trying to achieve a rebuilding target by a given time places unrealistic demands on the science, and forces reliance on forecasts and estimates of biomass-based reference points, which may be very uncertain. Emphasis on meeting fishing mortality targets rather than on exact schedules for attaining biomass targets may result in strategies that are more robust to assessment uncertainties, natural variability and ecosystem considerations, and less prone to rapid changes in management measures, which have social and economic impacts that may be more severe than more gradual changes. The choice between a rapid or gradual response involves tradeoffs between economic and social impacts and ecological/resource risks, which should be evaluated. The current approach is designed for the nations' most valuable, high-volume stocks, but over half of the nation's stocks have not been assessed and their status is unknown, rendering application of MSY-based control rules unrealistic. Alternate paradigms should be considered for these data-poor stocks.

The Committee offers comments on major issues of rebuilding with a long-term view at further improving the efficiency of the current approach to stock rebuilding. These issues directly or indirectly relate to the overarching issue of what is the appropriate balance between prescription and flexibility in stock rebuilding. Many of our comments could serve as suggestions for research and application to future revisions of National Standard Guidelines to improve the overall performance of stock rebuilding programs and thereby enhance the benefits derived from fisheries in the future.

1

INTRODUCTION

Challenges of Fishery Management Today

Fisheries provide a critical source of food and livelihood for millions of people. When managed properly, fisheries can augment the ecological, social, and economic goods and services that nations rely upon. However, while many countries are moving towards sustainable approaches to fisheries management, challenges still exist (Costello *et al.*, 2012). Committees convened by the National Academies have already provided reviews on a number of fisheries related issues including methods to improve fisheries stock assessments (National Research Council, 1998a; 1998b), commercial and recreational fisheries data collection and management (National Research Council, 2000a; 2006a), fisheries management (National Research Council, 1994a; 1999a), ecosystem based management (National Research Council, 1994b; 1999b; 2001, 2002a, 2006b), training and recruiting of fisheries scientists and social scientists (National Research Council, 2000b) and how fisheries science relates to the law (National Research Council, 2002b). But, rebuilding depleted fisheries is particularly challenging because it usually requires short-term sacrifices at a time when the fishing industry is already under pressure due to reduced yields and increasing costs. This management challenge is further complicated when data are poor or the system is poorly understood, when climate or habitat change, when ecosystem function and multispecies dynamics have to be considered, and when the socio-economic consequences to stakeholders need to be addressed. There are a growing number of reviews and interpretations of the causes and consequences for overexploited fisheries (Botsford *et al.*, 1997; Jackson *et al.*, 2001; Myers and Worm, 2003; 2005; Pauly and Maclean, 2003; Hilborn and Hilborn, 2012), but the reality is that this problem persists globally (FAO, 2012; CEA, 2012).

A number of countries have committed to ending overfishing through international agreements such as the United Nations Fish Stocks Agreement of 1995[1] and the 2002 World Summit on Sustainable Development[2], Smaller countries and regions have begun assessing the problem through regional efforts such the Global International Waters Assessment (GIWA) that the United Nations Environmental Program/Global Environment Facility (GEF) called for in 1999[3]. Effective management solutions to overfishing at the international and regional level remain elusive in part because strong economic and institutional barriers complicate management, and the diversity of fisheries (ranging from large-scale industrial pelagic and

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[1] http://www.un.org/depts/los/convention_agreements/texts/fish_stocks_agreement/CONF164_37.htm

[2] www.unmillenniumproject.org/.../131302_wssd_report_reissued.pdf

[3] http://www.unep.org/dewa/giwa/areas/reports/r23/giwa_regional_assessment_23.pdf

demersal fisheries to small-scale multi-species coastal fisheries) require different management approaches in order to be effective (CEA, 2012).

The Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA)[4] is the legislation that currently governs marine fisheries management in the United States. The Act is implemented by the Secretary of Commerce through the National Marine Fisheries Service and eight Regional Fisheries Management Councils. According to the most recent stock assessments of the National Marine Fisheries Service, approximately twenty percent of evaluated fisheries are overfished (NMFS, 2012). Overfished refers to a stock that is below the minimum stock size threshold, which is often based on maximum sustainable yield (MSY)-derived reference points. The fishing industry and other stakeholders in regions with overfished stocks are concerned about the effects of the mandate to rebuild on their livelihoods, leading to a request for a National Academies' review and analyses of the Rebuilding Plans required by the MSFCMA and their success.

STATE OF FISHERIES TODAY

Global fisheries

The Food and Agriculture Organization of the United Nations (FAO, 2012) recently reported that "[c]apture fisheries and aquaculture supplied the world with about 148 million [metric tons] of fish in 2010 (with a total value of US$217.5 billion), of which about 128 million [metric tons] was utilized as food for people, and preliminary data for 2011 indicate increased production of 154 million [metric tons], of which 131 [metric tons] was destined as food…" Aquaculture fisheries make up approximately 31% of this weight and more than half the value, reflecting that sector's growth in a global effort to meet demand while harvest levels for capture fisheries have leveled off at approximately 90 million metric tons since the 1980s (FAO, 2012).

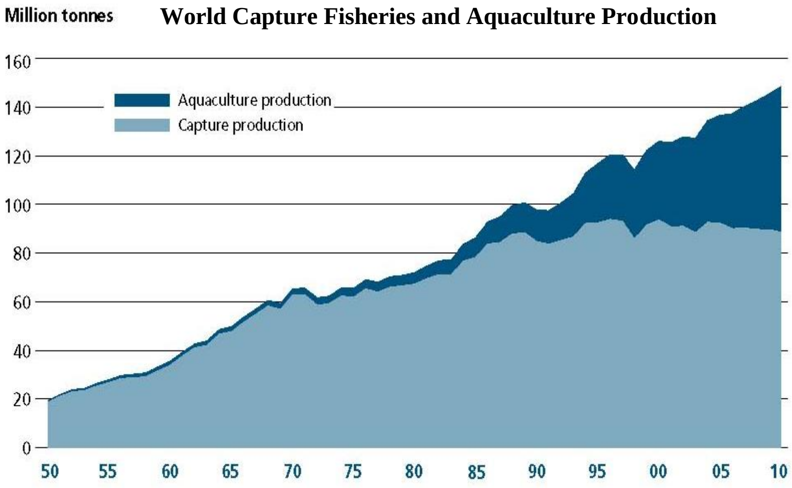

FIGURE 1.1: Global productions of capture fisheries and aquaculture in metric tons. SOURCE: FAO, 2012.

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[4] 16 U.S.C. §§1801-1884

U.S. Fisheries

According to FAO estimates, the United States had the fourth highest capture fishery landings behind China, Peru, and Indonesia in 2011, the most recent year for which global data are available (FAO, 2012). Figure 1.2 provides the volume of commercial landings in the United States for the past 50 years. Most notable is the nearly 30% jump in reported U.S. landings from the late 1980s to the mid-1990s. This jump primarily corresponds to the increased landings of pollock in the U.S. by joint venture and later fully-Americanized fisheries off Alaska (see chapter 2 for more detail). The pollock fishery was and remains the largest volume fishery in the United States[5].

The total U.S. commercial landings were 4.5 million metric tons valued at $5.3 billion in 2011 (NOAA, 2013). Of that, 3.6 million metric tons were edible finfish and shellfish, and the remaining 884,052 metric tons were caught for reduction and other industrial uses (NOAA, 2013). The largest catches by volume in 2011 were contributed by pollock, menhaden, salmon, flatfish (excluding halibut), and cod. In terms of value of production, the top five groups were crabs, salmon, scallops, shrimp, and lobster (NOAA, 2012).

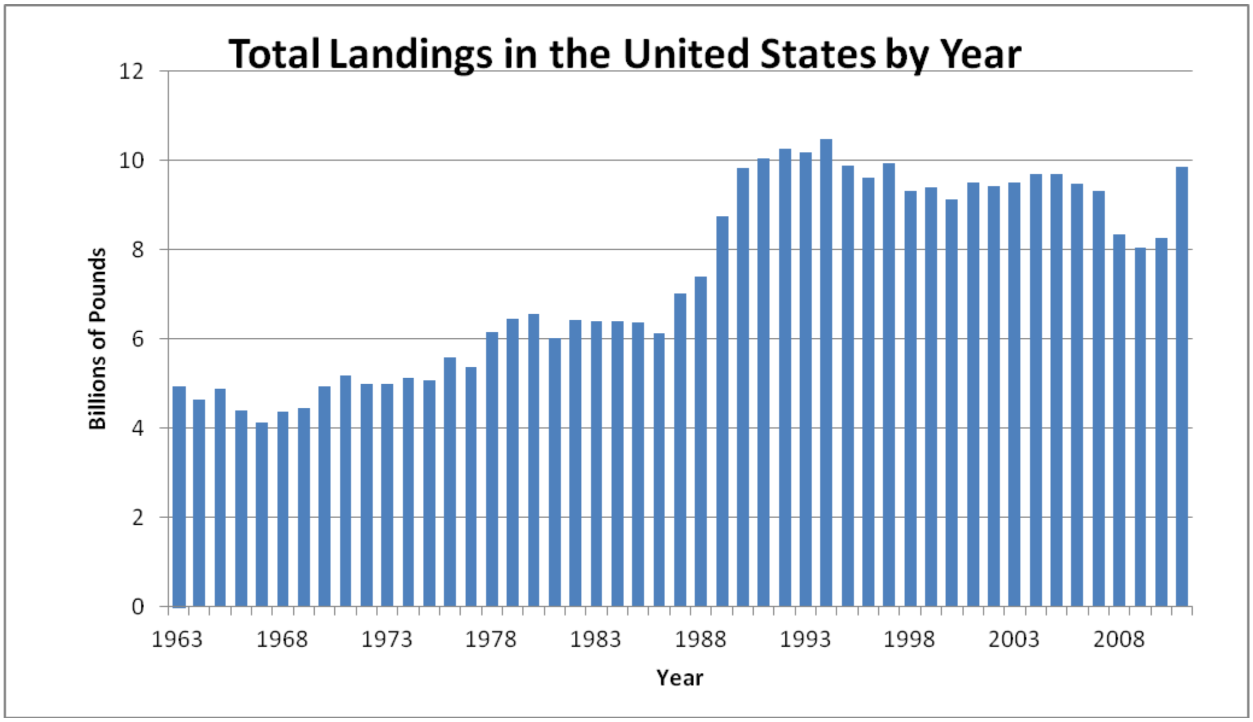

FIGURE 1.2: Volume of U.S. Domestic Commercial Landings over the past 50 years.
SOURCE: NOAA data compiled by committee.

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[5] http://www.st.nmfs.noaa.gov/commercial-fisheries/commercial-landings/annual-landings-with-group-subtotals/index

Of the United States, Alaska accounted for both the highest weight and the highest value of production, followed by Louisiana, Virginia, California, and Washington, in terms of weight, and by Massachusetts, Maine, Louisiana, and Washington, in terms of value (NOAA, 2012).

In 2011, U.S. consumers spent an estimated $83.4 billion on fishery products, which includes $56.5 billion at food service establishments and nearly $25.7 billion in retail sales for home consumption. The remaining 1.3 billion was spent on industrial fish products. (NOAA, 2012)

The large social and economic impacts of recreational fisheries are very important. The National Oceanic and Atmospheric Administration (NOAA) estimates that in 2011, domestic[6] recreational fisheries accounted for 69 million fishing trips (NOAA, 2012). Recreational catches are relatively minor in weight overall, but for some species, such as red drum and spotted seatrout, they exceed commercial catches.

Combined, U.S. commercial and recreational fisheries generated $166 billion in sales impacts, contributed nearly $44 billion to the Gross National Product and supported 1.4 million jobs in the fishing sector and across the broader economy (NOAA, 2012).

CONTEXT FOR THIS REPORT

Concern over food security and the maintenance of historic fisheries paired with increasing demand on fisheries resources has caused many nations to develop rebuilding strategies. At the World Summit on Sustainable Development, many governments committed to "[m]aintain or restore stocks to levels that can produce maximum sustainable yield with the aim of achieving these goals for depleted stocks on an urgent basis and where possible not later than 2015." Prior to the 1970s, federal management of marine fisheries in the United States was minimal. The first major piece of federal legislation to govern marine fisheries went into effect on March 1, 1977 and was known then as the Fishery Conservation and Management Act (FCMA)[7]. The Magnuson-Stevens Fisheries Conservation and Management Act (MSFCMA or "the Act"), as it is now known, has undergone many important changes through its history. Originally intended to reduce foreign fishing in U.S. waters, the MSFCMA has become increasingly focused on conservation as the declining state of fish stocks has become more apparent. In particular, the 1996 amendments of the Act[8] required that fishery managers develop plans to rebuild overfished fish stocks and that, where possible, the time frame for rebuilding not exceed 10 years (see Chapter 2). The 2006 Amendment of the Act[9] added additional requirements such as ending overfishing immediately, annual catch limits and accountability measures.

Still, despite these requirements, some fish stocks continue to be overfished and some have not rebuilt. Efforts to end overfishing and rebuild stocks have been accompanied by economic and social impacts that some stakeholders consider unreasonable and/or unnecessary. In light of these issues, the MSFCMA and the related rebuilding plans continue to be scrutinized,

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[6] Including fisheries in the continental United States, Hawaii, Alaska, and Puerto Rico.
[7] FCMA, Pub. L. No. 94-265, 90 Stat. 331 (1976).
[8] Sustainable Fisheries Act, Pub. L. No. 104-297, 110 Stat. 3559 (1996).
[9] Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006, Pub. L. 109-479, 121 Stat. 3575 (2006).

and their effectiveness re-evaluated. In 2010, in a letter signed by United States Senator Olympia Snowe and United States Representative Barney Frank, Congress requested that the National Oceanic and Atmospheric Administration (NOAA) fund a study by the National Academies (NAS) regarding the MSFCMA's rebuilding requirements. Based on the letter request from Congress, and with further input from NOAA, the NAS developed a statement of tasks (Box 1.1).

Box 1.1
Statement of Task

1. Evaluate methods and criteria used (1) to set target fishing mortality and biomass levels for rebuilding overfished stocks, and (2) to determine the probability that a particular stock will rebuild by a certain date. Consider the quantity and quality of information available for defining maximum sustainable yield (MSY)-based reference points or proxies for such reference points. Compare these methods and criteria to those used in major fishery management settings outside the U.S.

2. Assess the effects of uncertainty in current stock abundance, population dynamics, and variability in recruitment in setting rebuilding targets. Identify criteria for adjusting rebuilding targets and schedules based on new information and updated stock assessments.

3. Provide an overview of the success of rebuilding plans under the MSA and compare to success of approaches used outside the U.S. Using a few representative rebuilding plans, identify factors (such as fishing mortality rate, life histories, uncertainty in stock assessments, and others) that affect the timeframe over which a stock is rebuilt.

4. Consider the effects of climate and environmental conditions, habitat loss and degradation, ecological effects of fishing on the food chain, and ecological interactions among multiple species, and identify ways to adjust rebuilding plans to take these factors into account.

5. Assess the types of information needed and current understanding of the economic and social impacts of rebuilding programs, particularly on fishing communities. Identify the economic, social, and ecological tradeoffs of rebuilding a fishery associated with shorter or longer rebuilding times. Evaluate available methods for integrating these social, economic and ecological factors when designing and evaluating rebuilding plans.

6. Summarize how the social, economic and ecological impacts of rebuilding plans are affected by the structure of fisheries management measures, e.g., limited entry, catch shares systems, and closed areas.

7. Identify the biological, ecological, social and economic knowledge gaps that impede the implementation and effectiveness of rebuilding programs, and determine what additional data and analyses are needed to address those gaps.

With these tasks in mind, the NAS formed a committee to develop this report. The committee consisted of eleven expert scientists from diverse scientific backgrounds and broad experience in different national and international fisheries. Included in the committee were experts on fisheries management, fisheries science, biological oceanography, ecosystem-based management, environmental policy, economics, and applied mathematics. In response to Congress's inquiries regarding the rebuilding requirements of the MSFCMA, and based on the statement of task, the committee put together this final report.

REPORT ORGANIZATION

The report is divided into seven chapters. The goal of this chapter is to provide an introductory overview of the challenges of fisheries management, overfishing and rebuilding. A brief synopsis of the current state of global and domestic fisheries is given, followed by the origins and context for the report.

Chapter 2 explores the evolution and rationale of the MSFCMA from its origins in 1976 as the Fishery Conservation and Management Act of 1976, its subsequent amendments that introduced rebuilding requirements and accountability into the management of the nation's fisheries, and the guidelines for rebuilding fish stocks.

Chapter 3 provides a detailed and technical review of current federally implemented rebuilding plans, and the outcomes of those plans in terms of trends in fishing pressure and stock size, and changes in stock status with respect to overfishing thresholds and biological reference points. It provides detailed information on a subset of stocks assessed by quantitative fish stock-assessment methods, and discusses what progress has been made in rebuilding those stocks to date. Finally, although strict comparability among regions is not possible given the different realities of fisheries and fishery management institutions, Chapter 3 provides a brief review of rebuilding approaches and outcomes in a few other countries and regions in order to place the U.S. situation within an international perspective.

Chapter 4 discusses the technical considerations associated with implementing a rebuilding plan. The chapter discusses the probabilities of meeting rebuilding deadlines as well as the challenges and issues with incorporating uncertainty and using the "best available science." Chapter 4 also addresses data poor stocks and the challenges associated with the Maximum Sustainable Yield (MSY) paradigm. Finally, it presents a series of diverse case studies that illustrate the range of issues, challenges and outcomes with implementing rebuilding plans for domestic and international stocks.

Chapter 5 introduces the ecological factors that are or may be incorporated into rebuilding plans. It provides a discussion of ecosystem-based management approaches and the challenges that impact rebuilding efforts. In the context of fisheries management and rebuilding plans, Chapter 5 also addresses climate change and shifting baselines, habitat loss, and ecological interactions.

Chapter 6 focuses on "the human dimension" of fisheries rebuilding. It considers the socio-economic aspects of fisheries, and discusses methods of evaluating the social and economic outcomes of fisheries management. Chapter 6 also incorporates a section on the role of governance and markets in potential socio-economic outcomes of rebuilding plans. The chapter concludes with a few illustrative case studies.

Chapter 7 takes a strategic look into the future and considers some of the issues that current practitioners and managers are challenged by—including shrinking resources, prescriptive constraints, defining rebuilding success, rebuilding under Ecosystem Based Fisheries Management, mixed stocks, and data limitations.

Chapter 2

U.S. FISHERIES MANAGEMENT AND THE LAW

HISTORY OF U.S. FISHERIES MANAGEMENT

Until March 1, 1977, marine fisheries management in the United States (U.S.) was minimal (Magnuson, 1977). This was the effective date of the Fishery Conservation and Management Act of 1976 (FCMA as it was known then).[1] Prior to the FCMA, management of marine fisheries in the U.S. was generally limited to controls implemented by individual states in waters within their respective jurisdictions, pursuant to the Submerged Lands Act of 1953, which provided states with jurisdiction over submerged lands and natural resources within three miles of their respective coastlines (Magnuson, 1977).[2] The Atlantic States Marine Fisheries Commission (ASMFC) was formed through an interstate compact and approved by Congress in 1942[3] to coordinate state fisheries management, but it lacked direct management authority. Similar interstate commissions developed between Pacific States in 1947[4] and in the Gulf of Mexico in 1949.[5]

In addition to state management of marine fisheries, the federal government managed some fisheries under the auspices of international fishery management organizations such as the International Commission for Northwest Atlantic Fisheries Organization (ICNAF, now the North Atlantic Fisheries Organization),[6] the International Pacific Halibut Commission (IPHC),[7] and the Inter-American Tropical Tuna Commission (IATTC).[8]

ICNAF was founded in 1949[9] because of increasing fishing pressure on stocks in international waters. At the time, the U.S. and Canada were the main participants in the fishery. With arrival of distant water fishing vessels from Europe (primarily U.S.S.R., East and West Germany, Poland, Spain, Portugal) and Asia (primarily Japan) in the 1960s, ICNAF became more active. In particular, it needed to respond to the collapse of Georges Bank haddock. Overfishing of other groundfish, silver hake, herring, and mackerel followed. By the mid-1970s, ICNAF had established a Total Allowable Catch (TAC) and national allocations for all of the

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[1] Fishery Conservation and Managemnt Act (FCMA), Pub. L. No. 94-265, 90 Stat. 331 (1976).
[2] Submerged Land Act, Pub. L. No. 83-31 § 3-4, 67 Stat. 29, 30 (1953), 43 U.S.C. §§ 1311-1312 (2012).
[3] Pub. L. No. 77-539, 56 Stat. 267 (1942); 16 U.S.C. § 4107(c)(1) (2012).
[4] Pub. L. No. 80-232, 61 Stat. 419 (1947) ; 16 U.S.C. § 4107(c)(2) (2012).
[5] Pub. L. No. 81-66, 63 Stat. 70 (1949); 16 U.S.C. § 4107(c)(3) (2012).
[6] http://www.nafo.int/; www.nafo.int/about/frames/hist-early.html; http://journal.nafo.int/J23/anderson.pdf
[7] http://www.iphc.int/publications/pamphlet/1IPHCHistoryPage.pdf
[8] http://www.iattc.org/pdffiles/iattc_convention_1949.pdf
[9] http://www.nafo.int/

targeted stocks and an overall TAC for all stocks combined. For example, the 1975 overall TAC (850,000 metric tons) was lower than the sum of the individual TACs (1,053,000 metric tons) to take account of biological interactions between the stocks. By 1977, the overall TAC was reduced to 525,000 metric tons to take account of technical interactions (i.e., bycatch) and to prevent overfishing of some vulnerable stocks (Brown *et al.,* 1979). Biological and technical interactions are discussed below (Anderson, 1998).

Arguably ICNAF was making progress in bringing overfishing under control. For example, following the initiation of TAC management of Georges Bank haddock beginning in 1972, there was a large year-class produced in 1975 with the potential to rebuild the fishery by the late 1970s (Clark *et al.,* 1982).

However, the damage had already been done, and public opinion (because of distant water fishing off Alaska and the west coast, as well as the northwest Atlantic) demanded that the U.S. extend its jurisdiction to 200 nautical miles (Congressional Research Service, 1976). In 1976, as part of the FCMA, the U.S. claimed exclusive fishery management authority over those waters contiguous to its territorial sea and extending 200 nautical miles from its shoreline.[10] This was originally referred to as the FCZ (Fishery Conservation Zone). Later, however, the U.S. extended its claim to include jurisdiction over other economic activity, and the zone was renamed the Exclusive Economic Zone (EEZ).[11]

In nearly four decades since the FCMA was adopted by Congress, U.S. fishery management law has evolved. The FCMA has been reauthorized and amended several times. There have been three important phases in the Act's history, which are marked by the initial passage of the Act in 1976,12 and subsequent amendments in 1996 (also known as the Sustainable Fisheries Act or SFA) and 2006. We refer to these phases as "Americanization," "Rebuilding," and "Accountability." While the three phases are discussed in more detail below, Table 1.1 summarizes some important elements of each. In addition to undergoing substantive changes, the Act was renamed twice. In 1980, the Act was renamed the Magnuson Fishery Conservation and Management Act (MFCMA) to honor Senator Warren Magnuson for his contributions to the Act.[13] The Act was given its most recent title, the Magnuson-Stevens

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[10] FCMA, Pub. L. No. 94-265, 90 Stat. 331 (1976); 16 U.S.C. § 1811-12 (2012). Interestingly, several other countries (particularly in Latin America) had already extended their jurisdiction to 200 miles because of concern about U.S. tuna fishing off their coasts (Nandan, 1987). In the case of tuna fishing, the U.S. objected to these extensions of jurisdiction. In fact, the extension of U.S. fishing jurisdiction to 200 miles initially excluded highly migratory species such as tunas. FCMA, Pub. L. No. 94-265, 90 Stat. 331 (1976). The law was amended in 1990 to make the EEZ applicable to highly migratory species subject to international treats ratified by Congress. Fishery Conservation Amendments of 1990, Pub. L. No. 101-627, 104 Stat. 4436-39 (1990).

[11] In 1983, President Ronald Reagan, through Presidential Proclamation, asserted jurisdiction over an Exclusive Economic Zone extending 200 nautical miles from the shoreline. The proclamation stated that, "[w]ithin the Exclusive Economic Zone, the United States has, to the extent permitted by International Law, (a) sovereign rights for the purpose of exploring, exploiting, conserving and managing natural resources, both living and non-living, of the seabed and subsoil and the superjacent waters and with regard to other activities for the economic exploitation and exploration of the zone…" Exclusive Economic Zone of the United States of America, Proclamation No. 5030, 48 Fed. Reg. 10605 (March 10, 1983).

[12] FCMA, Pub. L. No. 94-265, 90 Stat. 331 (1976).

[13] Salmon and Steelhead Conservation and Enhancement Act of 1980, Pub. L. No. 96-561 § 238, 94 Stat. 3300 (1980).

Fishery Conservation and Management Act (MSFCMA) in 1996 to acknowledge the influence of Senator Ted Stevens of Alaska.[14]

TABLE 2.1: Phases of the MSFCMA.

Phases	**Time Period**	**Important Elements**
Americanization	1977-1995	• Extended Jurisdiction to 200 miles • Created the objective as Optimum Yield (OY) • Required fishery management plans (FMP) in accordance with National Standards • Established co-management between eight Regional Fishery Management Councils and the Federal Government • Made provisions for foreign fishing to continue off the USA until fisheries were Americanized
Rebuilding	1996-2006	• Changed definition of OY to make $F \geq F_{\mathrm{MSY}}$ overfishing • Required overfished stocks to be rebuilt • Limited the rebuilding time to 10 years with exceptions
Accountability	2007-present	• Called for overfishing to end immediately • Required annual catch limits (ACLs) • Required accountability measures if ACLs are exceeded • Strengthen the role of Scientific and Statistical Committees (SSCs).

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[14] Pub. L. No. 104-208 §§211(a)-(b) (1996).

Phases of the MSFCMA

Americanization

The first phase of the Act is referred to as the “Americanization” phase because one of the original objectives of the legislation was to reduce the prominence of foreign fishing off the United States’ coasts. This Americanization is evidenced by the significant expansion of U.S. jurisdiction over fisheries as far as 200 nautical miles from shore.

When the FCMA was passed, it was largely seen as a way of excluding distant water fleets and allowing fisheries of the U.S. to be “Americanized.”[15] Many stakeholders and members of Congress did not believe it was necessary to regulate U.S. fisheries to any significant degree. While in 1976, this may have been true in the short-term for Alaska and the west coast, it was not true for the Northeastern United States. ICNAF had already prohibited significant foreign fishing for the most important species targeted domestically off the northeastern U.S., and there was already more U.S. fishing capacity than target stocks like cod, haddock, and yellowtail flounder could sustain. For example, by the late 1970s, the TAC for Georges Bank haddock was quickly exceeded by U.S. fishing vessels.

There are five general resolutions encompassed in the FCMA. First, the Act demarked a geographic zone adjacent to the United States’ shoreline in which the U.S. government has jurisdiction over fishery resource management. Second, the Act was designed to further conservation and to establish optimum yields for fishery resources, taking into account social and economic factors. Third, the Act promotes the harvest and processing of fishery resources by U.S. fishers and companies. Fourth, the Act was to establish an institutional framework and an enforcement authority to carry out the implicit and explicit objectives of the Act. Finally, the Act was developed to ensure management of fisheries is based on the best scientific information available (National Research Council, 1994a).

The Act also articulated more specific goals for fisheries management through the promulgation of seven national standards in Title III, Section 301 (a) (see Box 2.1).[16] Given that this report is evaluating the effectiveness of rebuilding plans, it is primarily concerned with National Standard 1, but National Standards 2 and 8 are also important considerations. However, all National Standards are relevant because fishery management plans, including rebuilding plans, must adhere to them.

The FCMA established eight regional fishery management councils (RFMCs) designated to cover the following geographic areas (See Box 2.2 and Figure 2.1): New England, Mid-Atlantic, South Atlantic, Caribbean, Gulf of Mexico, Pacific, North Pacific, and Western Pacific Fishery Management Councils.[17] RFMCs are responsible for preparing fishery management plans (FMPs) for achieving Optimum Yield and satisfying the National Standards.[18] The Federal Government reviews the FMPs to assure they comply with the National Standards and other

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[15] H.R. Rep. No. 94-445, at 1, 2, 24-36, 42, 43 (1975);
http://www.nmfs.noaa.gov/stories/2011/20110411roadendoverfishing.htm

[16] FCMA, Pub. L. No. 94-265 § 301(a), 90 Stat. 331, 346 (1976).; Sustainable Fisheries Act, Pub. L. No. 104-297 § 106, 110 Stat. 3559, 3570 (1996).

[17] FCMA , Pub. L. No. 94-265 § 302(a), 90 Stat. 331, 347 (1976).

[18] FCMA , Pub. L. No. 94-265 § 302(h), 90 Stat. 331, 350 (1976).

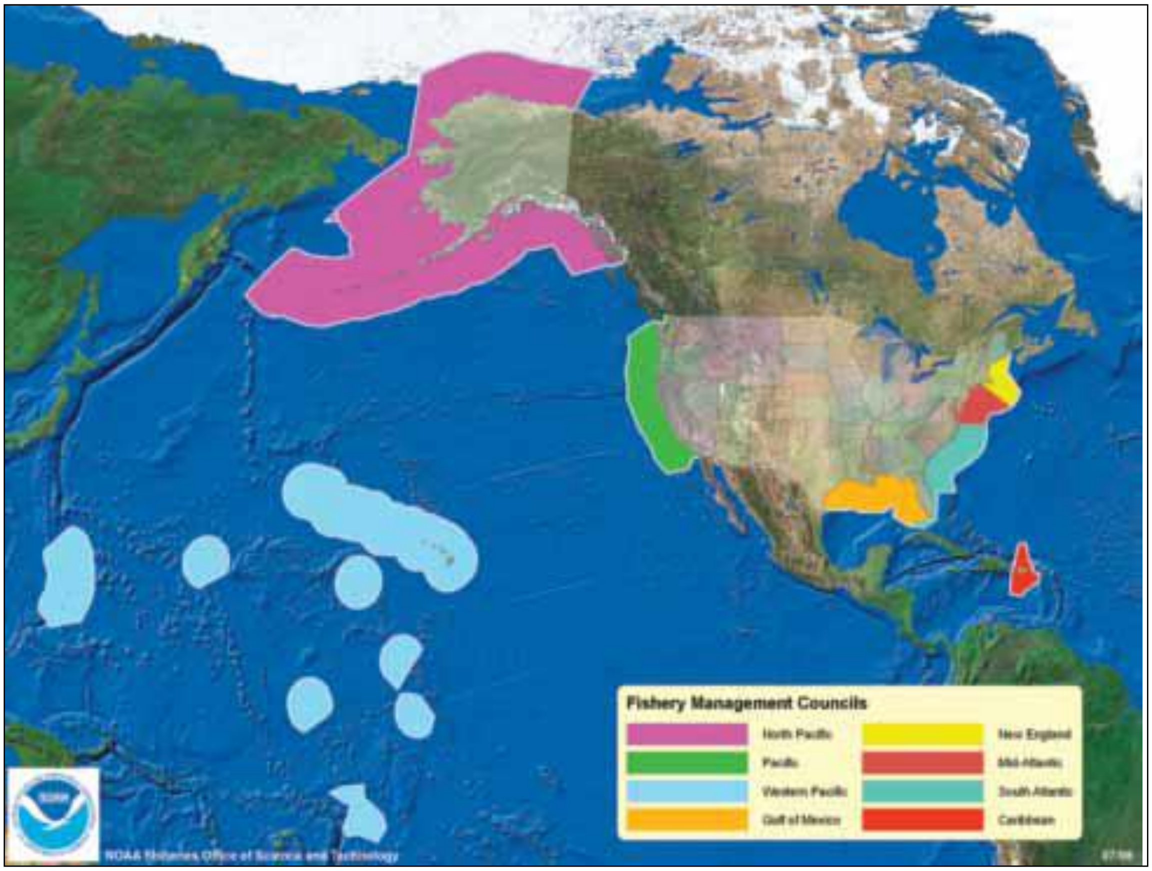

FIGURE 2.1: Map of the Jurisdiction of each U.S. Fishery Management Council
SOURCE: http://www.fisherycouncils.org/USFMCsections/USRFMCintro.pdf

applicable Federal law.[19] If they do, the Plans are approved and the federal government is responsible for implementing them, including monitoring of the fisheries and enforcement. This Committee considers the councils to be an imaginative form of co-management involving the federal government, state governments (i.e., state officials are council members) and citizen stakeholders.[20]

In addition to the eight RFMCs, the Highly Migratory Species Division of the National Marine Fisheries Service is responsible for managing Atlantic highly migratory species (HMS) as a result of the 1990 Amendment of the MFCMA.[21] HMS are tunas and tuna-like species (e.g., billfish including swordfish and marlins) and most sharks species. The HMS Division implements measures recommended by International Commission for Conservation of Atlantic Tunas (ICCAT). The public input is obtained from the HMS Advisory Committee. Management is promulgated under the Consolidated HMS Fishery Management Plan (2006) and its amendments.[22] The Plan is the umbrella for implementing ICCAT recommendations as well

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[19] FCMA , Pub. L. No. 94-265 §§ 301(a), 304(a), 90 Stat. 331, 346, 352 (1976).
[20] FCMA , Pub. L. No. 94-265 § 302-04, 90 Stat. 331, 347-51 (1976).
[21] Fishery Conservation Amendments of 1990, Pub. L. No. 101-627, § 103 (1990).
[22] http://www.nmfs.noaa.gov/sfa/hms/hmsdocument_files/FMPs.htm

as domestic management of non-ICCAT highly migratory species (e.g., sharks) and other measures required by U.S. law (e.g., requirements of the Endangered Species Act and Marine Mammal Protection Act).

The FCMA effectively reduced foreign fishing within the United States' EEZ from approximately 60% of the commercial catch in 1981 to approximately 1% in 1991. Meanwhile, domestic fisheries grew. Foreign fishing in the U.S. EEZ is insignificant today although there is some foreign ownership of U.S. fishery enterprises (National Research Council, 1994a).

While the Act was successful at Americanizing the fisheries, many problems persisted. Most notably, overfishing was a serious problem in some regions (e.g., New England), but not all (e.g., stocks under the jurisdiction of the North Pacific FMC). According to Parsons (1993), U.S. fisheries management was problematic because of "continued overfishing of some stocks; lack of coordination between councils and the NOAA/National Marine Fisheries Service in setting research agendas; conflicts among users; the vulnerability of the fishery management process to delays and political influence; lack of accountability; inconsistency in state and federal management measures; and adoption of unenforceable management measures."

Some of the problems with management under the FCMA during the Americanization period were growing pains. An entirely new system of co-management needed to be put in place. In addition to growing pains, fisheries management under the FCMA suffered from confused or conflicting objectives. The problem related to the definition of Optimum Yield, which was:

> "…the amount of fish –
> (A) which will provide the greatest overall benefit to the Nation, with particular reference to food production and recreational opportunities; and
> (B) which is prescribed as such on the basis of the maximum sustainable yield from such fishery, as modified by any relevant economic, social, or ecological factor." (National Research Council, 1994a)

This definition of OY has been criticized on policy and technical grounds. In terms of policy, the NRC (1994a) said "Unfortunately, this definition is so broad that it can be used to justify almost any quantity of catch." This issue was addressed by the 1996 Amendment of the Act[23] as described below. From a technical perspective, Sissenwine (1978) questioned if MSY is an adequate basis for OY because of species interactions, environmental variability and other factors. Technical issues are also considered below.

During this period of implementation of the FCMA, the U.S. Secretary of Commerce issued guidelines, known as "602 guidelines," to help interpret Optimum Yield and encourage conservation. The guidelines called for quantitative (or measurable) definitions of overfishing. In the discussion of overfishing, the 602 guidelines highlighted the need to avoid "recruitment overfishing."[24] Recruitment overfishing is generally understood to mean avoiding reductions in spawning stock size that jeopardize future recruitment. However, the 602 guidelines lacked a precise scientific or legal definition, which meant the occurrence of recruitment overfishing was

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[23] Sustainable Fisheries Act, Pub. L. No. 104-297, 110 Stat. 3559 (1996).

[24] Guidelines for the Preparation of Fishery Management Plans Under the FCMA, 50 C.F.R. Part 602 (1989).

usually debatable. The concept of recruitment overfishing focuses on fishing activities and stock responses to be avoided, not on targets or objectives such as MSY and/or OY.

Another provision of the 602 Guidelines was a limited exception to the requirement to prevent overfishing. The exception states:

> "Harvesting the major component of a mixed fishery at its optimum level may result in the overfishing of a minor (smaller or less valuable) stock component in the fishery. A council may decide to permit this type of overfishing if it is demonstrated by analysis (paragraph (f)(5) of this section) that it will result in net benefits to the Nation, and if the council's action will not cause any stock to require protection under the Endangered Species Act."[25]

This exception provided flexibility to generate net benefits even if it meant sacrificing long-term yield of some species so long as no species was at risk of extinction. However, this provision could be abused, and it was modified in subsequent guidelines. As is discussed later in the report, while this loss of flexibility may prevent abuse, it may also result in a substantial loss of potential sustainable yield.

The 602 guidelines also required Stock Assessment and Fishery Evaluation (SAFE) reports to document the performance of fishery management. However, the National Research Council (1994a) stated that "[t]he implementing regulations, known as the '602 guidelines,' do not provide the specification and guidance needed."

It is also noteworthy that the National Research Council (1994a) recommended, "…ensuring that harvest does not reduce stock abundance below levels that can sustain maximum yields over the long term. For currently overfished stocks, harvest levels must allow rebuilding the stock over specified periods of time to a level that can support sustainable maximum yields." We do not know if this recommendation influenced Congress, but the Act was amended in 1996 along the lines that the National Research Council recommended.

Rebuilding

The 1996 amendments to the Act were made by the Sustainable Fisheries Act (SFA).[26] In addition to changing the title of the Act to the Magnuson-Stevens Fishery Conservation and Management Act, the 1996 amendment made many important and substantive changes as well. In addition to the three new national standards, the most important aspects of the 1996 amendments were:

- A change in the definition of optimum yield from MSY as "modified" by ecological, economic and social factors, to as "reduced" by these factors.[27]
- The requirement rebuilding of overfished fisheries[28], and

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[25] Guidelines for the Preparation of Fishery Management Plans Under the FCMA, 50 C.F.R. § 602.11(c)(8) (1989).

[26] Sustainable Fisheries Act, Pub. L. No. 104-297, 110 Stat. 3559 (1996).

[27] Sustainable Fisheries Act, Pub. L. No. 104-297, § 102(7), 110 Stat. 3559, 3562 (1996).

[28] Sustainable Fisheries Act, Pub. L. No. 104-297, §§ 108(a)(1), (7), 110 Stat. 3559, 3574, 3575 (1996).

- The requirement to identify of Essential Fish Habitat (EFH)[29] and to take steps to conserve it. Requirements with respect to EFH are considered in Chapter 5.

Specifically, the SFA defined Optimum Yield from a **fishery** (emphasis added) as:

> "…the amount of fish which—
> (A) will provide the greatest overall benefit to the Nation, particularly with respect to food production and recreational opportunities, and taking into account the protection of marine ecosystems;
> (B) is prescribed on the basis of the maximum sustainable yield from the fishery, as *reduced* by any relevant social, economic, or ecological factor; and
> (C) in the case of an overfished fishery, provides for rebuilding to a level consistent with producing the maximum sustainable yield in such fishery." (emphasis added)[30]

In addition, the SFA states that:

> ''… The terms 'overfishing' and 'overfished' mean a rate or level of fishing mortality that jeopardizes the capacity of a fishery to produce the maximum sustainable yield on a continuing basis."[31]

In another section labeled "Overfishing," the Act describes the requirements for rebuilding overfished fisheries. It requires the Secretary of Commerce to report annually on fisheries that are overfished or approaching the condition of being overfished. For these fisheries, within one year, the appropriate RFMC must develop a Fishery Management Plan (FMP):

> "(A) to end overfishing in the fishery and to rebuild affected stocks of fish; or
> (B) to prevent overfishing from occurring in the fishery whenever such fishery is identified as approaching an overfished condition."[32]

Furthermore,

> ''For a fishery that is overfished, any fishery management plan, amendment, or proposed regulations prepared ….shall— (A) specify a time period for ending overfishing and rebuilding the fishery that shall—
>
> i) *be as short as possible*, taking into account the status and biology of any overfished stocks of fish, the needs of fishing communities, recommendations by international organizations in which the United States participates, and the interaction of the overfished stock of fish within the marine ecosystem; and
> ii) *not exceed 10 years*, except in cases where the biology of the stock of fish, other environmental conditions, or management measures under an international agreement in which the United States participates dictate otherwise;…." (emphasis added)[33]

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[29] Sustainable Fisheries Act, Pub. L. No. 104-297, §§ 101(1) ,(2), (7) ,102(3),108(a)(3), 110 Stat. 3559, 3560, 3561, 3574, 3575 (1996).

[30] Sustainable Fisheries Act, Pub. L. No. 104-297, § 102(7), 110 Stat. 3559, 3562 (1996).

[31] Sustainable Fisheries Act, Pub. L. No. 104-297, § 102(8), 110 Stat. 3559, 3562 (1996).

[32] Sustainable Fisheries Act, Pub. L. No. 104-297, § 109(e), 110 Stat. 3559, 3584 (1996).

[33] Sustainable Fisheries Act, Pub. L. No. 104-297, § 109(e), 110 Stat. 3559, 3584 (1996).

The SFA also requires the U.S. Secretary of Commerce to review rebuilding plans at routine intervals not exceeding two years and to take immediate action to revise plans when "adequate progress" is not being made.[34]

The SFA profoundly changed U.S. marine fisheries management. It shifted the emphasis from avoiding undesirable conditions (e.g., recruitment overfishing) to achieving high long-term yields on a sustainable basis (MSY). It attempted to put "teeth" in the law when it came to stopping overfishing and rebuilding fisheries. Despite this improvement, however, the law still required legal and scientific interpretation with respect to several of its provisions.

Guidance documents developed by the National Marine Fisheries Service (NMFS) in 1998, known as the National Standard 1 Guidelines (NS1G), replaced the 602 guidelines. The NS1G provided additional clarification as to how the new provisions should be implemented. In particular, the term "fishery" was interpreted in the NS1G as meaning a stock of fish rather than a group of fishing operations targeting a similar (assemblage of) species, using similar gear, during the same period of the year and/or within the same area. The NS1G also clarified that the terms "overfished" and "overfishing" are used to describe biomass and mortality rate, respectively. The NS1G stated that the biomass (B) level that defines "overfished" is a level lower than B_{MSY}, the biomass at which Maximum Sustainable Yield (MSY) is achieved, and that the minimum, or default, biomass indicative of an overfished stock is half of the biomass associated with maximum sustainable yield (½ B_{MSY}).[35] Furthermore, the NS1G allowed for rebuilding times in excess of 10 years in cases where the probability of rebuilding within 10 years, with zero fishing mortality, is less than 50%. In these limited cases, the allowable rebuilding time was the time necessary to rebuild with a 50% probability given zero fishing mortality plus the mean generation time of the species.[36]

The Guidelines also called for FMPs to specify an "MSY Control Rule" that characterized a fishing mortality strategy to achieve the maximum long term average yield (i.e., MSY). The control rule defined overfishing and overfished levels. In practice, the fishing mortality (F) strategy was generally to maintain a constant F strategy at or below F_{MSY}, unless a rebuilding plan was required. However, the Guidelines were flexible enough to allow F to exceed F_{MSY} for a period of time so long as the stock's long-term capacity to produce MSY was not jeopardized and the rebuilding objective was expected (typically with a probability of 0.5) to be achieved.[37]

As with the Americanization phase, there were growing pains during the rebuilding phase of MSFCMA implementation. Rebuilding plans were developed and implemented, and several stocks were rebuilt (e.g., Georges Bank scallops). By 2006, 10 stocks that had been declared overfished had been rebuilt. However, overfishing of some stocks continued, and some stocks were rebuilt more slowly than expected or not at all.[38]

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[34] Sustainable Fisheries Act, Pub. L. No. 104-297, § 109(e), 110 Stat. 3559, 3584 (1996).

[35] Separate technical guidelines (Restrepo *et al.*, 1998) indicated that it should be higher than ½ B_{MSY} for most species. In practice, ½ B_{MSY} became the most common specification of the overfished threshold level.

[36] Magnuson-Stevens Act Provisions, National Standard Guidelines, 63 Fed. Reg. 24212, 24231 (May 1, 1998).

[37] Magnuson-Stevens Act Provisions, National Standard Guidelines, 63 Fed. Reg. 24212, 24229-24231 (May 1, 1998).

[38] http://www.nmfs.noaa.gov/sfa/statusoffisheries/2006/2006RTCFinal_Report.pdf

Accountability

The accountability phase of management under the MSFCMA is just beginning. The 2006 Amendment to the MSFCMA made three important changes relevant to the Committee's Statement of Tasks:

- It strengthens the role of scientific advice when it comes to conserving stocks and avoiding overfishing. The scientific advice usually comes from the RFMCs' Scientific and Statistical Committees (SSCs) as required by the Act, but it allows for advice from another "peer review process" established by the Secretary of Commerce or a council.[39] In practice, peer review processes for stock assessments exist in most regions of the country and the results of these processes are used as input to SSCs (Sissenwine and Rothschild, 2011).
- It requires Fishery Management Plans to end overfishing immediately, although Congress initially (following enactment of the amendment) allowed two years to put in place rebuilding plans that end overfishing immediately.[40] Prior to the 2006 Amendment, Rebuilding Plans could allow overfishing during some of the rebuilding period so long as rebuilding was expected to be achieved within the time limit allowed (T_{MAX} as defined above).
- It requires accountability measures if the fishery exceeds its annual catch limit.[41]

Specifically, the MSFCMA as amended in 2006 states that Fishery Management Councils will:

> "…develop annual catch limits for each of its managed fisheries that may not exceed the fishing level recommendations of its scientific and statistical committee or the peer review process …"[42]

It calls on FMPs, FMP Amendments, or proposed regulations:

> "(A) to end overfishing *immediately* in the fishery and to rebuild affected stocks of fish; or (B) to prevent overfishing from occurring in the fishery whenever such fishery is identified as approaching an overfished condition." (emphasis added)[43]

It also requires FMPs to:

> "…establish a mechanism for specifying annual catch limits in the plan (including a multiyear plan), implementing regulations, or annual specifications, at a level such that overfishing does not occur in the fishery, including measures to ensure accountability."[44]

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[39] Magnuson-Stevens Fishery Conservation and Management Reauthorization Act, Pub. L. No. 109-479 §103(c)(3), 121 Stat. 3575, 3581 (2006).

[40] Magnuson-Stevens Fishery Conservation and Management Reauthorization Act, Pub. L. No. 109-479 §104(a)(10), 121 Stat. 3575, 3584 (2006).

[41] Magnuson-Stevens Fishery Conservation and Management Reauthorization Act, Pub. L. No. 109-479 §104(a)(10), 121 Stat. 3575, 3584 (2006).

[42] Magnuson-Stevens Fishery Conservation and Management Reauthorization Act, Pub. L. No. 109-479 §103(c)(3), 121 Stat. 3575, 3581 (2006).

[43] 16 U.S.C. §1854(e)(3)(A)-(B) (2012).

[44] Magnuson-Stevens Fishery Conservation and Management Reauthorization Act, Pub. L. No. 109-479 §104(a)(10), 121 Stat. 3575, 3584 (2006).

In light of the 2006 Amendment of the Act, the Federal Government issued new guidelines in 2009.[45] The revised Guidelines interpret the annual catch limit language in the Act as requiring a "hard" limit on catch (known as an "ACL") rather than implementing regulations or annual specifications establishing other forms of management (e.g., input controls such as closed areas, effort limits, gear restrictions). The Guideline gives three categories of circumstances under which there is an exception to the requirement for a hard limit on catch:

- *Life cycles*: This category of exceptions specifically applies to species with a one-year life cycle,[46]
- *International fishery agreements*: This category applies to fisheries that are subject to international agreements,[47]
- *Flexibility*: This category, "among other things" applies to management of endangered species, harvest from aquaculture operations, and species with unusual life history characteristics[48] (Pacific salmon are given as an example).

Prior to the addition of the "annual catch limit" text to the Act and the Agency's interpretation in the NS1G, some fisheries were managed by input controls such as closed areas and season, gear restrictions, and effort limits. One of the reasons for using these measures instead of a hard limit on catch is that FMCs felt that data limitations and the ability to enforce a catch limit were inadequate to support a limit on catch (e.g., most fisheries under the jurisdiction of the Caribbean Fishery Management Council). Reference to "among other things" under the Flexibility category could be interpreted as applying to data-poor situations and to situations where capacity to monitor and enforce a catch quota is poor. However, it has not been applied to such circumstances to date.

The NS1G also introduce the idea of an annual catch target ("ACT" as an option).

The ACT triggers an action to avoid exceeding the ACL. A key aspect of the Guidelines is that they direct the RFMCs to take account of scientific uncertainty and management uncertainty in Fishery Management Plans although the term "uncertainty" is absent from the MSFCMA. The revised NS1G also modifies the default biomass level associated with an overfished stock stating that it should be the greater of either ½ B_{MSY} or the minimum stock size at which rebuilding can occur within 10 years with no fishing mortality. The new NS1G also advises the RFMCs to take account of scientific uncertainty and management uncertainty as they develop their rebuilding plans, and provides means for doing so.

Figure 2.2[49] describes the relationship between catch levels described in the NS1G, and illustrates how the ACT safeguards against overfishing by taking account of uncertainties:

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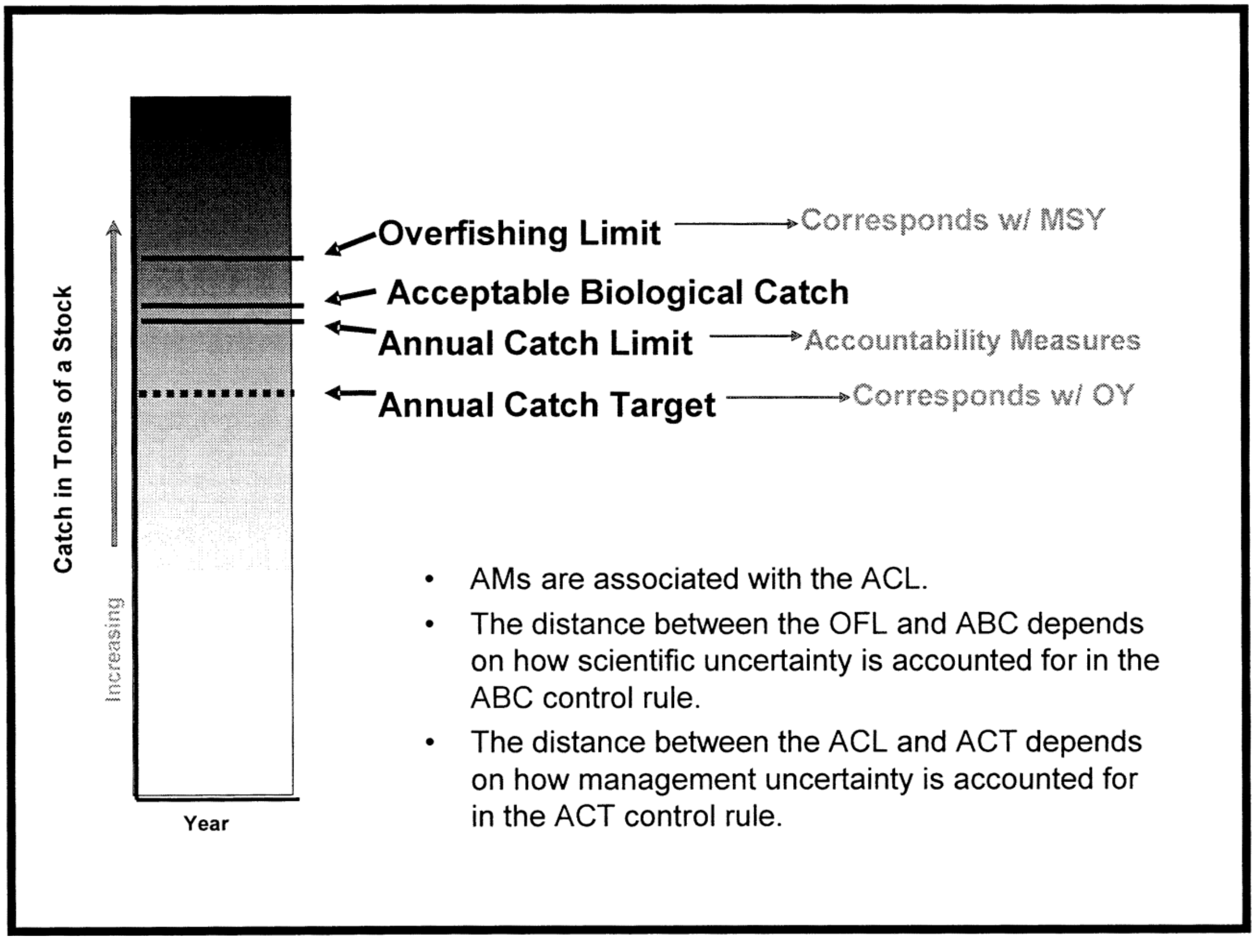
[45] 50 C.F.R. 600.310 (2009); 74 Fed. Reg. 3178 (Jan.16, 2009).

[46] 50 C.F.R. 600.310 (h)(2)(i) (2009).

[47] 50 C.F.R. 600.310 (h)(2)(ii) (2009).

[48] 50 C.F.R. 600.310 (h)(3) (2009).

[49] The diagram was used by the National Marine Fisheries Service to describe the Guidelines and it appeared in a draft guidelines. It was not included in the final guidelines, but it is an accurate characterization of the terms therein.



- AMs are associated with the ACL.
- The distance between the OFL and ABC depends on how scientific uncertainty is accounted for in the ABC control rule.
- The distance between the ACL and ACT depends on how management uncertainty is accounted for in the ACT control rule.

FIGURE 2.2: Relationships between various catch levels and limits
SOURCE: draft National Standard 1 Guidelines.[50]

The Overfishing Limit (OFL) corresponds to the catch applying the maximum fishing mortality threshold (MFMT), which is used to determine when overfishing is occurring (i.e., F>MFMT). F_{MSY} is the upper limit of the MFMT but it can be a function of stock size. The Acceptable Biological Catch (ABC) is a reduced catch to take account of scientific uncertainty in OFL. The Annual Catch Limit (ACL) reduces the catch an additional amount to take account of management uncertainty (e.g., the within year catch estimate is lower than the actual catch). The Annual Catch Target (ACT) is even lower as a safeguard against exceeding the ACL.

The Guidelines call for Accountability Measures (AM) if the ACL is exceeded. AMs are intended to avoid exceeding future ACLs and to mitigate adverse impacts on the stock that might have resulted from the excess catch. There are "in-season" accountability measures (such as a closure of the fishery when the estimated catch equals the ACT) and measures applied in the future years (such as time or area closures or a reduction in the ACL or ACT). According to the Guidelines:

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[50] 73 Fed. Reg. 32526, 32534 (June 9, 2008).

> "…If catch exceeds the ACL for a given stock or stock complex more than once in the last four years, the system of ACLs and AMs should be re-evaluated, and modified if necessary, to improve its performance and effectiveness. A Council could choose a higher performance standard (e.g., a stock's catch should not exceed its ACL more often than once every five or six years) for a stock that is particularly vulnerable to the effects of overfishing, if the vulnerability of the stock has not already been accounted for in the ABC control rule."[51]

The Guidelines do not explicitly indicate how scientific uncertainty and management uncertainty should be taken into account in the ABC and ACL respectively. Nor do the Guidelines change the previous interpretation of the time limit for rebuilding overfished stocks, which define T_{MIN} as the time it takes to rebuild to B_{MSY} with a probability of 0.50 with $F=0.0$. According to the Guidelines:

> " If T_{MIN} for the stock or stock complex exceeds 10 years, then the maximum time allowable for rebuilding a stock or stock complex to its B_{MSY} is T_{MIN} plus the length of time associated with one generation time for that stock or stock complex. 'Generation time' is the average length of time between when an individual is born and the birth of its offspring."[52]

There is a change in guidance on the way the biomass level corresponding to an overfished stock (or minimum stock size threshold, MSST) is specified. The Guidelines state:

> "… The MSST or reasonable proxy must be expressed in terms of spawning biomass or other measure of reproductive potential. To the extent possible, the MSST should equal whichever of the following is greater: One-half the MSY stock size, or the minimum stock size at which rebuilding to the MSY level would be expected to occur within 10 years, if the stock or stock complex were exploited at the MFMT … Should the estimated size of the stock or stock complex in a given year fall below this threshold, the stock or stock complex is considered overfished."[53]

The Guidelines do not specify the probability of the stock rebuilding to the MSY level when fishing at the MFMT (F_{MSY}). Therefore the guidelines are incomplete with respect to the specification of MSST. Since there is no guarantee that a stock will rebuild, the Guidelines go on to say:

> "If a stock or stock complex reached the end of its rebuilding plan period and has not yet been determined to be rebuilt, then the rebuilding F should not be increased until the stock or stock complex has been demonstrated to be rebuilt. If the rebuilding plan was based on a T_{TARGET} that was less than T_{MAX}, and the stock or stock complex is not rebuilt by T_{TARGET}, rebuilding measures should be revised, if necessary, such that the stock or stock complex will be rebuilt by T_{MAX}. If the stock or stock complex has not rebuilt by T_{MAX}, then the fishing mortality

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[51] 74 Fed. Reg. 3178, 3208-09 (Jan. 16, 2009)

[52] 74 Fed. Reg. 3178, 3212 (Jan. 16, 2009).

[53] 74 Fed. Reg. 3178, 3206 (Jan. 16, 2009).

> rate should be maintained at F_{REBUILD} or 75 percent of the MFMT, whichever is less."[54]

T_{TARGET} is the rebuilding time used in a rebuilding plan. T_{MAX} is the maximum rebuilding time allowed according to the previous quoted text from the Guidelines.

The Guidelines allow overfishing of a stock under certain limited circumstances. According to the Guidelines:

> "Harvesting one stock at its optimum level may result in overfishing of another stock when the two stocks tend to be caught together (This can occur when the two stocks are part of the same fishery or if one is bycatch in the other's fishery). Before a Council may decide to allow this type of overfishing, an analysis must be performed and the analysis must contain a justification in terms of overall benefits, including a comparison of benefits under alternative management measures, and an analysis of the risk of any stock or stock complex falling below its MSST. The Council may decide to allow this type of overfishing if the fishery is not overfished and the analysis demonstrates that all of the following conditions are satisfied:
> (1) Such action will result in long-term net benefits to the Nation;
> (2) Mitigating measures have been considered and it has been demonstrated that a similar level of long-term net benefits cannot be achieved by modifying fleet behavior, gear selection/configuration, or other technical characteristic in a manner such that no overfishing would occur; and
> (3) The resulting rate of fishing mortality will not cause any stock or stock complex to fall below its MSST more than 50 percent of the time in the long term, although it is recognized that persistent overfishing is expected to cause the affected stock to fall below its B_{MSY} more than 50 percent of the time in the long term."55

These limited circumstances under which overfishing is allowed to achieve long-term net benefits when there is bycatch is more restrictive than the exception in the 602 Guidelines discussed above. For example, the paragraph 3 above forbids a stock from having a 50% probability of falling below its MSST whereas the 602 Guidelines refer to Endangered Species Act listing (presumably a lower stock size than MSST).

The 2006 Amendments of the MSFCMA also provides for "widespread market-based fishery management through limited access privilege programs, and calls for increased international cooperation."[56] Market-based fishery management will be considered in Chapter 6 of this report. International aspects of the Act will be discussed later in this Chapter of the report.

On May 3, 2012, the National Marine Fisheries Service announced it would consider revising the NS1G and it solicited comments to be submitted by 12 October.[57] The comments

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[54] 74 Fed. Reg. 3178, 3212 (Jan. 16, 2009).
[55] 74 Fed. Reg. 3178, 3213 (Jan. 16, 2009).
[56] http://www.nmfs.noaa.gov/msa2007/index.html
[57] 77 Fed. Reg. 26238, 26238-26240 (May 3, 2012).

are summarized at:
http://www.nmfs.noaa.gov/sfa/domes_fish/NS1/ns1_anpr_comments_summary.pdf.

International Provisions of the MSFCMA

International considerations in the Act are prominent and far reaching (e.g., including provisions for a Tsunami warning system). The most relevant international aspects of the Act are:

- The requirements of NS1G to prevent overfishing and rebuild overfished stocks do not apply if the stock is subject to management by an international agreement adhered to by the U.S. (i.e., which it has ratified),[58]
- Requirements for reporting to Congress on the performance of international fisheries management such as efforts to eliminate illegal, unreported and unregulated fishing (IUU),[59] and
- That the U.S. will promote the provisions of the MSFCMA concerning overfishing and rebuilding overfished stocks internationally.[60]

The third point above is illustrated by the following text from the Act:

> "… If a relevant international fisheries organization does not have a process for developing a formal plan to rebuild a depleted stock, an overfished stock, or a stock that is approaching a condition of being overfished, the provisions of this Act in this regard shall be communicated to and promoted by the United States in the international or regional fisheries organization.''[61]

Science, Nature and the Law

Debates about fisheries management commonly end with the phrase "but it's the law!" The Committee contends that what is the law is not always as clear as it is portrayed. This contention is not based on legal considerations; rather it is a consequence of fisheries management being based on scientific concepts about sustainability and productivity of fishery resource populations.

The scientific concepts are characterizations of nature. However, science is imperfect in its characterizations. Consequently, the law sometimes oversimplifies scientific concepts, applies them inaccurately, or in an unclear way. In practice, what is represented as being the law is actually a combination of Executive Branch policies and legal judgments constrained by court rulings. It may or may not be the best interpretation of scientific concepts, and sometimes there are other reasonable scientific interpretations. Most importantly, interpretations of the law must be consistent with the realities of nature. The Act does not seem to recognize the dynamic nature

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[58] 50 C.F.R. 600.310 (2012).
[59] 16 U.S.C. § 1826 (2012).
[60] 16 U.S.C. § 1812 (2012).
[61] 16 U.S.C. § 1812 (2012).

of fish stocks and limits of science. While the NS1G help, they provide little practical guidance for many, if not the majority of stocks (e.g., numerous stocks for which data and knowledge about population and ecosystem dynamics are too limited to apply most aspects of the Guidelines).

Earlier in this report we described several cases in which it has been necessary to interpret the MSFCMA in order to operationalize it for fisheries management. These interpretations included:

- Using the term "overfished" to refer to a low biomass level. The Act frequently uses the terms "overfishing" and "overfished" interchangeably. Section 3, which defines terms, says "'overfishing' and 'overfished' mean a rate or level of fishing mortality that jeopardizes the capacity of a fishery to produce the maximum sustainable yield on a continuing basis."[62] The Act is also silent on the stock size that qualifies as being overfished and in need of rebuilding.
- The maximum allowable time for rebuilding.
- Applying the term "fishery" to individual stocks.

One of the most important interpretations required to apply the MSFCMA concerns Maximum Sustainable Yield (MSY). The discussion that follows highlights some of the scientific concepts and realities of nature that make interpretation and implementation of the law difficult. Some of these realities may also provide more scientific justification for flexibility than is commonly recognized.

Maximum Sustainable Yield (MSY)

MSY is a key concept used by the MSFCMA and fisheries management worldwide. The Act requires that fisheries be managed to achieve optimum yield prescribed as such as MSY reduced by ecological, economic and social factors.[63] However, MSY depends on many aspects of fisheries and ecosystems that are not addressed in the Act. At any point in time, the MSY of a stock of fish depends on:

- *Fishing practices*: Fishing mortality is an age- and size- specific rate vector. Changing the relative mortality by size or age changes MSY.
- *Environmental conditions*: Virtually all biological and ecological rates depend on environmental conditions. Some conditions are more favorable than other conditions in terms of the production of a population and MSY.
- *Biological interactions*: Fish stocks compete with each other and they prey on each other. Thus, the MSY of a species depends on the abundance of all the other species with which it interacts.

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[62] Sustainable Fisheries Act, Pub. L. No. 104-297, § 102(8), 110 Stat. 3559, 3562 (1996) (amending FCMA, Pub L. No. 94-265 § 3, 90 Stat. 331, 336 (1976)); 16 U.S.C. § 1802(34) (2012).
[63] Sustainable Fisheries Act, Pub. L. No. 104-297, § 102(7), 110 Stat. 3559, 3562 (1996).

- *Technical interactions*: Fishing for one species often results in mortality of other species as a result of bycatch. As a result it is usually impossible to apply the desired fishing mortality to achieve MSY simultaneously to several interacting stocks.
- *Scientific uncertainty*: There are several reasons that estimates of MSY and management reference levels associated with it (e.g., B_{MSY}) are uncertain even for well-studied stocks, and the situation is worse for many stocks that are considered data limited. Estimates may change more rapidly than actual stock conditions change.

As a result, MSY is a moving target. The factors that make MSY dynamic are discussed in greater detail in Chapters 4 and 5 of this report. The MSFCMA largely ignores the complexities associated with MSY and MSY reference points. The NS1G acknowledge several of the complexities, but the guidance for taking them into account is general. An exception is the guidance concerning technological interactions, which is so restrictive that it is rarely (if ever) applied even though technological interactions are common and they have important implications for fisheries management.

It should also be noted that the MSY concept is about biological yield (number, weight, or volume of fish). It does not take account of the value of the fish, cost of catching the fish, distribution of benefits from fishing, or social impacts of fishing or alternatively of prohibiting fishing. There are economic concepts that are analogs of MSY (e.g., Maximum Economic Yield). The human dimension of fisheries rebuilding is addressed in Chapter 6.

Status determinations

The MSFCMA requires an annual report to Congress on the status of fisheries. The MSFCMA also requires that management plans be developed to prevent overfishing and rebuilding plans are required to rebuild overfished stocks.[64] While the MSFCMA uses the terms overfished and overfishing interchangeably, the NS1G indicate that there is “overfishing” when fishing mortality exceeds F_{MSY} and a stock is “overfished” if stock size falls below the Minimum Standing Stock Threshold (MSST), defined in terms of stock size relative to the stock size associated with MSY.[65] At various times, MSST has been interpreted either as a stock size unlikely to occur randomly unless fishing mortality rate exceeds F_{MSY} or a stock size level from which the stock will recover to B_{MSY} in 10 years if $F = F_{\mathrm{MSY}}$. Neither concept has been precisely specified (see discussion above). In practice, MSST is generally set at ½ $B_{\mathrm{MSY.}}$

A scientific and management challenge of status determinations is that fisheries are virtually never prosecuted in a manner that achieves the absolute maximum long-term average yield, and MSY reference levels (in terms of yield, fishing mortality and biomass) are dynamic. The challenge is even more difficult for data limited stocks.

Status determinations are usually based on current or recent fishing practices (e.g., no change in selectivity or no change in size or age preference of the fishery). The dynamic nature of MSY is often taken into account by estimating the MSY reference levels over a period of time during which average conditions that affect MSY are believed to be the same as current

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[64] 16 U.S.C. §§ 1801-1881.
[65] 50 C.F.R. 600.310 (e)(iv)(2)(B)-(E) (2009).

conditions. In practice this usually means making a choice between using the entire time series of available data or data from a more recent period that is deemed more reflective of current conditions (see Chapter 3). In some cases, there is no conclusive scientific basis for making the choice even though the choice has a major effect on status determinations, rebuilding targets and rates, and social and economic impacts of fishery management.

Fishery versus Stock and the Mixed-Stock Exception

The MSFCMA refers to overfishing *fisheries* and overfished *fisheries*, although it also refers to rebuilding affected *stocks*. A common scientific interpretation of the term "fishery" is a group of fishing operations targeting similar (assemblage of) species, using similar gear, during the same period of the year and/or within the same area. The latter is known is also known as a "métier" (ICES, 2010a).

The MSY concept and MSY reference points could be applied to fisheries in the sense of a métier. However, the NS1G interpret a fishery as a single stock of fish. This interpretation creates the likelihood that long-term sustainable yield of some stocks will need to be sacrificed to prevent overfishing of other stocks in a mixed-stock fishery. It is also likely that this constraint will be necessary to rebuild some overfished stocks. The sacrifice in yield and benefits from fishery may be large in the case where the stock at risk of overfishing or in need of rebuilding is small (low potential yield or value) compared to the stocks for which yield and value is sacrificed (e.g. see canary rockfish).

The NS1G acknowledge the mixed-stock problem by providing an exception (referred to as the "mixed stock exception") under certain conditions. The NS1G text for the mixed-stock exception was provided earlier in this chapter. However, the conditions for the exception are demanding:

- The stock cannot be overfished (i.e., below the MSST). If it is overfished, presumably it must be rebuilt which usually means fishing mortality must be lower than the overfishing level.[66]
- The probability of the stock falling below the MSST cannot exceed 50%. For minor (usually data poor) stocks for which the mixed stock exception might be beneficial it may not be feasible to estimate such a probability.[67]

Another issue is the rationale for limiting the probability of falling below MSST to 50%. Ideally, there should not be concern about jeopardizing long term yield from the stock because the mixed stock exemption requires that net benefits from the fishery be higher when it is applicable. If it is about long-term risk of recruitment failure, or worse yet, extinction, then MSST is the wrong threshold. As discussed earlier, specification of MSST was based on the time it takes to rebuild to B_{MSY}, not recruitment failure or extinction.

If the mixed stock exception is applied, and the stock falls below MSST (expected 50% of the time), it may then be necessary to rebuild the stock (this is not clear from the guidelines). If so, then F will have to be reduced below the overfishing level until the stock rebuilds.

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[66] 74 Fed. Reg. 3178, 3213 (Jan. 16, 2009).

[67] 74 Fed. Reg. 3178, 3213 (Jan. 16, 2009).

Presumably F can then be increased again. Thus, applying the mixed-stock exception potentially creates a “yoyo” effect of increasing and decreasing F as stock size falls below MSST and is then rebuilt to B_{MSY}.

Rebuilding time

The law says the rebuilding time should be as short as possible and the rebuilding time is “shall… not exceed 10 years, except in the cases where the biology of the stock …” or some other considerations “…dictate otherwise.” (The complete text is given in the previous section of this report).[68] If a stock cannot rebuild with greater than 50% probability with F=0.0, the implicit interpretation of the NS1G is that biology of the stock dictates that the rebuilding time can exceed 10 years. There could have been other interpretations, such as the biology of the stock only dictates that the rebuilding time can exceed 10 years if there is zero probability with F=0.0. The Committee does not have a view on the implied interpretation in the NS1G except to point out that it is by necessity an interpretation because the law is not specific enough.

With regard to the law setting 10 years as the maximum rebuilding time unless other factors dictate otherwise, the Committee notes that many factors are relevant to rebuilding time, including:

- Mean generation time of the species to be rebuilt. The longer the mean generation time, the longer it will take to rebuild, all other factors being equal.
- Degree of depletion of the stock. The more depleted the stock, the longer it will take to rebuild.
- Environmental and ecological conditions. If conditions are favorable for the stock, it can rebuild faster than if they are unfavorable.
- Strength of year-classes (i.e., recruitment) entering the fishery. A stock will rebuild faster if year-classes entering the fishery at the time a rebuilding plan is initiated are relatively large, and vice versa.

All of these factors might be a reason the biology of the stock dictates that the rebuilding time could be less than or exceed 10 years. They are also reasons that on scientific grounds alone, one would not justify 10 years, or any other specific value, as a standard for rebuilding time although 10 years is probably a reasonable time for many stocks. If the biology of the stock or other factors dictate that the rebuilding time can exceed 10 years, the NS1G allow an increase in T_{MAX} to T_{MIN} years plus one mean generation time of the stock to be rebuilt.

One problem with this interpretation of the law is that it creates the potentially counter intuitive situation whereby a more pessimistic stock assessment results in a higher allowable fishing mortality rate. This occurs when the more pessimistic assessment means a stock can no longer be rebuilt in 10 years, leading to an increase in the allowable rebuilding time (often by a factor of 2 or more depending on mean generation), and in turn a higher fishing mortality rate.

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[68] Sustainable Fisheries Act, Pub. L. No. 104-297, § 109(e), 110 Stat. 3559, 3584 (1996).

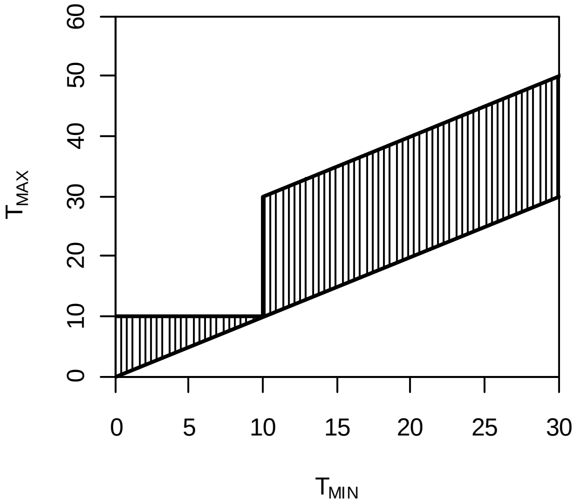

FIGURE 2.3: Relationship between T_{MIN}, and T_{MAX} for a stock with a generation time of 20 years. T_{TARGET} has to be selected from within the shaded region. The vertical line at T_{MIN} = 10 years indicates the discontinuity in the specification of the time horizon available for rebuilding resulting from the addition of one generation time, once it is determined that the stock cannot rebuild within 10 years (i.e. $T_{MIN} > 10$).

This is more than a hypothetical problem. The 9th Circuit Court ruled on this exact issue.[69] In this case, NMFS's 2002 fishing harvest level for darkblotched rockfish (Sebastes crameri), based on the NS1G on rebuilding time, was challenged. In 2000, NMFS determined that the darkblotched rockfish stock was "overfished" but could be rebuilt within 10 years. The following year revisions to their assessment indicated that the stock could not be rebuilt within 10 years. Subsequent calculations led to an increased allowable harvest level in 2002. The court ruled that dramatically increasing the fishing pressure and annual catch in the circumstance when a stock is in significantly worse shape than previously thought was incompatible with the Act.

NOAA proposed to eliminate this counter intuitive situation in 2005 in proposed revisions to the NS1G. The proposal was:

1) The "minimum time for rebuilding" means the amount of time it is expected to take to rebuild a stock to its MSY biomass level in the absence of any fishing mortality, starting in the first year after a stock is determined to be depleted. In this context, the term "expected" means to reach a 50-percent probability of attaining the B_{TARGET}. Also, technical updates to the minimum time (T_{MIN}) calculations must be retrospective to the same starting date.
2) If the minimum time for rebuilding a stock plus one mean generation time for the stock is 10 years or less, then the maximum time allowable for rebuilding that stock to its B_{TARGET} is 10 years.

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[69] Natural Res. Def. Council, Inc. v. Nat'l Marine Fisheries Serv., 421 F.3d 872 (9th Cir. 2005).

3) If the minimum time for rebuilding a stock plus one mean generation time for the stock exceeds 10 years, then the maximum time allowable for rebuilding a stock to its B_{TARGET} is the minimum time for rebuilding that stock, plus the length of time associated with one mean generation time for that stock.

This proposal by NOAA was not adopted, and in 2005, the 9th Circuit in that case ultimately held that:

> "Whatever the outer limits of the range of permissible constructions of the Act,…what lies beyond them is an interpretation allowing the Agency, upon discovering that a species is in significantly worse shape than previously thought, to increase dramatically the fishing pressure on that species. Increasing the annual take in these circumstances is simply incompatible with making the rebuilding period as short as possible."70

Accounting for uncertainty

The Guidelines do not explicitly indicate how scientific uncertainty and management uncertainty should be taken into account in the allowable biological catch (ABC) and annual catch limits (ACL) respectively. However, some RFMCs and Scientific and Statistical Committees (SSCs) have applied an approach known as the "P*" approach. P* is the allowable probability that the ABC will exceed the overfishing level (OFL) (Shertzer et al., 2010). For example, P*= 0.25 has been used for some fisheries, and a court ruling for summer flounder in the Mid-Atlantic region makes it clear that it should not exceed 0.50. The U.S. Court of Appeals for the District of Columbia described a catch with only an 18% chance of preventing overfishing as only existing "in Superman Comics' Bizarro world, where reality is turned upside down…."[71] The Settlement Agreement required at least 50% chance of preventing overfishing. Another approach is to apply a constant multiplier to OFL to calculate ABC (e.g., ABC = 0.75 OFL).

The Guidelines call for an additional reduction in catch from the ABC to the ACL to take account of management uncertainty. Furthermore, they call for (as an option) an ACT that is even lower such that the probability of exceeding the ACL should not exceed 25% (i.e., the Guidelines say the ACL should not be exceeded more frequently than one out of four years).

Another consideration is the accuracy of F_{MSY} proxies when F_{MSY} cannot be estimated with stock-specific data. The proxies are based on experience with fisheries management worldwide. Proxies are another source of uncertainty, and if they are selected conservatively, as some have argued (Rothschild and Jiao, 2011), they may also mean a reduction in yield. The total reduction to account for uncertainty is unspecified, but it could be substantial.

It should be noted that the 2009 version of the NS1G's introduction of guidance on taking account of uncertainty in setting ABCs and ACLs was not aimed at rebuilding plans. In the cases of rebuilding plans, the catch must be reduced below OFL in order to rebuild the stock

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[70] Natural Res. Def. Council v. Nat'l Marine Fisheries Serv., 421 F.3d 872, 881 (9th Cir. 2005).

[71] Natural Res. Def. Council, Inc. v. Daley, 209 F.3d 747, 754 (D.C. Cir. 2000).

within the rebuilding period with an acceptable probability. The court ruling cited above makes it clear that the probability must be 0.50 or greater, but there is no further guidance.

The role of Scientific and Statistical Committees

The MSFCMA charges Scientific and Statistical Committees (SSCs) (or some other peer review process) with recommending an Acceptable Biological Catch that may not be exceeded. Presumably, the intent is to separate conservation decisions and allocation decisions (i.e., who gets the fish), and to take politics and value judgments out of the former. In reality, this objective is difficult to achieve.

The primary reason that it is difficult to achieve is uncertainty. For example, SSCs are expected to recommend an ABC reduced from OFL to take account of scientific uncertainty. The P* method described above is one way to account for scientific uncertainty, but it should be up to managers to decide how much risk of exceeding OFL is acceptable. For well-studied fisheries where the probabilities can be estimated, it may be possible for RFMCs to give guidance on the risk. In fact, the NS1G call for RFMCs to develop ABC control rules that presumably would specify a risk level. However, SSCs have often been left to recommend an ABC without guidance on risk. Similarly, for rebuilding plans, managers need to decide on the probability of reaching the rebuilding target within the rebuilding period.

While preventing overfishing or ensuring a high probability of rebuilding a stock is ultimately a management responsibility, managers need to be informed by science. Obviously, a lower probability of overfishing means overfishing will be less frequent, but what is the right probability? Managers need to be informed by science about the potential yield that is foregone when the probability of overfishing is decreased, and about the conservation implications if overfishing occurs (keeping in mind that overfishing is not necessarily unsustainable). The acceptable probability of overfishing or not rebuilding within the maximum allowable time should be based on analysis, not intuition or emotion.

CHAPTER 2 FINDINGS

2.1: The MSFMCA bases the success or failure of fisheries management on the MSY concept. However, it does not take account of the complexity and dynamic nature of the MSY concept.

2.2: National Standard Guidelines operationalize the MSFMCA with respect to overfishing and other aspects of the Act. These guidelines are by necessity a blend of legal, policy and scientific interpretations of the Act. In some cases, there are alternative interpretations that would have been reasonable from a scientific point of view. For example, there is a discontinuity in rebuilding times at 10 years. There are alternatives that avoid this problem.

2.3: U.S. Fisheries management has evolved substantially since 1977 when the U.S. extended its jurisdiction to 200 miles. The evolution has been in the direction of being more prescriptive and precautionary in terms of preventing overfishing and rebuilding overfished fisheries. However, the trade-offs between precaution, ecosystem impacts and net benefits from fisheries have not been fully evaluated.

3

REVIEW OF FEDERALLY IMPLEMENTED REBUILDING PLANS

Introduction

Fishery Management Plans are developed with a main goal of preventing overfishing, and Rebuilding Plans are required to rebuild overfished stocks. The implementation nationwide of the rebuilding requirements of the Magnuson-Stevens Fishery Conservation and Management Act (MFCMA) posed difficult challenges for both the National Marine Fishery Service (NMFS), responsible for providing the technical support for the determination of stock status, and for the design of Rebuilding Plans, and for the Regional Fishery Management Councils (RFMCs) in charge of selecting and implementing Rebuilding Plans for every stock determined to be overfished. In this chapter we review the approach followed to implement the mandates of the MFCMA, as well as the outcomes of the adopted management actions from the perspective of single-stock rebuilding. After introducing some basic concepts and definitions, we discuss the components of formal Rebuilding Plans, the methods used to conduct rebuilding analyses, and the characteristics of some of the adopted plans.

Our review of outcomes of rebuilding plans proceeds from a broad summary of fish stock status nationwide, to a more detailed analysis of the evolution of the status of those stocks that were determined to be overfished since 1997 (using criteria specified by the 1996 Sustainable Fisheries Act), based on the annual reports presented to Congress on the status of federally-managed fisheries. A more in-depth analysis of estimated trends in fishing mortality and stock size was conducted for a subset of the stocks that are assessed by means of quantitative models, using the most recent stock assessment results made available by the NMFS. The empirical analysis of outcomes of rebuilding plans, as is often the case in policy analysis, has limitations because such policy interventions are uncontrolled experiments that lack the counterfactual: we do not know what the stock trajectories would have been without a rebuilding plan. Even if causality cannot be inferred, outcomes can be examined conditional on initial stock status, as evaluated in retrospect, and on the effectiveness of management to regulate fishing mortality. The analysis focused on three main questions approached in a step-wise manner: 1) how reliable are the classifications of stock status that triggered the implementation of Rebuilding Plans? 2) how successful were Rebuilding Plans at reducing fishing mortality? and 3) how are stock sizes responding? The underlying causes of failures to reduce fishing mortalities as planned are examined for a few selected cases. Finally, although strict comparability among regions is not possible given the different realities of fisheries and fishery management institutions, a brief review of rebuilding approaches and outcomes in a few other countries and regions is presented to place the U.S. situation within an international perspective.

Stock Status Determination

Under the National Standard 1 Guidelines (NS1G, see Chapter 2), a stock is overfished when its stock size (or stock biomass, *B*) is less than the Minimum Stock Size Threshold (MSST)[1], which for many stocks is defined to be ½ B_{MSY} (Figure 3.1(a) horizontally shaded area). A stock is subject to overfishing when the fishing mortality rate (*F*) exceeds the Maximum Fishing Mortality Threshold (MFMT) which cannot exceed F_{MSY}[2], indicated by the vertically shaded area in Figure 3.1(a). Consequently, stocks in the cross-hatched region of Figure 3.1(a) are both overfished and subject to overfishing.

Generic phase plane plots (often referred to as "Kobe" plots) are used to illustrate the stages of a fishery based on the relationship between fishing mortality rate (relative to F_{MSY}) and stock size (relative to B_{MSY}). Over time, stocks move in the Fishing mortality-Biomass phase plane. By plotting a fishery by stage from unfished, through stages of increased fishing, overfishing, overfished, reduced fishing, rebuilding, and ultimately to a position above B_{MSY}, phase plane plots illustrate the generic intent of the MSFCMA for overfished stocks (Figure 3.1b). A stock is rebuilt when stock size is at levels appropriate to achieve Maximum Sustainable Yield (MSY), and this outcome is maintained on average by keeping $F{=}F_{MSY}$ (although a lower fishing mortality should reduce the risk of overfishing taking place and of the stock becoming overfished again).

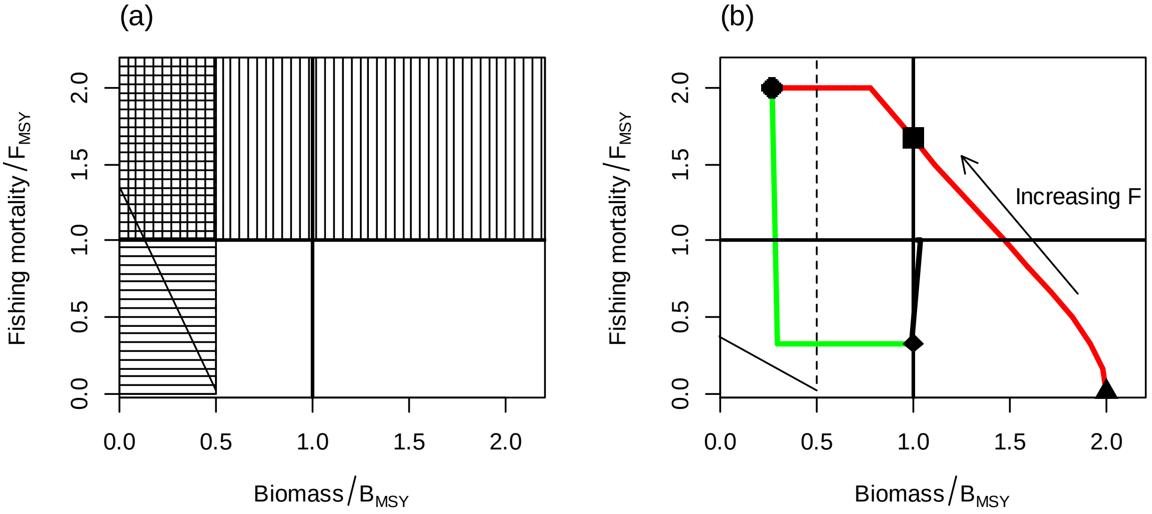

FIGURE 3.1: A phase plane plot, which is a graphical representation of changes in the status of the fishery with time with respect to realized fishing mortality and biomass. (a) The phase plane template showing regions of relative fishing mortality F/F_{MSY} and relative biomass B/B_{MSY}. Regions that are shaded horizontally represent overfished states ($B < ½\ B_{MSY}$), while regions that are shaded vertically indicate that overfishing is taking place ($F > F_{MSY}$). Crosshatched areas therefore represent the situation where a population is overfished and overfishing is taking place. (b) An example trajectory that a population might take from an unfished state as fishing gradually increases to passage into a period when overfishing is taking place, leading to biomass levels that are overfished. At this point, a

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[1] 50 C.F.R. 600.310 (e)(iv)(2)(B)-(E) (2009).
[2] 50 C.F.R. 600.310 (e)(iv)(2)(B)-(E) (2009).

Rebuilding Plan might go into effect, leading to reduced fishing mortalities that consequently allow biomass to increase and move back about B_{MSY}.

The triangle in the bottom right corner of the Figure 3.1b indicates a stock at its unfished level. Fishing mortality increases from the start of the fishery so biomass declines, as indicated by the arrow and the red line. The stock is subject to overfishing once fishing mortality exceeds F_{MSY} (1.0 on the y-axis of Figure 3.1b). The biomass drops below B_{MSY} (solid square) and then below the MSST, given fishing mortality is substantially in excess of F_{MSY}. The stock would be considered to be in an overfished state once it drops below MSST. A Rebuilding Plan is then implemented (solid circle) and fishing mortality rates are reduced to F_{MSY} or below. The stock increases under lower fishing mortality (green lines in Figure 3.1b), eventually recovering to B_{MSY} (Figure 3.1b diamond), when fishing mortality is increased back to close to F_{MSY}. Figure 3.1b is idealized for several reasons. For example, it includes no assessment error, no variability in recruitment, and it assumes that management decisions are implemented exactly.

In reality, the ability to provide scientific management advice, including stock status determinations and stock projections used to develop Rebuilding Plans, is subject to several sources of uncertainty. These sources can be categorized as:

- *Data uncertainty*: Data uncertainty results from two main sources: 1) Bias – how the data represent the processes being monitored (e.g., changes in commercial catch per unit effort [CPUE] relative to actual changes in biomass), and 2) Variation – how variable are the sample observations of the system (which can be influenced by system variability, but also by the methods and frequency used to observe it).
- *Model uncertainty*: All models characterize nature in a simplified manner. In many cases there may exist several plausible, but different, assumptions that are supported by the data to a similar extent, but which may have very different implications for stock status and management. Choices of assumptions contribute to model uncertainty.
- *Implementation uncertainty*: Management actions are devised to control fishing mortality, but only do so indirectly through catch and effort controls, and technical measures (discussed below). The effectiveness of regulations is uncertain given enforcement challenges, and the fact that fisher responses to regulations are difficult to anticipate.
- *Unpredictability of nature*: Some aspects of nature are quite variable (e.g., the size of future year classes) and therefore very difficult to predict. Future states of nature, including population size, are modeled probabilistically, so that a reasonable range of future outcomes can be described in the context of evaluating Rebuilding Plans. Assumptions about the likelihood of future states of nature lead to uncertainty.

Stock assessment scientists use the best-available scientific methods to estimate stock size and fishing mortality rate. They can also make reasonable short-term projections of future stock size, especially for the well-studied fish populations. Future production is more difficult to estimate because the relationship between recruitment and biomass of spawning fish is highly variable and uncertain. F_{MSY} is often set to an assumed value derived from life history parameters (i.e., a “proxy”) based on general experience with similar stocks elsewhere when it cannot be estimated with an acceptable degree of confidence. Setting a proxy for B_{MSY} is more challenging than setting a proxy for F_{MSY}, because B_{MSY} depends on F_{MSY} and because B_{MSY} relies on an estimate of (or assumption about) the absolute magnitude of the average recruitment at which MSY is achieved.

Unfortunately, many U.S. stocks are not well enough studied to allow application of MSY-based control rules. For so-called “data-limited stocks,” it may only be possible to describe trends in terms of relative abundance, but not in absolute terms. In more extreme cases, only basic biological information (e.g., growth rate) is available, and catch data are unavailable or of questionable quality (e.g., most stocks in the Caribbean). A range of estimation methods and approaches exist to deal with these diverse situations (see Chapter 4 for more information related to estimating F_{MSY} and B_{MSY} in data-rich and data-poor situations).

Implementation of Rebuilding Requirements

Development and review of Rebuilding Plans

As a result of the 2006 Amendment to the MSFCMA, Fishery Management Plans including Rebuilding Plans must now be designed to end overfishing immediately. RFMCs have two years to develop a Rebuilding Plan for stocks that are declared overfished by NMFS based on the stock assessments conducted and reviewed through the respective Council processes[3]. However, RFMCs and NMFS can proactively propose and implement measures aimed at reducing fishing mortality even before a Rebuilding Plan is formally adopted. The nature of the technical analyses used to develop Rebuilding Plans, and the specific elements of a Rebuilding Plan depend on the information available. The formal Rebuilding Plan is adopted by the RFMC and sent to the Secretary of Commerce for approval. There are opportunities for public input, as well as for input from RFMC advisory bodies, during the development of a Rebuilding Plan.

A Rebuilding Plan normally includes the following components:

- A target time period (T_{TARGET}) for rebuilding the stock to B_{MSY} (or its proxy). This target is bounded below by T_{MIN}, the minimum time to rebuild to B_{MSY} in the absence of all future fishing, and T_{MAX}, the maximum rebuilding time (see Figure 2.5 in Chapter 2). T_{MAX} is 10 years if the stock can rebuild in 10 years or less, with 50% probability, under zero fishing mortality.
- The values for parameters such as MSST, B_{MSY}, T_{MIN} and T_{TARGET} when the Rebuilding Plan was first developed.
- The harvest strategy to be applied during the rebuilding period. Most harvest strategies are based on a constant target fishing mortality or on control rules that decrease the target fishing mortality when biomass drops below predefined thresholds. In other cases, the harvest strategy is based on a constant catch or a time-series of pre-specified catch levels. The harvest strategy often also includes restrictions on where and when fishing can take place and the gear types that can be used.
- A general discussion of the types of management measures that will be used to implement the Rebuilding Plan.

Rebuilding plans require scientists and managers to make choices about targets, limits, and the probability of rebuilding. These implicit and explicit choices reflect judgements about expected benefits and costs, and the level of risk that can be tolerated.

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[3] Magnuson-Stevens Fishery Conservation and Management Reauthorization Act, Pub. L. No. 109-479 §103(c), 121 Stat. 3575, 3581 (2006).

The values for MSST, B_{MSY}, T_{MIN} and T_{TARGET} can change given new information on stock status and/or productivity, or changes in stock assessment methodology and the assumptions on which assessments are based.

For stocks with quantitative stock assessments, the development of a Rebuilding Plan involves two key technical aspects: (a) a rebuilding analysis that quantifies trade-offs between exploitation rate, rate of recovery and time-to-recovery, and (b) an evaluation of the socioeconomic consequences of different harvest strategies and choices for T_{TARGET} (see Chapter 6 for further discussion of the evaluation of socioeconomic factors) The process followed by the Pacific Fishery Management Council (PFMC) to develop Rebuilding Plans for overfished groundfish stocks is provided in Box 3.1 as an example. Rebuilding analyses typically involve conducting stochastic simulations under various harvest strategies to calculate the probability of recovery. A number of choices regarding how population projections are undertaken need to be made. These choices are described and evaluated in Chapter 4.

BOX 3.1
Example: Groundfish Management by the Pacific Fishery Management Council (PFMC)

For groundfish stocks, the MSST is 25% of unfished reproductive output (often quantified as 25% of unfished spawning biomass) for rockfishes and groundfishes while it is 12.5% of unfished reproductive output for flatfishes. MSST differs between these classes of stock given the relative productivity of flatfishes compared to groundfishes (PFMC, 2011b). Productivity is low for most groundfish stocks managed by the PFMC so T_{MAX} is generally T_{MIN} plus one mean generation time (exceptions to this are Pacific hake, *Merluccius productus*, and petrale sole, *Eopsetta jordani*). For a stock that is newly declared to be overfished, T_{TARGET} is the year in which rebuilding is predicted to occur with 50% probability under the selected harvest strategy. The harvest strategy is a constant spawning biomass-per-recruit strategy (equivalent to a constant fishing mortality strategy if the mix of fishing gears does not change over time), although the rebuilding plan for one overfished stock (yelloweye rockfish) involved a ramp down in catches (PFMC, 2006). Most PFMC rebuilding analyses allow for stochasticity in future recruitment while some also allow for uncertainty in B_{MSY} as well as the current population age-structure and reproductive output. The PFMC has Terms of Reference for rebuilding analyses for groundfish species (e.g., PFMC, 2012). The key aspects of these Terms of Reference include:

- B_{MSY} should be defined using the proxies established by the PFMC Scientific and Statistical Committee (SSC) although direct estimates of B_{MSY} can be used if they are judged to be robust (none are at present).
- T_{MIN} is the year in which rebuilding to B_{MSY} occurs with 50% probability if all fishing ceased the year after the stock was declared overfished.
- Mean generation time is defined as the mean age of the net maternity function (i.e., the product of the survivorship and fecundity-at-age). When growth and/or natural mortality are changing over time, survivorship and fecundity-at-age are based on recent estimates to reflect current conditions.
- Projections should be conducted for a full range of possible harvest strategies, including setting fishing mortality equal to zero (a strategy with a T_{TARGET} equal to T_{MIN}) and fishing mortality so recovery occurs with 50% probability by T_{MAX}, as well as a range of harvest strategies with times to recovery with 50% probability between T_{MIN} and T_{MAX}.
- The analysed management strategies and choices for T_{TARGET} for the overfished species are grouped into ‘alternatives’ that involve different combinations of ACLs for each overfished species.

The results of the rebuilding analyses and any socio-economic analyses (discussed in Chapter 6) are reviewed by the Council’s SSC and by other Council advisory bodies (see, for example, the Groundfish Advisory Panel review of impacts in PFMC (2006)). The management strategy adopted is used in combination with the results of stock assessments to set ACLs.

Rebuilding Plans and annual setting of management regulations are based on a “best” assessment or a model-averaged set of assessment scenarios (see Chapter 4).

For stocks for which there is no new stock assessment (update or full), the biannual review of Rebuilding Plans consists of checking that catches are below ACLs. In contrast, when there is a new stock assessment, the SSC review process evaluates:

(1) the catches of the overfished species relative to the annual ACLs summed over the period of rebuilding;
(2) whether the rebuilding analyses met the appropriate technical requirements;
(3) the year in which rebuilding is predicted to take place under the current harvest strategy relative to the T_{TARGET} that was specified in the Management Plan Amendment 16-4 (PFMC, 2006) and the current T_{TARGET} (which may differ from the value specified in Amendment 16-4 if the Council changed T_{TARGET} since the Rebuilding Plan was established).

On this basis, the SSC determines which stocks are not rebuilding at the expected rate, and which are very unlikely to rebuild by the specified T_{TARGET} under the current harvest rate, and whether that rate will allow recovery by the updated value of T_{MAX}. The latter situation can arise if a major change to the stock assessment has occurred. The SSC also recommends whether current harvest rates are a reasonable starting point for developing ACLs for the next biennial management cycle.

The choice of rebuilding time depends on the inherent productivity of the stock, as determined by its natural mortality, growth rate, and age-specific fecundity. Productivity can be quantified by the mean generation time (the average age of the mothers of offspring in an unfished population), as well as the extent of compensation in the stock-recruitment relationship (the extent to which per capita recruitment increases on average as biomass declines, often quantified using the “steepness” parameter). Figure 3.2 shows the time to rebuild to B_{MSY} predicted using a standard age-structured population dynamics model, as a function of biomass relative to B_{MSY} and fishing mortality relative to F_{MSY} for two choices of generation time and steepness (0.5 and 0.9). Figure 3.2 is highly idealized because it ignores, for example, stochastic dynamics, and multispecies interactions. However, it nevertheless illustrates how populations are projected forward when conducting rebuilding analyses. T_{MIN} is the time-to-rebuild when $F/F_{MSY}=0$ (the y-origin). T_{MIN} is less than 10 years for the stock with low steepness and a short generation time (Figure 3.2 top, left) only if the stock is initially above $0.4B_{MSY}$. In contrast, T_{MIN} is less than 10 years for most initial stock sizes if steepness is high (bottom panels).

Rebuilding to B_{MSY} can occur relatively rapidly (less than twice the generation time) with fishing rates of $0.7F_{MSY}$ and less, when B/B_{MSY} is close to 0.5. Larger reductions in F are required to achieve rebuilding in any given time period when the stock is more depleted; the dependence on the depletion level is nonlinear so that very low Fs are required when the stock is highly depleted (say less than 0.1 B_{MSY}). Thus, the adoption of a harvest control rule that reduces F when stocks are depleted below B_{MSY} will allow rebuilding even in the absence of a formal Rebuilding Plan, especially for stocks with high steepness (see Myers *et al*. [2002] for a meta-analysis of steepness values, which suggests that steepness tends to be high rather than low for most exploited stocks). On the other hand, keeping fishing mortalities closer to the long-term target (say a little below F_{MSY}) would still achieve rebuilding but the rebuilding period may be considerably longer, depending on the initial depletion and population parameters. Figure 3.2 also illustrates some of the effects of the 10-year discontinuity in the rule used to define T_{MAX} discussed in Chapter 2 (Figure 2.5). If the stock in the lower right panel of Figure 3.2 (Generation Time = 18.3 years and natural mortality M=0.1yr^{-1}) were depleted to ~$0.12B_{MSY}$, the fishery on it should effectively be closed given that it can (just) rebuild in 10 years in the absence of fishing. However, if it were even slightly more depleted (say $0.1B_{MSY}$) then T_{MAX} would increase to more than 28 years (i.e., 10 years plus one mean generation time) and hence the rebuilding fishing mortality could be as large as $0.8F_{MSY}$.

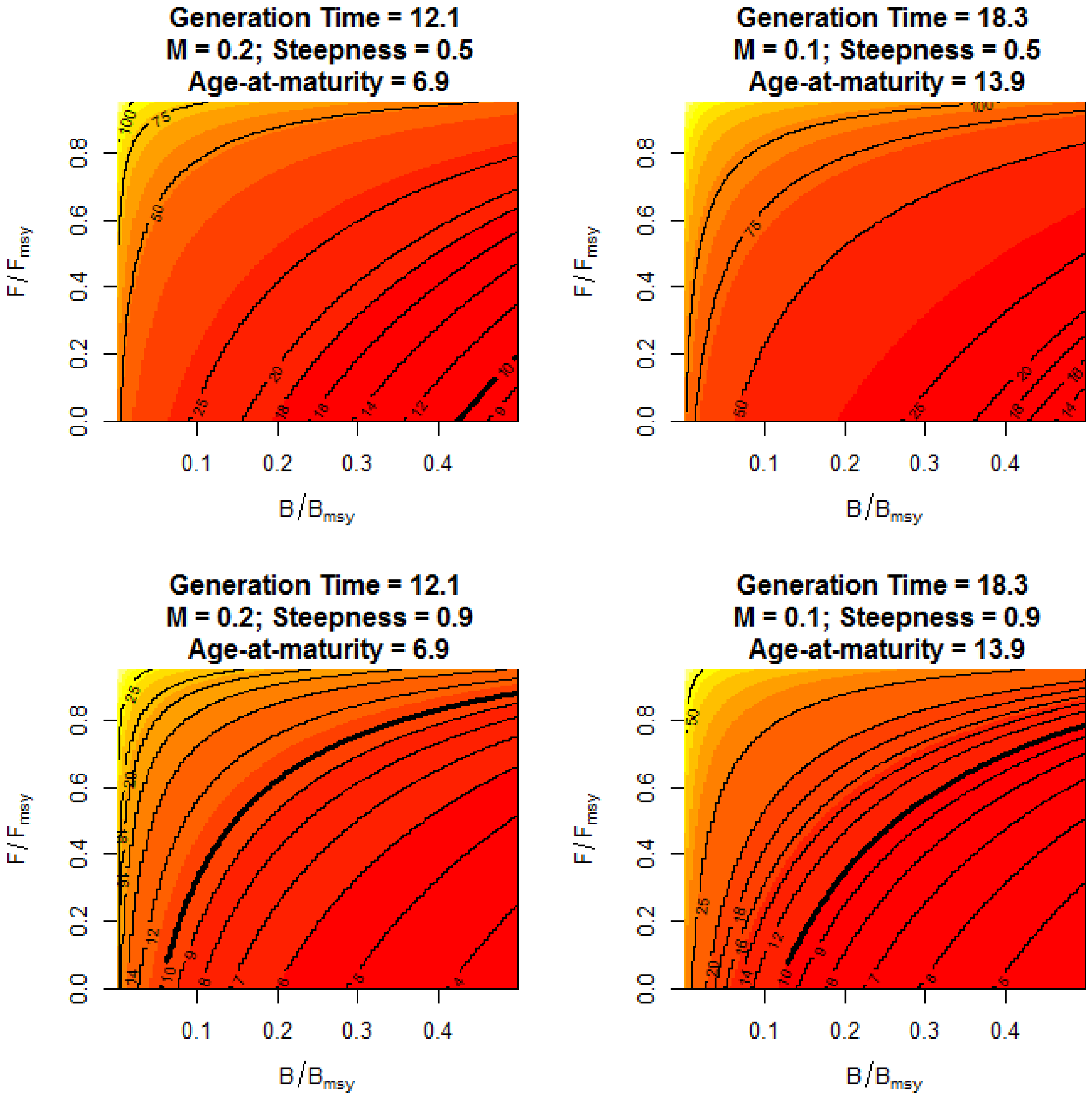

FIGURE 3.2: Time to rebuild to B_{MSY} as a function of biomass relative to B_{MSY} and fishing mortality relative to F_{MSY}. Results are shown for two choices for each of the steepness of the stock-recruitment relationship and the extent of productivity (quantified by natural mortality and age at maturity, resulting in mean generation times of 12.1 and 18.3 years).

Quantitative rebuilding analyses are not conducted for all overfished stocks. Reasons for this include the lack of a quantitative stock assessment on which to base forecasts, cases in which the “stock” is actually a complex of stocks, and lack of personnel to conduct the technical analyses. The 2006 reauthorization of the MSFCMA includes the requirement that that all stocks have Annual Catch Limits, which are difficult to estimate without accurate stock assessments. The reauthorization has increased the demand for stock assessment information, particularly in data-poor situations.

The amount, extent, and timing of inclusion of socio-economic information in the analyses that support Rebuilding Plans differ among regions and for species within regions (see Chapter 6).

Management controls

Several management controls can be implemented to ensure that fishing does not exceed target levels (Hilborn and Walters, 1992). Most RMFCs adopt a mix of management measures using a combination of input (gear restrictions, spatial and temporal restrictions in effort) and output (daily or seasonal trip or bag limits) controls. Prior to the 2006 reauthorization of the MSFCMA, some RFMCs (e.g., New England) tried to reach target catch levels using only input controls. However, RFMCs are now required to define ACLs as output controls for all stocks to comply with the NS1G.

Review of stocks and stock status

As required by the SFA, the status of U.S. stocks has been reported to Congress on an annual basis since 1997. In terms of fishing mortality, stocks are classified as being subject to overfishing or not being subject to overfishing. The classification of biomass status is more complicated. If stock size is below the MSST, the stock is classified as overfished, thus a Rebuilding Plan is required. Stocks subject to a Rebuilding Plan are classified as "rebuilding" if stock size is between the MSST and B_{MSY}. Stocks in this range that are not subject to a Rebuilding Plan and all stocks larger than B_{MSY} are classified as "not overfished". There are also many stocks with "unknown" status. Reports to Congress during the early implementation of the SFA are less reliable and often used pre-SFA criteria, which did not distinguish between overfishing and overfished. Most of the stocks whose status is reported to Congress are assessed as single stocks, but some are assessed as stock complexes, which contain a group of species with similar geographic distribution and life history, and that co-occur in a fishery. All are referred to here as "stocks."

Out of the total number of stocks identified in the reports to Congress, 230 of them (contributing over 90 percent of the total fishery landings) were selected for their importance to commercial and recreational fisheries, and are used to construct a "Fish Stock Sustainability Index" (FSSI4) as an indicator of management and stock performance. Stocks in the FSSI group are scored according to their status in terms of stock size (one point awarded if B > MSST and another point if B > 0.80 BMSY), whether overfishing is taking place (one point if overfishing is not occurring), and whether "overfished" and "overfishing" status are known (half a point each). The FSSI hence provides a summary of trends in whether major stocks (those which constitute the bulk of the total catch in U.S. waters) are overfished or subject to overfishing.

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[4] These reports are available at http://www.nmfs.noaa.gov/sfa/statusoffisheries/SOSmain.htm

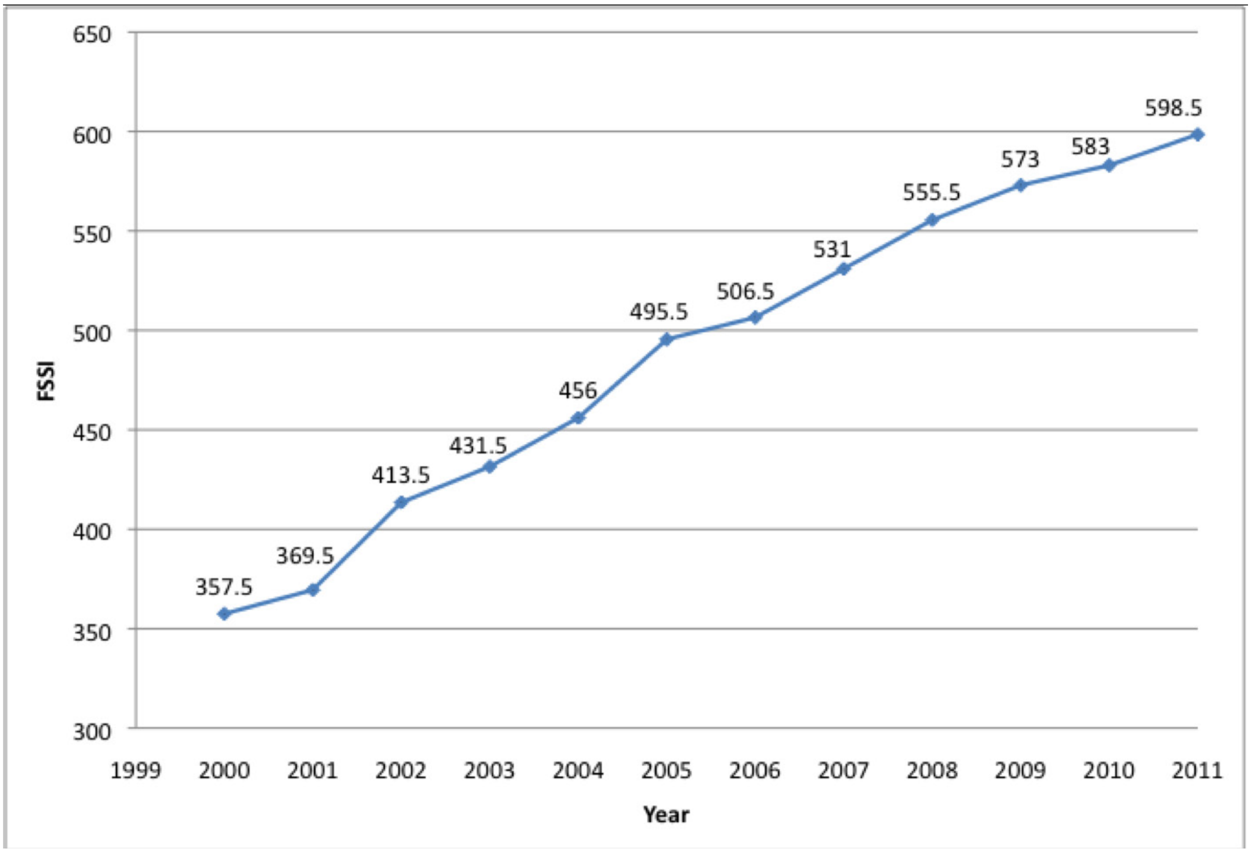

FIGURE 3.3: Fishery Status Sustainability Index for U.S. Fisheries since 2000.
SOURCE: NOAA.

In addition to the species included in the FSSI, there are 248 stocks or stock complexes included in the report to Congress. The status of many FSSI and non-FSSI stocks relative to overfishing and being in an overfished state is unknown because not all of these stocks are assessed using quantitative methods. Some stocks cannot be classified, e.g., in situations where no assessment (qualitative or quantitative) has been conducted for the stock or it has not been possible to estimate B_{MSY} and F_{MSY} with acceptable reliability. In some cases, although no analytical assessment is available for a given stock, the stock status is evaluated based on trends in survey data or commercial catch per unit of effort. This is, for example, the case of the Pribilof Islands blue king crab, and also the stocks of silver hake and the species of skates from NEFMC.

Of the 230 FSSI stocks, only 168 were classified in terms of both overfishing and overfished status in quarter 3 of 2012: 23% of these were considered overfished (B < MSST) and fishing mortality was estimated to be above F_{MSY} in more than half of the overfished stocks (Table 3.1). The majority of the FSSI stocks for which status related to overfishing and being in overfished state was determined (77%) were not overfished and not subject to overfishing; overfishing was still taking place in 18% of the stocks. The FSSI has shown a steady improvement since 2000 (Figure 3.3). Status could be determined for only a small fraction of the non-FSSI stocks, and the majority of them were neither overfished nor subject to overfishing. There are currently 50 stocks under Rebuilding Plans[5]. In 1997, when the first report to Congress was prepared, 30% of assessed stocks (86 out of 279) were overfished,

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[5] http://www.nmfs.noaa.gov/sfa/statusoffisheries/2012/third/Q3_2012_FSSI_nonFSSIstockstatus.pdf

although for most species, the existing overfishing definitions were based wholly or in part on a fishing mortality rate, not biomass levels. [6]

Current estimates of B/B_{MSY} and F/F_{MSY} for 137 stocks (117 FSSI and 20 non-FSSI) assessed using quantitative methods (made available to the Committee by NMFS) are shown in Figure 3.4. Points that are above the horizontal line correspond to stocks that are subject to overfishing ($F>F_{MSY}$); stocks on the left of the vertical line ($B<B_{MSY}$) may be classified as "overfished" when $B<$MSST, or as "rebuilding" (MSST$\leq B<B_{MSY}$) when they were once declared overfished, but are now estimated to be above MSST but not yet rebuilt ($B>B_{MSY}$). A stock is defined to be "rebuilt" if it has recovered to the estimated B_{MSY}. Of the stocks included in this figure, which correspond to stocks for which information is available to allow estimation of B/B_{MSY} and F/F_{MSY}, 21% (29 of 137 stocks) are currently below MSST, 6% (8 of 137 stocks) were overfished but are now above MSST and are under Rebuilding Plans, and 17% (23 of 133 stocks) are being fished at an intensity in excess of F_{MSY}.

TABLE 3.1: Summary of stock status for FSSI and non-FSSI stocks reported to Congress as of 30 September 2012 by overfishing and overfished category.

		FSSI stocks			Non-FSSI stocks		
		Overfishing?			Overfishing?		
		Yes	No	Unknown	Yes	No	Unknown or NA
Overfished?	Yes	21	17	1	1	3	
	No	3	111	8		30	7
	Rebuilding	2	9				
	Approaching Overfished	4	1		1		
	Unknown	2	24	27		31	175
	TOTAL	32	162	36	2	64	182

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[6] http://www.nmfs.noaa.gov/sfa/statusoffisheries/Archives/StatusofFisheriesReportCongress1997.htm#Summary

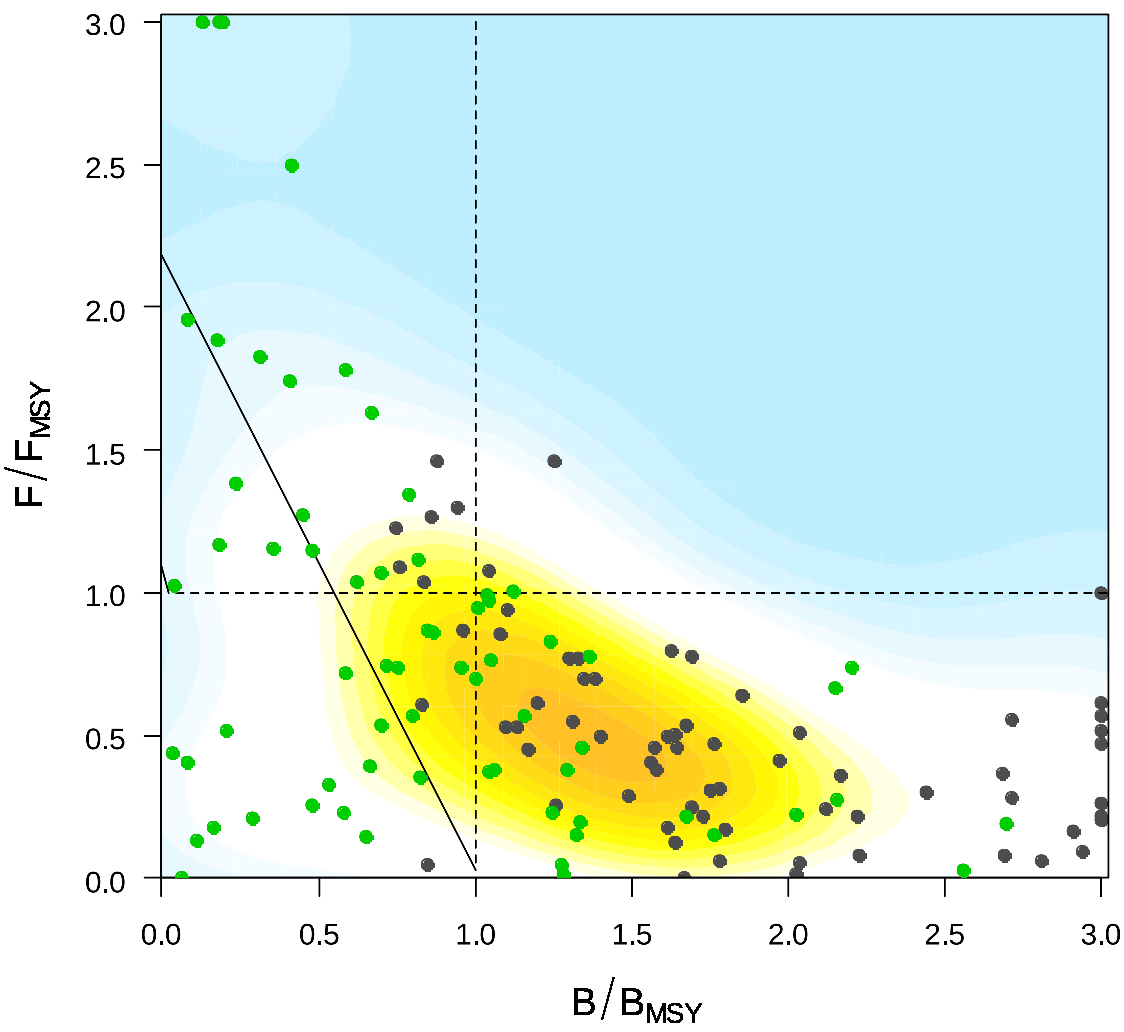

FIGURE 3.4: Realized F/F_{MSY} versus B/B_{MSY} for 137 assessed U.S. fish stocks (117 FSSI stocks and 20 non-FSSI stocks). Stocks for which $F/F_{MSY} > 3$ are reported as $F/F_{MSY} = 3$ and stocks for which $B/B_{MSY} > 3$ are reported as $B/B_{MSY} = 3$. Stocks which have at some time been declared overfished and are included in this study are indicated as green dots while other stocks are indicated as black dots.

Over the period 1997-2011, after the SFA was signed into law, 85 federally managed stocks (79 FSSI and 6 non-FSSI) were at some point declared to be overfished or approaching an overfished state, and were therefore subject to the requirement to be placed under a Rebuilding Plan (Table 3.2). This list of stocks was compiled by staff of the NMFS at the request of the committee; it does not include stocks that were declared overfished using pre-SFA criteria not based on biomass reference points (e.g., some South Atlantic grouper species), or stocks that are no longer listed as individual stocks (e.g., many shark species that were added to the Large Coastal Shark Complex and cannot be retained in commercial or recreational fisheries).

TABLE 3.2: The 85 stocks that have been declared overfished and their current status relative to F_{MSY}, B_{MSY}, and MSST (as of 2012 status report to Congress, quarter 3). The column “included in the evaluation set” indicates whether the stock is one of the 55 considered for detailed analysis. “Analytical assessment” indicates whether there is an accepted analytical assessment for the stock; the current overfished and overfishing status are status relative to B_{MSY} and F_{MSY} from the most recent assessment (see the last column for the most recent year for which estimates of biomass and fishing mortality are available). The column “First year of the plan” indicates the year when a Rebuilding Plan was adopted, while “Timeframe for rebuilding” is the current length of the rebuilding plan (which may differ from the length of rebuilding period when the plan was first developed.) It should be noted that some stocks (e.g., Bering Sea Tanner crab) have been declared overfished for a second time, having rebuilt under an earlier Rebuilding Plan.

Stock	**Jurisdiction**	**Included in evaluation set?**	**Analytical assessment?**	**Current overfished status**	**Overfishing Now**	**Year declared overfished**	**First year of the plan**	**Year declared rebuilt**	**Time frame for rebuilding**	**Last Year with assessment results**
Queen conch – Caribbean	CFMC	No	No	Overfished	Yes	2000	2005		15	
Caribbean Grouper Unit 1	CFMC	No	No	Overfished	Yes	2000	2005		25	
Caribbean Grouper Unit 2	CFMC	No	No	Overfished	No	2000	2005		30	
Caribbean Grouper Unit 4	CFMC	No	No	Overfished	Yes	2005	2005		10	
Gray triggerfish - Gulf of Mexico	GMFMC	No	Yes	Overfished	Yes	2008	2008		6	2010
Greater amberjack - Gulf of Mexico	GMFMC	Yes	Yes	Overfished	Yes	2001	2003		10	2009
Red snapper - Gulf of Mexico	GMFMC	Yes	Yes	Overfished	No	2000	2001		32	2008
Albacore - North Atlantic	HMS	Yes	Yes	Overfished	Yes	1999	2007		Not specified	2007
Bigeye tuna – Atlantic	HMS	Yes	Yes	Rebuilding	No	2000	2008		Not specified	2009
Blacknose shark – Atlantic	HMS	No	Yes	Overfished	Yes	2008	2010		18	2009
Blue marlin – Atlantic	HMS	Yes	Yes	Overfished	Yes	1997	2001		Not specified	2009
Bluefin tuna - Western Atlantic	HMS	Yes	Yes	Overfished	Yes	1997	1999		19	2009
Dusky shark – Atlantic	HMS	Yes	Yes	Overfished	Yes	1999	2003		100	2009
Porbeagle – Atlantic	HMS	No	Yes	Overfished	No	2006	2008		100	2005
Sailfish - Western Atlantic	HMS	No	Yes	Rebuilding	Yes	1998	2008		Not specified	2007
Sandbar shark – Atlantic	HMS	Yes	Yes	Overfished	No	1999	2003		66	2009
White marlin – Atlantic	HMS	No	No	Overfished	Yes	1997	2001		Not specified	2004
Butterfish - Gulf of Maine / Cape Hatteras	MAFMC	No	Yes	Undefined	No	2004	2010		4	2008
Tilefish - Mid-Atlantic Coast	MAFMC	Yes	Yes	Rebuilding	No	1997	2001		10	2008
Acadian redfish - Gulf of Maine / Georges Bank	NEFMC	Yes	Yes	Rebuilt	No	2001	2004	2012	47	2010
American plaice - Gulf of Maine / Georges Bank	NEFMC	Yes	Yes	Rebuilding	No	2003	2004		10	2010

Atlantic cod - Georges Bank	NEFMC	Yes	Yes	Overfished	Yes	2002	2004		22	2010
Atlantic cod - Gulf of Maine	NEFMC	Yes	Yes	Overfished	Yes	2002	2004		10	2010
Atlantic halibut - Northwestern Atlantic Coast	NEFMC	Yes	Yes	Overfished	Yes	1997	2004		52	2010
Atlantic wolffish - Gulf of Maine / Georges Bank	NEFMC	No	Yes	Overfished	Yes	2010	2010		Not specified	2010
Ocean pout - Northwestern Atlantic Coast	NEFMC	Yes	No	Overfished	Yes	1999	2004		10	2010
White hake - Gulf of Maine / Georges Bank	NEFMC	Yes	Yes	Overfished	Yes	1999	2004		10	2007
Windowpane - Gulf of Maine / Georges Bank	NEFMC	No	Yes	Overfished	Yes	2010	2010		7	2010
Windowpane - Southern New England / Mid-Atlantic	NEFMC	Yes	Yes	Rebuilt	No	2002	2004	2012	10	2010
Winter flounder - Georges Bank	NEFMC	Yes	Yes	Rebuilding	No	1999 & 2010	2010	2003	7	2010
Winter flounder - Southern New England / Mid-Atlantic	NEFMC	Yes	Yes	Overfished	No	2003	2004		10	2010
Witch flounder - Northwestern Atlantic Coast	NEFMC	N0	Yes	Overfished	Yes	2010	2010		7	2010
Yellowtail flounder - Cape Cod / Gulf of Maine	NEFMC	Yes	Yes	Overfished	Yes	2002	2004		19	2010
Yellowtail flounder - Georges Bank	NEFMC	Yes	Yes	Overfished	No	2005	2006		10	2010
Yellowtail flounder - Southern New England / Mid-Atlantic	NEFMC	Yes	Yes	Rebuilt	No	2000	2004	2012	10	2011
Barndoor skate - Georges Bank / Southern New England	NEFMC	No	No	Rebuilding	No	2000	2003		Not specified	2010
Smooth skate - Gulf of Maine	NEFMC	No	No	Rebuilding	No	2000 & 2008	2010	2001	7	2010
Thorny skate - Gulf of Maine	NEFMC	No	No	Overfished	No	2000	2003		25	2010
Blue king crab - Pribilof Islands	NPFMC	Yes	No	Overfished	No	2003	2004		10	2010
Bocaccio - Southern Pacific Coast	PFMC	Yes	Yes	Rebuilding	No	1999	2000		22	2010
Canary rockfish - Pacific Coast	PFMC	Yes	Yes	Overfished	No	2000	2001		26	2010
Cowcod - Southern California	PFMC	Yes	Yes	Overfished	No	2000	2001		67	2008
Darkblotched rockfish - Pacific Coast	PFMC	Yes	Yes	Rebuilding	No	2001	2002		23	2010
Pacific ocean perch - Pacific Coast	PFMC	Yes	Yes	Overfished	No	1999	2000		18	2010
Petrale sole - Pacific Coast	PFMC	No	Yes	Rebuilding	No	2009	2012		4	2010
Yelloweye rockfish - Pacific Coast	PFMC	Yes	Yes	Overfished	No	2002	2003		71	2010
Black sea bass - Southern Atlantic Coast	SAFMC	Yes	Yes	Rebuilding	Yes	1999	2000		10	2010
Red porgy - Southern Atlantic Coast	SAFMC	Yes	Yes	Overfished	No	2000	2000		16	2004

Red snapper - Southern Atlantic Coast	SAFMC	Yes	Yes	Overfished	Yes	2000	2001		35	2009
Snowy grouper - Southern Atlantic Coast	SAFMC	No	Yes	Overfished	Yes	2000	2006		34	2002
Hancock Seamount Groundfish Complex	WPFMC	No	No	Overfished	Undefined	2000	1986		Not specified	
Red grouper - Gulf of Mexico	GMFMC	Yes	Yes	Rebuilt	No	2000	2003	2007	10	2008
Blacktip shark - Gulf of Mexico	HMS	No	Yes	Rebuilt	No	1999		2006	30	2004
Blacktip shark - South Atlantic	HMS	No	No	Undefined	Undefined	1999		2003	30	2004
Swordfish - North Atlantic	HMS	Yes	Yes	Rebuilt	No	1997	1999	2009	10	2008
Bluefish - Atlantic Coast	MAFMC	Yes	Yes	Rebuilt	No	1999	2001	2008	9	2010
Black sea bass - Mid-Atlantic Coast	MAFMC	Yes	Yes	Rebuilt	No	2000	2000	2009	10	2010
Scup – Atlantic Coast	MAFMC	Yes	Yes	Rebuilt	No	2000	2008	2009	7	2010
Summer flounder - Mid-Atlantic Coast	MAFMC	Yes	Yes	Rebuilt	No	1999	2000	2011	13	2010
Sea scallop - Northwestern Atlantic Coast	NEFMC	Yes	Yes	Rebuilt	No	1997	1999	2001	10	2009
Haddock - Georges Bank	NEFMC	Yes	Yes	Rebuilt	No	2000	2004	2010	10	2010
Haddock - Gulf of Maine	NEFMC	Yes	Yes	Approaching overfished	Yes	1999	2004	2011	10	2010
Pollock - Gulf of Maine / Georges Bank	NEFMC	Yes	Yes	Rebuilt	No	2002	2004	2010	10	2009
Silver hake - Gulf of Maine / Northern Georges Bank	NEFMC	No	No	Rebuilt	No	1997	1999	2002	10	2009
Silver hake - Southern Georges Bank / Mid-Atlantic	NEFMC	No	No	Rebuilt	No	1997	1999	2007	10	2009
Goosefish - Gulf of Maine / Northern Georges Bank	NEFMC / MAFMC	Yes	Yes	Rebuilt	No	1997	1999	2008	10	2009
Goosefish - Southern Georges Bank / Mid-Atlantic	NEFMC / MAFMC	Yes	Yes	Rebuilt	No	1997	1999	2008	10	2009
Spiny dogfish - Atlantic Coast	NEFMC / MAFMC	Yes	Yes	Rebuilt	No	1998	2000	2010	5	2010
Blue king crab - Saint Matthews Island	NPFMC	Yes	Yes	Rebuilt	No	1999	2000	2009	10	2010
Snow crab - Bering Sea	NPFMC	Yes	Yes	Rebuilt	No	1999	2000	2011	10	2010
Lingcod - Pacific Coast	PFMC	Yes	Yes	Rebuilt	No	1999	2000	2005	10	2008
Pacific hake - Pacific Coast	PFMC	Yes	Yes	Rebuilt	No	2002		2004	Not specified	2011
Widow rockfish - Pacific Coast	PFMC	Yes	Yes	Rebuilt	No	2001	2002	2011	14	2010
Chinook salmon - Northern California Coast: Klamath (fall)	PFMC	No	Yes	Rebuilt	No			2011	Not specified	2003
Coho salmon - Washington Coast: Queets	PFMC	No	Yes	Rebuilt	No	2009		2011	Not specified	2008
King mackerel - Gulf of Mexico	SAFMC / GMFMC	Yes	Yes	Rebuilt	No	1999	1987	2008	Not specified	2006

Yellowtail snapper - Southern Atlantic Coast / Gulf of Mexico	SAFMC / GMFMC	Yes	Yes	Rebuilt	No	2000	1992	2003	10	2010
Vermilion snapper - Gulf of Mexico	GMFMC	Yes	Yes	Rebuilt	No	2003	2004	2006	10	2010
Winter skate - Georges Bank - Southern New England	NEFMC	No	No	Rebuilt	No	2000 & 2007		2010	Not specified	
Gag – Gulf of Mexico	GMFMC	No	Yes	Overfished	Yes	2009	2012		10	2008
Scalloped hammerhead – Atlantic	HMS	No	Yes	Overfished	Yes	2011			Not specified	2005
Southern Tanner crab - Bering Sea	NPFMC	Yes	Yes	Overfished*	No	1999 & 2010	2000	2007	10	2011
Chinook salmon - California Central Valley: Sacramento (fall)	PFMC	No	Yes	Overfished	No	2010	2012		1	2003
Coho salmon - Washington Coast: Western Strait of Juan de Fuca	PFMC	No	Yes	Rebuilt	No	2009		2012	Not specified	2007
Red grouper - Southern Atlantic Coast	SAFMC	Yes	Yes	Overfished	Yes	2000	1992		15	2008

*****Stock re-classified as rebuilt in the last (quarter 4) 2012 status report to Congress based on stock assessment estimates included in this report.

Rebuilding Plans were adopted for most stocks that were declared to be overfished within two years (Figure 3.5). However, the time between overfished designation and implementation of a Rebuilding Plan has extended up to 10 years. Reasons for longer time periods between the declaration of overfished status and the establishment of a Rebuilding Plan include the delays associated with court decisions (e.g., Dusky shark and sandbar shark[7]), insufficient information to develop a Rebuilding Plan (e.g., Atlantic halibut), and differences in policy and law associated with stocks that are internationally managed (e.g., albacore and bigeye tuna in the Atlantic). Rebuilding Plans were not established for some stocks (e.g., Pacific hake, *Merluccius productus*, and Coho salmon, *Oncorhynchus kisutch*) because they were reclassified as rebuilt before a Rebuilding Plan could be developed and adopted (Table 3.2).

The length of the target rebuilding period (T_{TARGET}) differs among stocks (Figure 3.6). As expected, a large number of Rebuilding Plans are designed to rebuild stocks within 10 years. However, some plans focus on shorter rebuilding times, while others allow longer rebuilding times because recovery projections under zero fishing mortality indicate that rebuilding cannot occur with at least 50% probability within the 10-year rebuilding time threshold. Time horizons for rebuilding were not specified for nine stocks, which correspond to highly migratory stocks in the Atlantic (subject to international rebuilding plans), groundfish on Hancock Seamount (a species complex), barndoor skate (managed as part of a skate complex and only caught as bycatch), and Atlantic wolfish (too much uncertainty to select a rebuilding time). No rebuilding time was specified for king mackerel in the Gulf of Mexico because the Rebuilding Plan for this species was developed before the SFA.

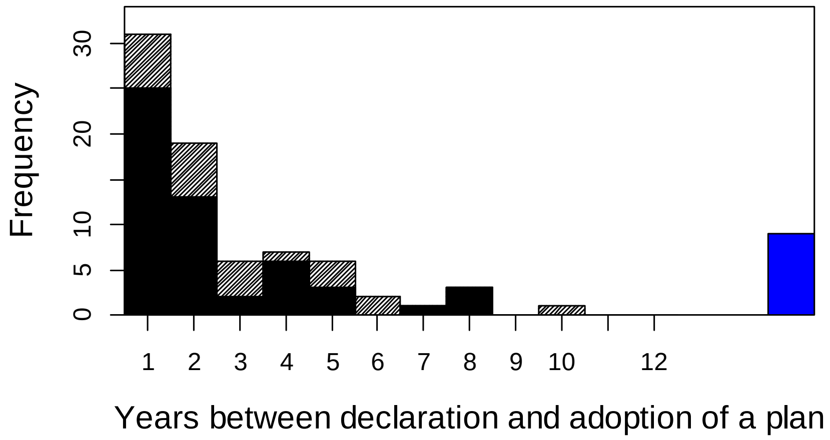

FIGURE 3.5: Number of years between a stock being declared overfished and the adoption of a Rebuilding Plan for the 85 stocks declared overfished since 1997. The solid bars denote the 55 stocks considered in detail in the analyses. The blue bar corresponds to stocks than rebuilt before a Rebuilding Plan was developed (5 stocks) or adopted (implementation delayed due to court decisions in 2 cases), or a rebuilding Plan has not yet been developed (2 stocks).

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[7] http:// http://www.nmfs.noaa.gov/sfa/statusoffisheries/Archives/StatusofFisheriesReportCongress2002.pdf

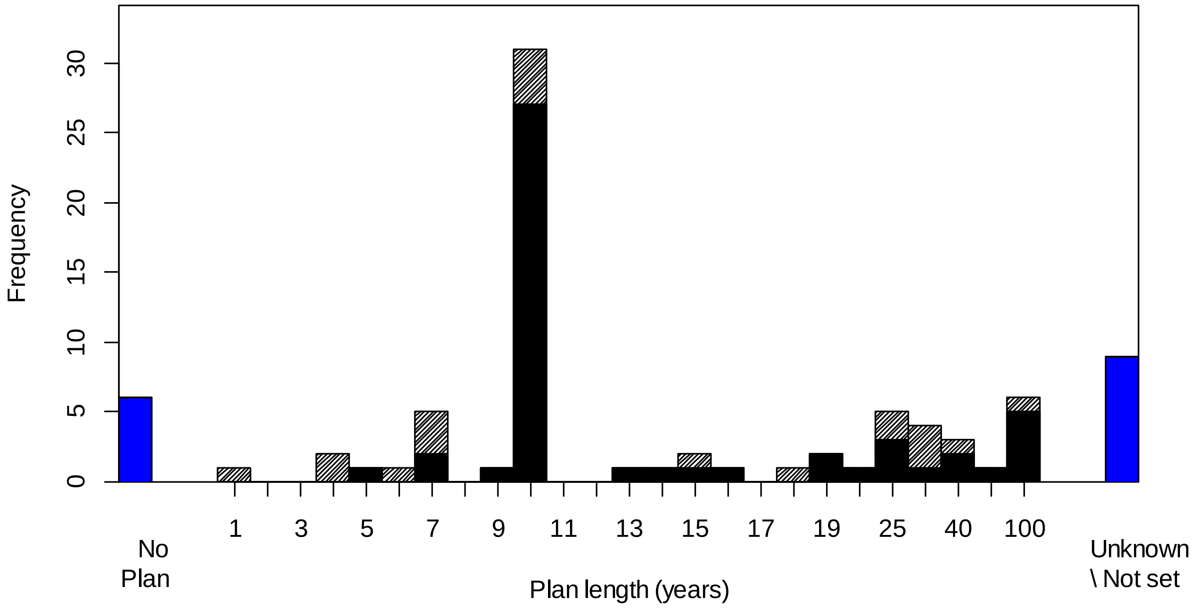

FIGURE 3.6: Length of the rebuilding plan for the 85 stocks considered in this study. Bars are for individual years until 20 years, and then for the following groups of years: 21-25 years, 26-30 years, 31-40 years, 41-50 years, and 51+ years. The solid bars denote the 55 stocks considered in detail in the analyses. Bars are shown in blue for the stocks for which a Rebuilding Plan was never developed and for the stocks for which a desired time to rebuild has never been set.

The majority of the adopted Rebuilding Plans are based on a constant fishing mortality strategy. The target fishing mortality (F_{ACL}) used to calculate the Accepted Biological Catches (ABC) depend on the $\mathrm{T}_{\mathrm{TARGET}}$ selected by the RFMC, which must be between $\mathrm{T}_{\mathrm{MIN}}$ and $\mathrm{T}_{\mathrm{MAX}}$. The F_{ACL} has to be lower than F_{MSY}, but how much lower depends on how each Council resolves the various trade-offs involved in the decision. For example, Figure 3.7 compares the values of F_{ACL}, scaled by F_{MSY}, used to calculate the 2012 ABCs for groundfish stocks subject to Rebuilding Plans in New England and in the Pacific coast. In general, the PFMC has selected rebuilding periods ($\mathrm{T}_{\mathrm{TARGET}}$) substantially shorter than the maximum established by the NS1G, and has therefore chosen lower values of F_{ACL} as a fraction of F_{MSY}. Petrale sole has a higher $F_{\mathrm{ACL}}/F_{\mathrm{MSY}}$. However, the Rebuilding Plan for this stock is not based on a constant fishing mortality strategy, but on the standard control rule used for stocks that are not overfished (the 25-5 rule[8]) (PFMC, 2011b), because rebuilding analyses showed that this rule was adequate to allow rebuilding to B_{MSY} within 10 years (Haltuch, 2011).

The rule followed by the New England Fishery Management Council (NEFMC)[9] set F_{ACL} by selecting the lower of 0.75 F_{MSY} or F_{REBUILD}, where the latter is the fishing mortality

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[8] The 25-5 harvest control rule is designed to prevent flatfish stocks from becoming overfished and serves as an interim rebuilding policy for stocks that are below the overfished threshold. This rule sets the ACL to the catch corresponding to F_{MSY} when the stock is at 25% of the unfished level (less a buffer to account for scientific uncertainty) and to 0 at 5% of this level.

[9] http://www.nero.noaa.gov/nero/regs/frdoc/12/12MulFW47EA.pdf

rate that achieves a 50% probability of rebuilding by the target year. In most cases for NEFMC stocks, the maximum allowable rebuilding period has been selected (typically 10 years) and F_{ACL} has been set at 75% F_{MSY}, but much larger reductions have been adopted when a stock was not rebuilding at the expected rate and the end of the rebuilding time frame was approaching. This was the case of the Southern New England / Mid-Atlantic stocks of yellowtail flounder and winter flounder, for which the original T_{TARGET} was set at 2014. The F_{ACL} used to calculate the 2012 ABC was 28% F_{MSY} for yellowtail flounder, corresponding to F_{REBUILD}, and 24% of F_{MSY} for winter flounder. The latter stock would not rebuild by 2014, so the NEFMC opted in 2009 to reduce fishing mortality as much as possible without closing the other fisheries in the area.

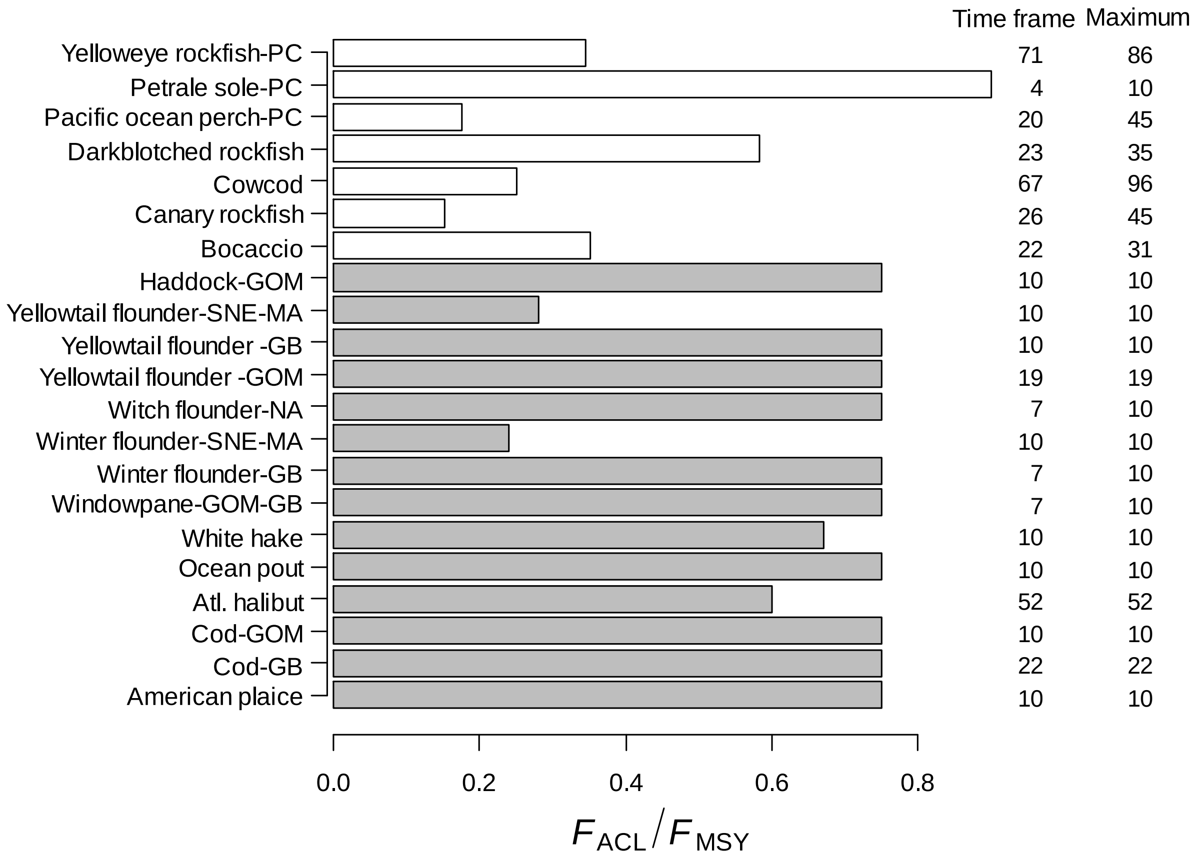


FIGURE 3.7: Target fishing mortalities (F_{ACL}) used to calculate 2012 Acceptable Biological Catches for groundfish stocks subject to Rebuilding Plans under the NEFMC (grey bars) and the PFMC (white bars), and the corresponding length of Rebuilding Plan. The values under the heading Maximum correspond to the maximum permissible rebuilding period. Fishing mortalities are scaled to estimated F_{MSY}. Rebuilding Plans are based on a constant fishing mortality strategy except for Petrale sole.

OVERVIEW OF OUTCOMES OF REBUILDING EFFORTS

Changes in reported stock status over time

Figure 3.8 shows trends in the stock status summaries for the 85 stocks in Table 3.2 relative to being in an overfished state. The status of most of these stocks was not determined until 1999-2000 when the majority of the overfished designations occurred; thus the number of stocks whose status was undefined declined markedly between 1998 and 2000. The total number of stocks classified as overfished peaked in 2002 at 55 and has declined to an average of 45 since 2004. Although the rate of new overfished designations decreased markedly after 2000, an average of six new stocks per year has been designated as overfished during 2004-2010. The number of stocks rebuilt has increased consistently over time while the number of stocks in rebuilding status (MSST < B < B_{MSY}) has been relatively constant since 2004. Of the 85 stocks declared overfished since 1997, 42 are no longer classified as overfished: 11 are rebuilding and 31 were subsequently designated as rebuilt, one of which is currently considered undefined (Table 3.3). Four additional stocks that were declared rebuilt became overfished again and one is approaching overfishing (Table 3.2).

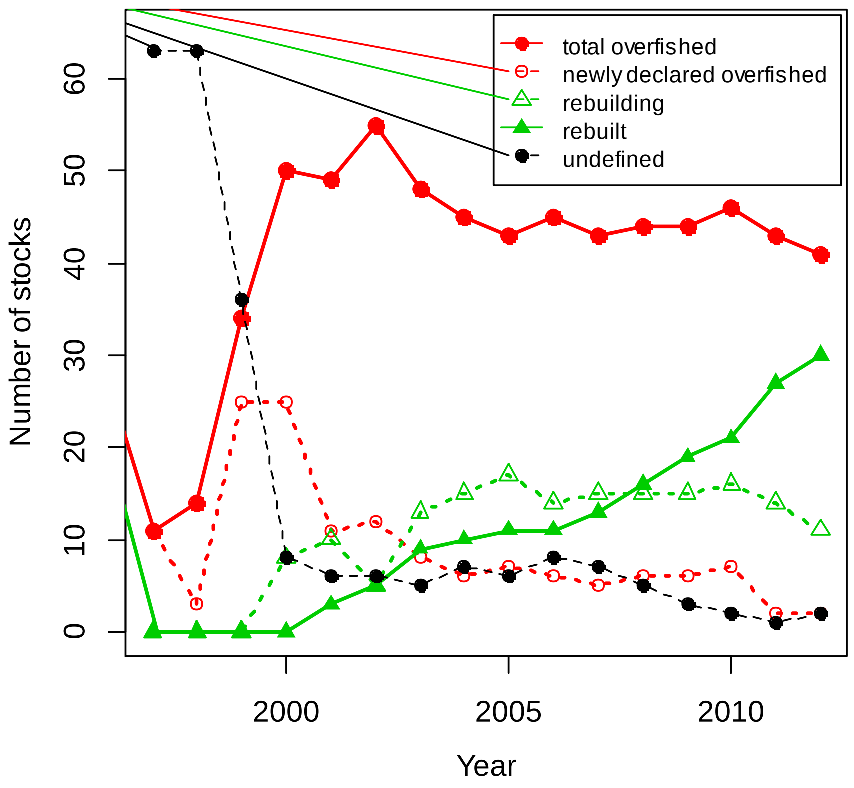


FIGURE 3.8: Number of stocks classified in the different biomass status categories each year for the set of 85 stocks declared overfished over the period 1997-2012.

TABLE 3.3: Summary of the current status of stocks that have been declared to be overfished under the SFA. Stocks are categorized by region.

Council	Number of stocks classified as overfished since 1997	FSSI stocks	Current status relative to MSST and B_{MSY}					Number subject to overfishing ($F > F_{MSY}$)	With analytical assessments	Number included in the evaluation set
			Number overfished (< MSST)	Number rebuilt (> B_{MSY})	Number approaching an overfished state	Number Rebuilding (> MSST & < B_{MSY})	Number Undefined			
New England	26	25	12	9	1	4	0	7	20	17
New England / Mid-Atlantic	3	3	0	3	0	0	0	0	3	3
Mid-Atlantic	6	6	0	4	0	1	1	0	6	5
South Atlantic	5	5	4	0	0	1	0	3	5	4
South Atlantic / Gulf	2	2	0	2	0	0	0	0	2	2
Gulf of Mexico	6	6	4	2	0	0	0	3	6	4
Caribbean	4	4	4	0	0	0	0	3	0	0
Pacific	14	10	5	6	0	3	0	0	14	9
North Pacific	4	4	2	2	0	0	0	0	4	3
Western Pacific	1	1	1	0	0	0	0	0	0	0
High Migratory	14	13	9	2	0	2	1	8	12	7
Total	85	79	41	30	1	11	2	24	72	55

For stocks that were classified as rebuilt, the number of years from being designated as overfished to being declared rebuilt was often less than was anticipated (Figure 3.9). In five of those cases (Pacific hake, Georges Bank winter skate, and three salmon stocks) the stocks were assessed to be rebuilt before a Rebuilding Plan was implemented. As discussed in the next section, some of these designations as 'rebuilt' correspond to situations when stock status was re-evaluated following an updated stock assessment rather than as the direct result of evidence of recovery following reduced fishing mortality.

The current status of the stocks that were declared overfished varies by region (see Table 3.3 for a summary). New England was the region that had the largest number of stocks declared overfished under the SFA (26 of the 85), followed by the Pacific west coast and the highly migratory stocks (14 stocks each). The contrast among regions is still present now, with New England still showing the largest number of overfished stocks despite several stocks rebuilding. The success in eliminating overfishing has also varied across regions. For example, none of stocks designated as overfished and managed by the PFMC and NPFMC are currently subject to overfishing, while overfishing is still taking place in seven stocks managed by the NEFMC and in eight stocks of the highly migratory category.

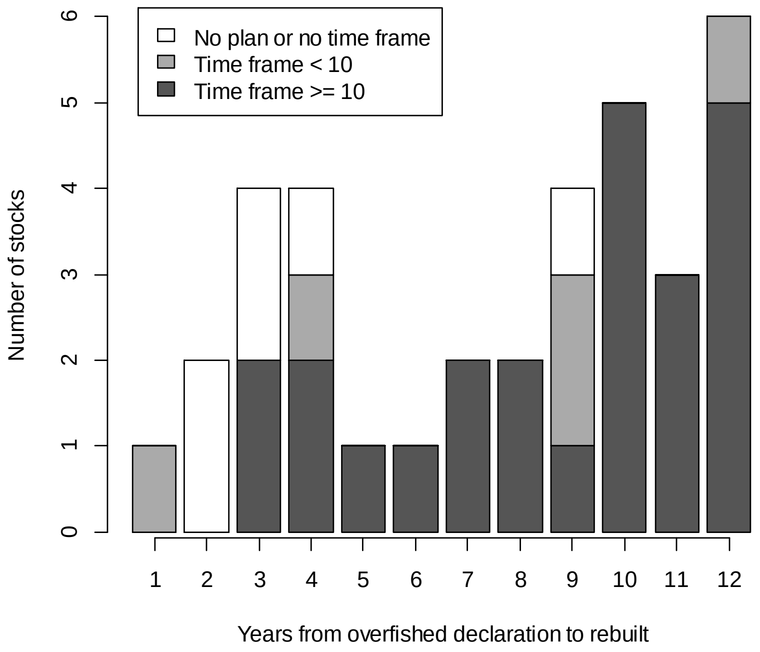


FIGURE 3.9: Frequency distribution of number of years between being declared to be overfished and rebuilt for the 85 stocks declared overfished since 1997. The figure is truncated at 12 years because insufficient time has passed to fill in the rest of the distribution.

Analysis of time series data

To evaluate the changes in fishing mortality and biomass following declaration of overfished status we used a subset of stocks that are assessed by quantitative methods. Of the 85 stocks declared overfished since 1996, 72 (85%) had a quantitative stock assessment that estimated biomass, fishing mortality, and their respective reference points B_{MSY} and F_{MSY} or their proxies (Table 3.2). Such stock assessments could have formed the basis for forecasting under different management arrangements. The analyses of trends and outcomes of management actions presented below are based on a subset of 55 of the 85 stocks in Table 3.2 for which (a) quantitative stock assessments are available, and (b) there are at least three years between when the stock was declared overfished (year YD) and the most recent year for which estimates of F/F_{MSY} and B/B_{MSY} are available. Time series of catches, fishing mortality, biomass and recruitment, and phase plane plots for all these stocks are provided in Appendix C to this report. The data and assessment estimates were assembled from the Species Information System database available at NOAA (as of September of 2012), complemented by information provided by individual assessment scientists at NOAA's Fishery Science Centers. Fishing intensity is expressed in several ways depending on the stock. In some cases, the actual fishing mortality rate for fully-recruited age classes is provided; in other cases fishing intensity is expressed as a proportional reduction in spawning biomass-per-recruit relative to that when the stock was unfished. Similarly, biomass may correspond to female spawning stock biomass, number of eggs or even male biomass (in the case of crab species). Trends in estimated F/F_{MSY} and B/B_{MSY} before and after the year each stock was declared to be overfished are shown in Figure 3.10, by region. These trends are used to evaluate (1) changes in initial stock status as a result of assessment updates, (2) estimated changes in F and (3) changes in B, in that order.

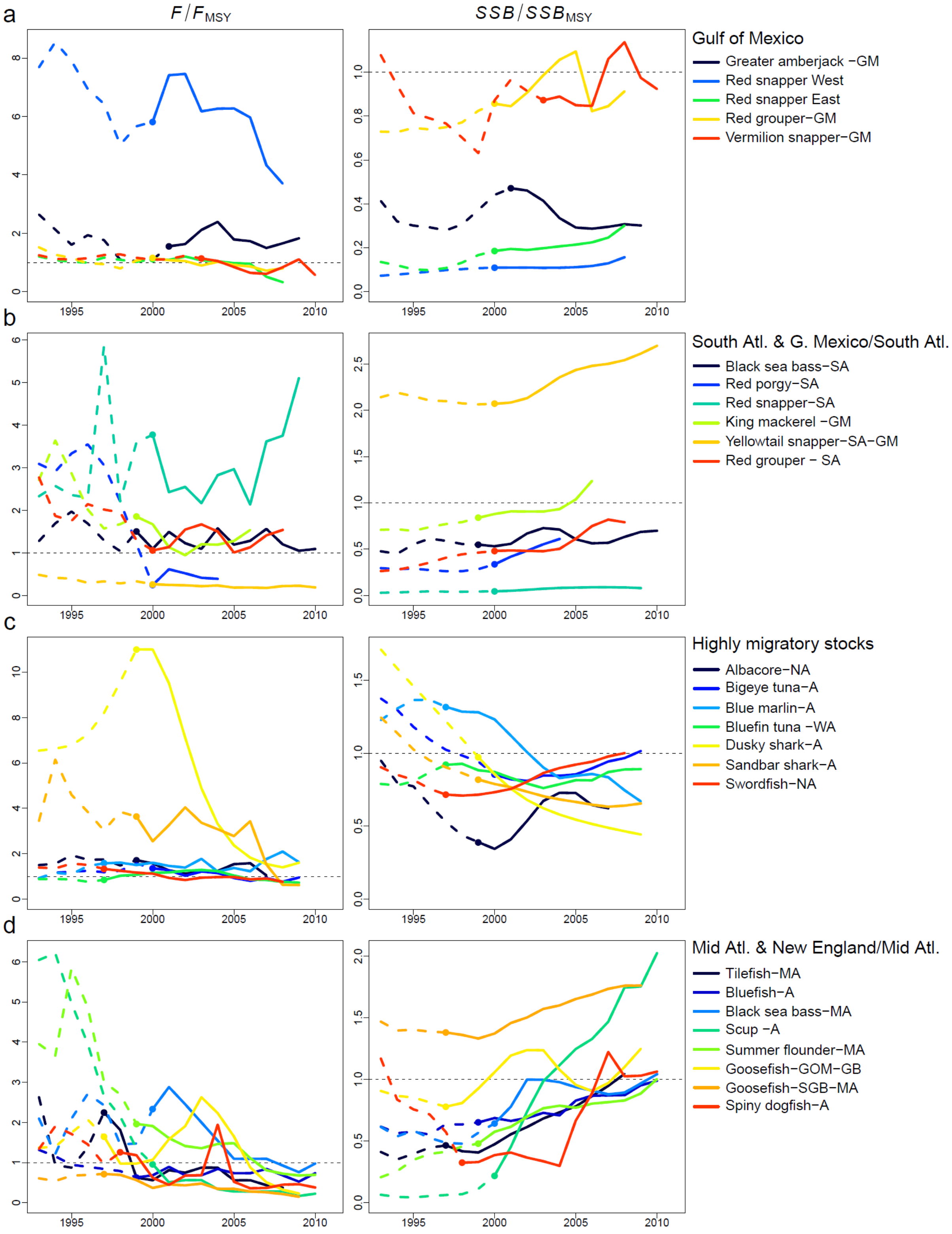


FIGURE 3.10: Trends in F/F_{MSY} and B/B_{MSY} since 1993 estimated by the most recent assessment for stocks in different jurisdictions. Dashed lines and solid lines correspond, respectively, to the periods before and after the year each stock was declared to be overfished (identified by a solid dot).

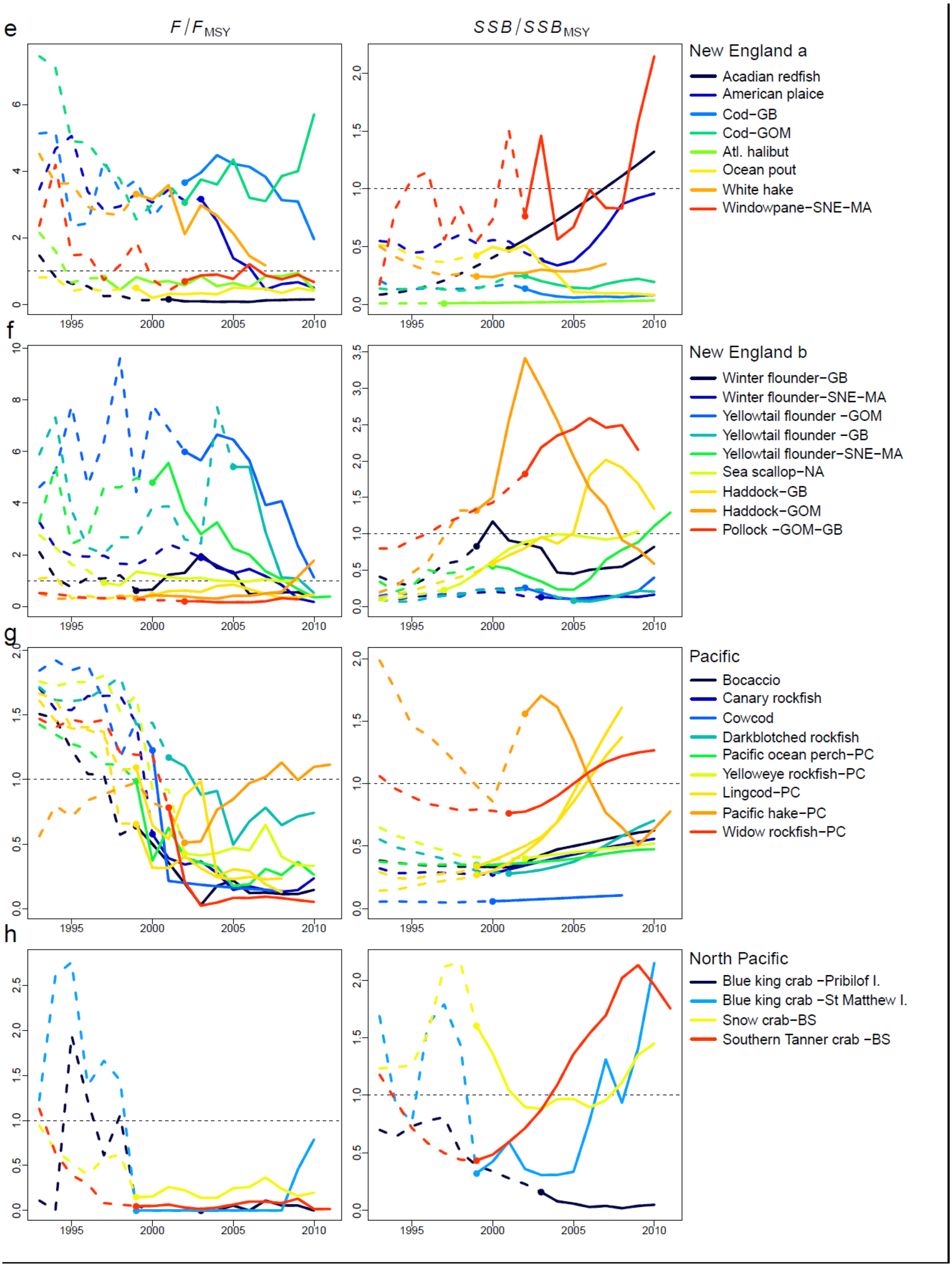


FIGURE 3.10: (cont.)

Status at the time the stock was declared overfished

Changes in the number of stocks classified in the different status categories reflect not only actual changes in stock status, but also the fact that when stock assessments are updated with new data, and perhaps under somewhat different assumptions, their original status category in a given year may change. Thus, to understand the responses to management actions in terms of fishing mortality and biomass, it is important to consider the status of each of the stocks at the time of overfished designation in the light of its most recent assessment. It should be noted that this perspective presumes that the most recent assessment is the most correct, but there is no way to guarantee this, and the variation in stock assessment results suggests that the results would likely be somewhat different were the comparisons below to be conducted in the future based on further updated assessments.

The changes in status of the 55 stocks evaluated relative to F_{MSY}, B_{MSY} and MSST based on the most recent stock assessments are summarized in Table 3.4, and Figures 3.11 and 3.12. Although at the time these stocks were declared overfished their biomass was estimated to be below the MSST, the most recent assessments indicate that 20 (36%) of these stocks were not overfished in the year before designation (YD-1), and 10 were actually above B_{MSY} (Table 3.4; Figure 3.11). Only 26 of the 55 stocks were both overfished and experiencing overfishing when declared overfished according to the latest assessments. Of the 21 stocks that would now be classified as rebuilt based on these assessments, six were already above B_{MSY} at the time of overfished designation and five more were below B_{MSY} but not overfished, according to the same assessment. The reason for these results is that every time a stock assessment is updated with new data the entire time series of F and B estimates change, including the F and B at time YD-1 and likely the value of the reference point B_{MSY}. In some cases, these changes are substantial, leading to a re-classification of the original stock status. This was for example the case of Pacific hake: although the stock was classified as overfished in 2002, updated assessments indicated that the stock was above B_{MSY} when declared to be overfished (see Figure C.33, Appendix C). It should also be recognized that the uncertainty in stock assessment results is such that some stocks currently considered not to be overfished may in fact be overfished.

TABLE 3.4: Status of the 55 stocks relative to F_{MSY}, B_{MSY} and MSST based on the most recent stock assessments. Results are shown for the year before the stock was declared overfished, three years after the overfished designation and in the most recent year with information on stock status.

	B<MSST	MSST≤B<B_{MSY}	B≥B_{MSY}	Total
(a) Status one year before being designated as overfished				
$F > F_{MSY}$	26	7	3	36
$F \leq F_{MSY}$	9	3	7	19
Total	35	10	10	55
(b) Status three years after being designated as overfished				
$F > F_{MSY}$	17	6	2	25
$F \leq F_{MSY}$	12	13	5	30
Total	29	19	7	55
(c) Status at last year covered by the assessment				
$F > F_{MSY}$	11	3	1	15
$F \leq F_{MSY}$	11	9	20	40
Total	22	12	21	55

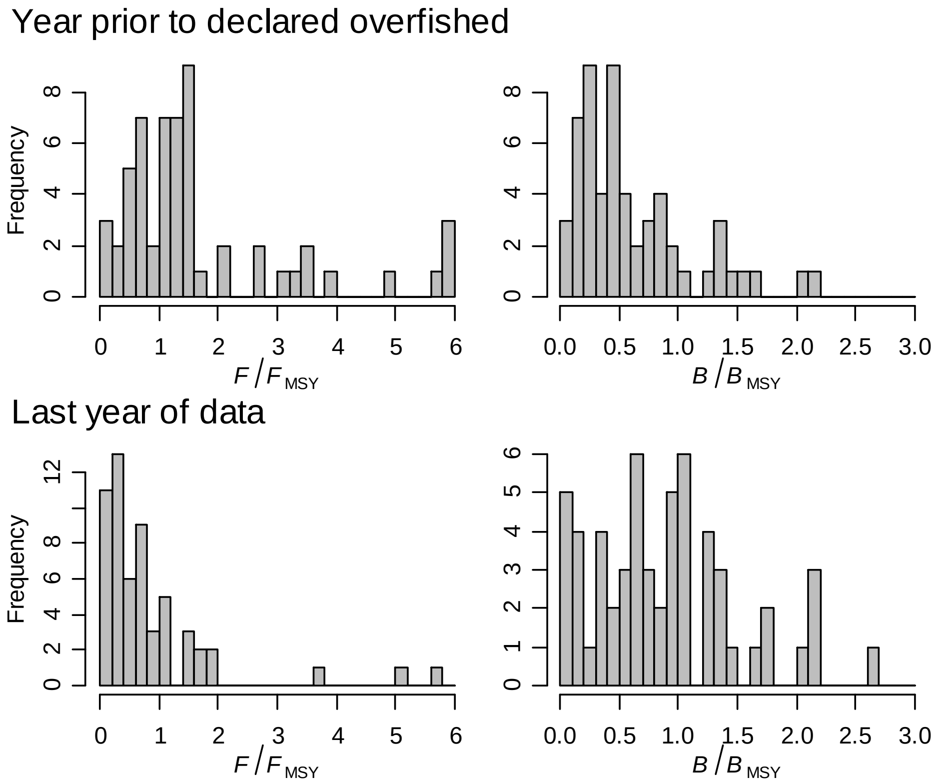
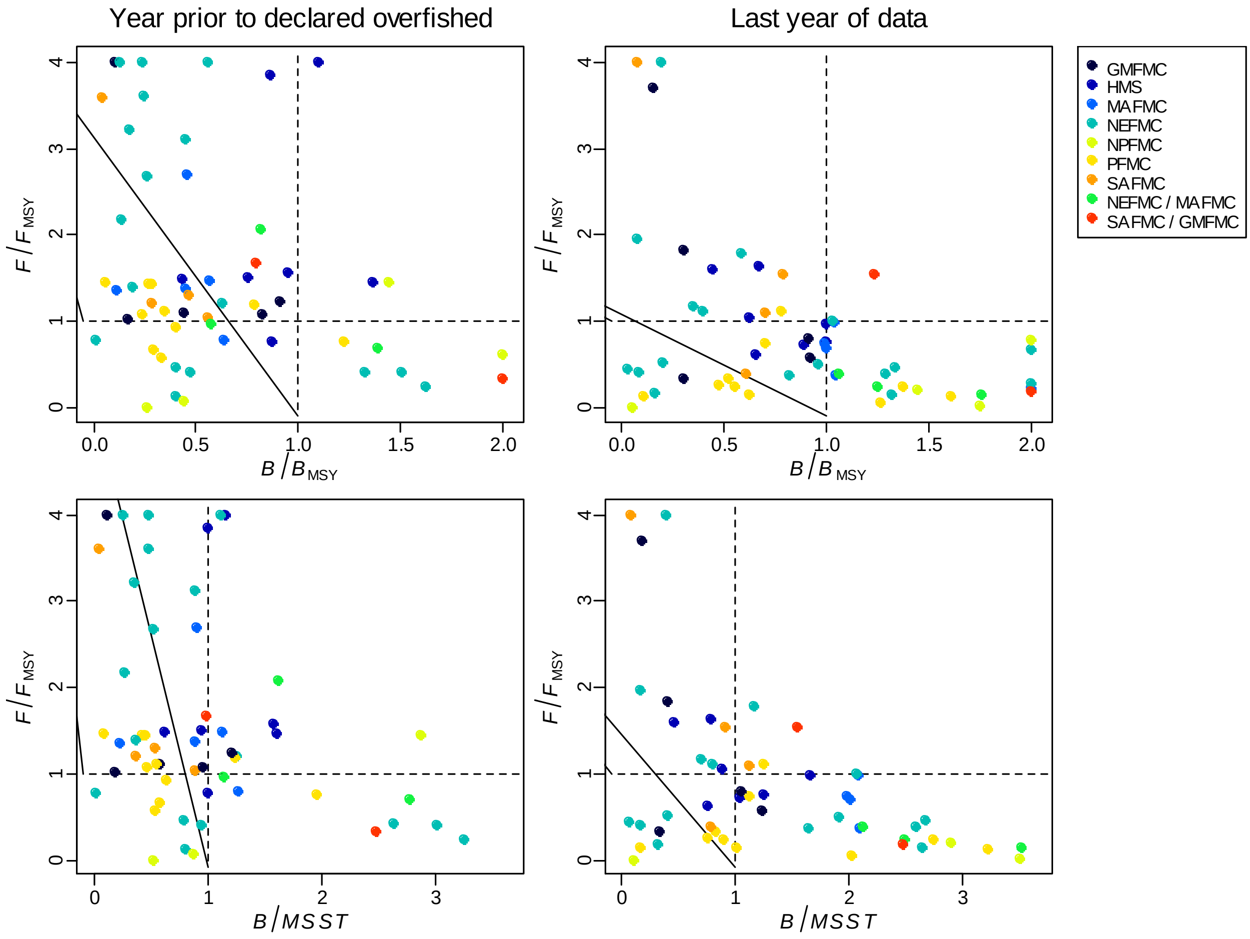


FIGURE 3.11: Distribution of stock status in terms of F/F_{MSY} and B/B_{MSY} the year before the stock was declared to be overfished and in the last year included in the assessment, all based on the most current assessments. Values of $F/F_{MSY} > 6$ are included in the largest class.

The initial status at the time of overfished designation varied across the regions, with overfished stocks in the Atlantic generally being subject to much higher levels of overfishing than in the Pacific (Figure 3.12)


Year prior to declared overfished

Last year of data

FIGURE 3.12: Stock status in terms of F/F_{MSY}, B/B_{MSY} and B/MSST the year before the stock was declared to be overfished and during the last year for which assessment information is available, based on estimates from the most recent assessment. Each dot represents a stock, and different colors correspond to Council jurisdictions. The scale of the F/F_{MSY} axis is truncated so that the dots at the upper limit correspond to $F/F_{MSY} > 4$.

Changes in fishing mortality

Of the 35 stocks that are now estimated to have been below MSST the year before they were declared overfished, 9 had fishing mortalities below F_{MSY} (Table 3.4). So, according to the most recent assessments, these stocks were overfished, but overfishing was not taking place. Reasons differ among stocks. Some stocks were depleted many years before they were declared overfished. This is the case for Atlantic halibut, which essentially was not targeted by any fishery when it was declared overfished (Figure C.4, Appendix C). Also, fishing mortality on ocean pout dropped below F_{MSY} in 1992, well before the stock was declared overfished under the SFA (Figure C.32, Appendix C). For other stocks, there were pre-emptive management actions preceding the overfished declaration. For example, the directed fishery for Pribilof Islands blue king crab was closed in 1999 by the State of Alaska when stock size and catch-rates dropped, although the stock was only declared overfished in 2003 (see Figure C.10, Appendix C). Yet in other cases, reductions in *F* were a byproduct of measures implemented to rebuild other stocks. This is the case of yelloweye rockfish (Figure 3.10g), declared overfished in 2002, whose fishing mortality had been reduced because of the restrictions placed on the west coast groundfish fishery to rebuild other "shelf" rockfish, in particular canary rockfish, which was declared overfished in 1999 (Wallace, 2001).

As expected, fishing mortality for most stocks dropped following the designation as overfished. According to current assessments, the fishing mortality at the time of overfished designation exceeded F_{MSY} for 36 stocks; this number dropped to 25 after three years and is 15 at present (Table 3.4, Figure 3.11). In the majority of the stocks for which overfishing was taking place, fishing mortality dropped either before or soon after the overfished designation, but the extent of the drop differed among regions (Figure 3.10). The reduction in *F* in the Pacific (Figure 3.10g, h) was more consistent across stocks and occurred earlier (around year 2000) than in the North Atlantic (Figure 3.10e, f). It should be noted also that the extent of overfishing was more severe for many of the Atlantic stocks than was the case in other regions: fishing mortality was well above F_{MSY} when stocks were declared overfished in the Atlantic, whereas in the Pacific it was generally either slightly above or below F_{MSY} at this time.

Although the decrease in *F* over the first three years after overfished designation tended to be larger the higher the initial F/F_{MSY} (Figure 3.14), in some cases it was insufficient to stop overfishing (dots above the solid line). Furthermore, in five stocks (GoM yellowtail flounder, the two cod stocks, GM red snapper and sandbar shark), *F* continued to increase even though it was more than three times F_{MSY} in the year of overfished designation.

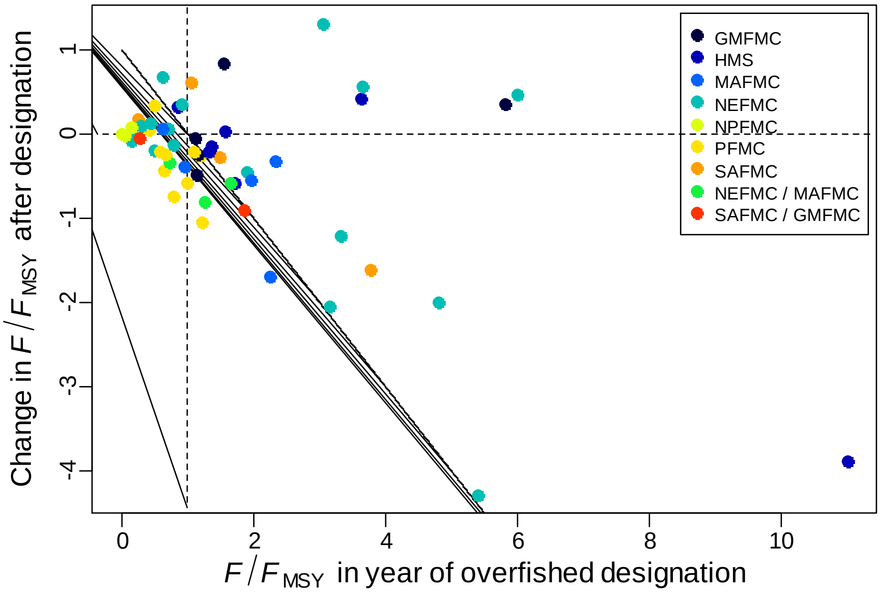


FIGURE 3.13: Change in fishing mortality relative to F_{MSY} following overfished designation as a function of F/F_{MSY} in the year when the stock was declared overfished (YD). Each point corresponds to a stock, colored coded by RFMCs. The change in F/F_{MSY} was calculated as F_{YD}/F_{MSY} -F_{YD+3}/F_{MSY}, i.e., the difference between the relative *F*s estimated for YD and three years later. The solid line, which has slope -1, indicates the change that would have been required to stop overfishing when F/F_{MSY} was larger than one (points to the right of the vertical dashed line). Points above the solid line correspond to stocks for which overfishing was taking place three years after they were designated as overfished.

When overfishing was successfully stopped, the number of years it took for fishing mortality to be reduced below F_{MSY} ranged from 2 to 10 years (Figure 3.14).

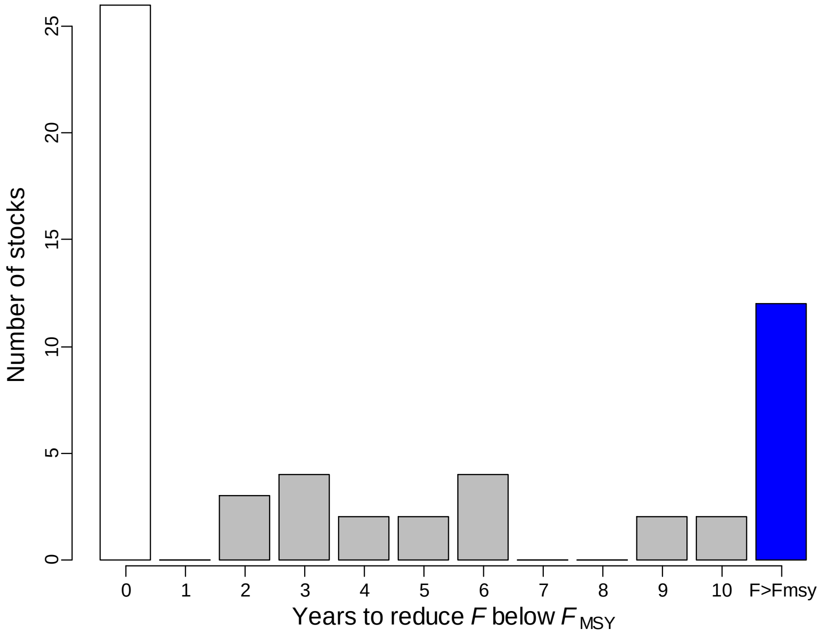


FIGURE 3.14: Number of years since each stock was declared to be overfished for fishing mortality to be reduced below F_{MSY}, according to the most recent assessment. Results are for the 55 stocks selected for detailed evaluation. The blue bar corresponds to stocks that were still subject to overfishing ($F > F_{\mathrm{MSY}}$) in the last year covered by the assessment. The white bar corresponds to stocks not subject to overfishing at the time of declaration according to current assessments.

Various factors lead to management not always being effective at stopping overfishing. In particular, the relatively poor performance for stocks managed by the NEFMC can be attributed to the combined effects of delays implementing Rebuilding Plans, difficulties implementing reduced target fishing mortalities, and biases in the stock assessments for some of the assessed stocks. Prior to the 2006 amendment, which required the use of ACLs, the primary means to implement the rebuilding target *F*s was through the control of days-at-sea and time/area closures. While days-at-sea were adjusted to try to not exceed a Target Total Allowable Catch (TTAC) derived from model projections under F_{ACL}, the catch was only controlled indirectly, and the fishery was not closed if the TTAC was exceeded. Before 2002, catches of Georges Bank and Gulf of Maine cod and yellowtail flounder routinely exceeded the respective TTACs, but new, more effective effort controls were introduced in 2004 and catches have been generally kept below the TTACs since then, except for SNE-MA windowpane and yellowtail flounder, and to a lesser extent Georges Bank yellowtail flounder (Figure 3.15). In the case of the two cod stocks, the TTACs were not exceeded. Thus, the failure of the rebuilding plans for these species (that were started in 2004) was not due to catches above the target amount.

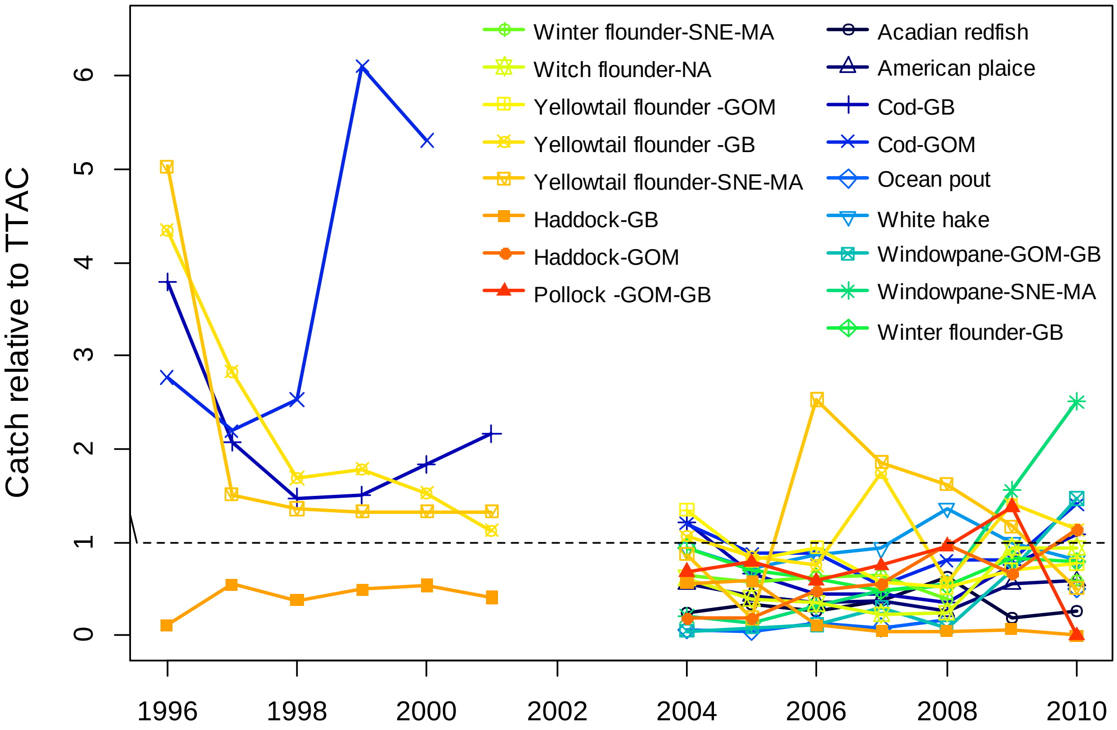


FIGURE 3.15: Catches of New England groundfish relative to Target Total Allowable Catches (TTAC) or ACLs (since 2007). TTACs were not set in 2002-2003. Catches from Georges Bank correspond to U.S. catches only. In all cases, data correspond to calendar-year catches.
SOURCE: NEFMC (data provided by Tom Nies)

In terms of stock assessment biases, there has been a tendency for some assessments to consistently overestimate stock size and underestimate fishing mortality (Legault, 2009). This tendency has been referred to as a “retrospective pattern”, a problem that has affected several stock assessments, primarily but not exclusively in New England. Figure 3.16 illustrates the problem with four NEFMC stocks for which current F is estimated to be higher than F_{MSY}. The lines in each panel correspond to the time series of estimates of F/F_{MSY} and B/B_{MSY} produced by successive stock assessments; the end point of each line is the value used to determine stock status in the year when the assessment was conducted, and used as initial value to conduct stock projections for setting catch limits. It is expected that the estimates of historical F and B will vary when assessments are updated using new data. However, there is a tendency for successive updates to always be in the same direction, i.e., there is a retrospective pattern indicative of a bias in the estimates. In the examples provided, most notably in the cases of Georges Bank cod and yellowtail flounder, historical F estimates have been adjusted upwards in each assessment update, and the opposite has occurred to the estimates of historical spawning stock biomass (*SSB*). This implies that the biomasses used to set allowable catches have been overestimated, leading to catches that were too high, and fishing mortalities well in excess of the F rebuilding targets. We note, however, that while this bias contributed to overfishing, at no time during this period were the initial retrospective estimates of F below the most recent estimates of F_{MSY}. The source of the retrospective pattern is not always clear, and thus the root of the problem cannot be corrected though modeling or data standardization. An *ad hoc* downward adjustment has been applied to the current estimates of *SSB* in cases of persistent retrospective patterns to avoid exceeding the target Fs (Legault, 2009).

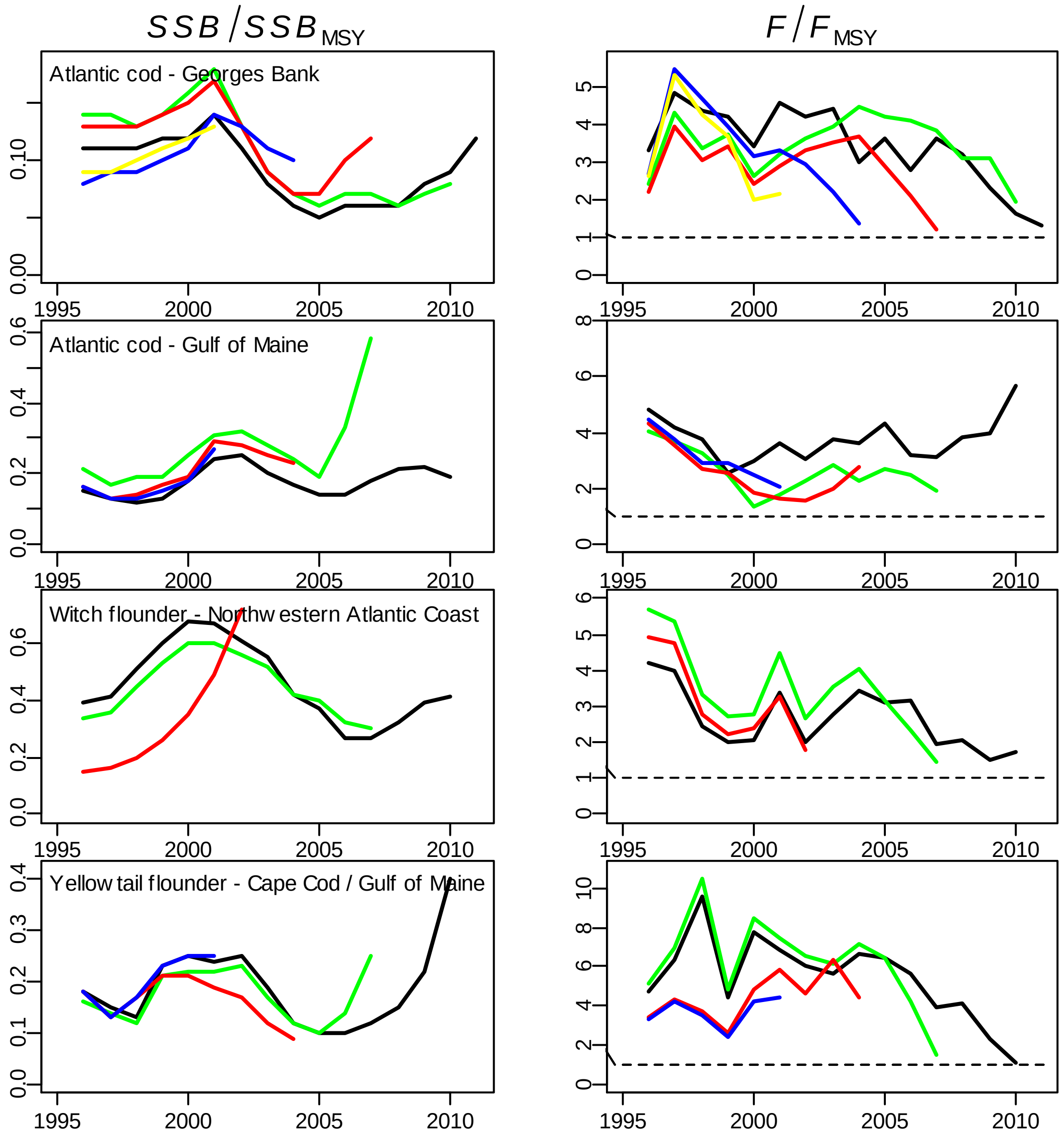


FIGURE 3.16: Time series of estimates of spawning biomass and fishing mortality, relative to the respective MSY reference points, produced by successive assessments for four stocks managed by the New England Fisheries Management Council. Each color represents a new updated assessment: black is the most recent, and green, red, blue and yellow are the preceding assessments, in that order. Time series are truncated in 1996 to focus on the most recent period.

In other cases, the failure of rebuilding plans to end overfishing has been due to difficulties to reduce overall fishing mortality when a species is caught as bycatch of a different fishery. This was, for example, the case of the red snapper stock in the Gulf of Mexico, whose juveniles are incidentally caught by shrimp trawl fisheries. The requirement to install devices in the shrimp nets to reduce discards lead to improvements but those were insufficient to end overfishing[10] (Cowen *et al.*, 2009). Subsequent rebuilding measures

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[10] http://sero.nmfs.noaa.gov/sustainable_fisheries/gulf_fisheries/red_snapper/overview/rebuilding/index.html

adopted in 2007 included a shrimp trawl fishing effort threshold to reduce bycatch, in addition to further reductions in commercial and recreational catch limits. Although the 2009 assessment (the most recent available when the database was compiled for this review) still estimated fishing mortalities in excess of the F_{MSY} proxy, estimated recent landings have led to a conclusion that overfishing has ended for this stock[11].

Changes in stock size

The outlook in terms of biomass status relative to B_{MSY} has also improved following overfished designation, but responses are smaller than for fishing mortality (Figure 3.11). While 35 stocks were below MSST one year before being designated as overfished, 29 were overfished after three years, and 22 in the most recent assessment (Table 3.4). Conversely, 10 stocks were above B_{MSY} the year before stocks were declared overfished, a number that has increased to 21 at present (Figure 3.11). Of the original 35 stocks that were below MSST, 10 rebuilt, 5 are rebuilding and 20 are still overfished (Figure 3.17). The delay in rebuilding reflects both the time to reduce F below F_{MSY} (Figure 3.12), and the time for biomass to rebuild once F is reduced. Stocks that rebuilt did so over periods that ranged from 5 to 13 years (Figure 3.17).

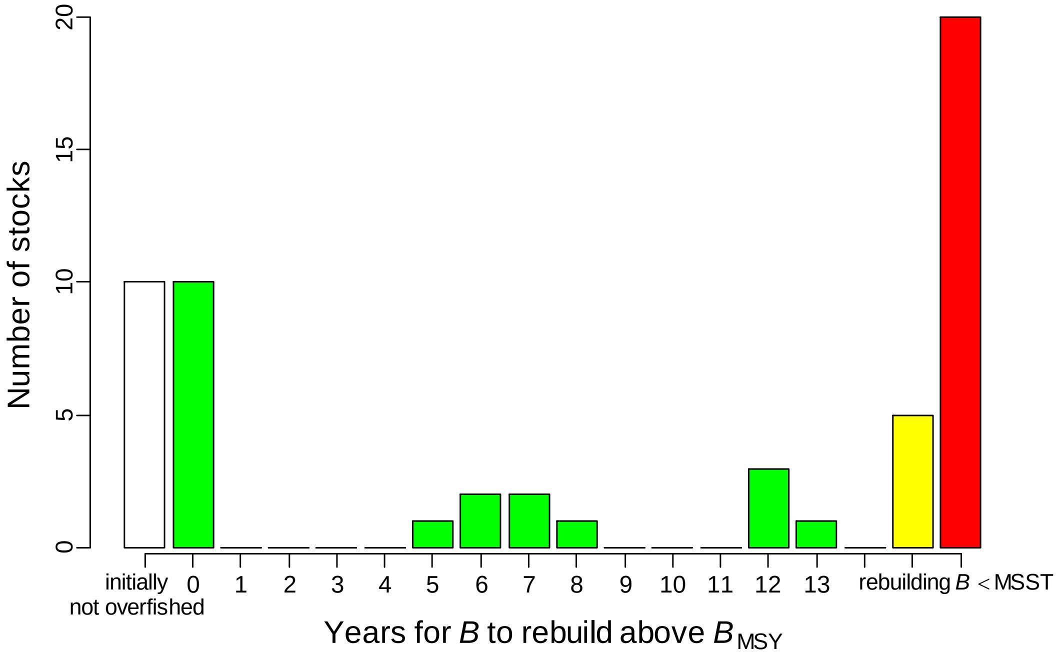

FIGURE 3.17: Number of years since each stock was declared to be overfished for biomass to rebuild above B_{MSY}. Results are based on the 55 stocks selected for detailed evaluation. The red bar corresponds to stocks that were still overfished in the last year covered by the assessment, and the yellow bar to stocks that exceeded MSST, but were still below B_{MSY}. The white and first green bars correspond, respectively, to stocks that according to current assessments were respectively above MSST and B_{MSY} in the year before being designated as overfished.

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[11] http://www.nmfs.noaa.gov/sfa/statusoffisheries/2012/first/Q1%202012%20FSSI%20Summary%20Changes.pdf

The difference in biomass between the year of the last assessment and the year the stock was declared overfished, scaled relative to B_{MSY} can be used to quantify the actual change in biomass following overfished designation. This metric, which looks at the direction of change of B, not whether it is approaching or has exceeded B_{MSY}, shows that 76% of the stocks (42 of 55) have increased in biomass since overfished designation (Figure 3.18). However, in a large fraction (58%) of the stocks that were below $B_{\mathrm{MSY},}$ the increase has not been sufficient to achieve rebuilding within the time period over which stock responses have been evaluated (Figure 3.18a, dots below the solid line).

A plot of the change in B/B_{MSY} as a function of the average F/F_{MSY} since the year of overfished designation (Figure 3.18b) shows that most stocks increased in biomass when fishing mortality was less or equal to F_{MSY}. Some stocks still showed appreciably rebuilding when fishing mortality was above F_{MSY} but less than 1.6 F_{MSY}. Conversely, none of the stocks that had fishing mortalities in excess of twice F_{MSY} achieved any marked increase in biomass. One stock that stands out as an exception is the Southern New England/Mid-Atlantic stock of yellowtail flounder, which increased in biomass with a mean F of 2.35 F_{MSY}. We note, however, that the upturn in biomass for this stock occurred after F had decreased to less than 2 $F_{\mathrm{MSY},}$ staying below F_{MSY} in recent years (C.63, Appendix C). In addition, the recent increase in biomass for this stock, expressed in units of B_{MSY}, appeared amplified by the use at the last assessment of a much lower value for B_{MSY} (corresponding to recent low recruitment levels[12]).

A low fishing mortality, on the other hand, has not been sufficient to achieve rebuilding to B_{MSY} for all stocks, in the time frame over which population responses have been assessed. Some stocks have not shown any significant rebuilding despite fishing mortality being controlled (e.g., Atlantic halibut) and there are two stocks that were below B_{MSY} when declared overfished, and declined appreciably even with $F < F_{\mathrm{MSY}}$: ocean pout and blue king crab (Pribilof Islands). Lack of recovery in the case of blue king crab has been explained by regime shifts (see further discussion in Chapter 5). Atlantic halibut had been overfished for about a century and was severely depleted ($< 0.1\ B_{\mathrm{MSY}}$) when fishing mortality was reduced (Figure C.4, Appendix C). These conditions have been associated with slow recovery rates and highly uncertain projections (Neubauer *et al.*, 2013). The other three stocks that decreased in biomass with average $F < F_{\mathrm{MSY}}$ were Gulf of Maine haddock, eastern Bering Sea snow crab, and Pacific hake, all of which were above B_{MSY} when declared overfished, and are considered rebuilt. Pacific hake biomass is largely driven by highly variable recruitment and increasing fishing pressure (Figure C.33, Appendix C). Similarly, Gulf of Maine haddock had a very strong recruitment pulse in 1999 and, according to the last assessment, had already rebuilt when it was declared overfished; the decrease in biomass followed the passing of this strong year class, coupled with recent increases in fishing mortality (Figure C.29, Appendix C).

The changes in fishing mortality and biomass (based on the most recent assessments) over the entire period since being declared overfished are summarized in Figure 3.19 by region.

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[12] Southern New England and Mid Atlantic Yellowtail flounder assessment summary for 2012. 54th SAW Assessment Summary Report. http://www.nefsc.noaa.gov/publications/crd/crd1218/partb.pdf

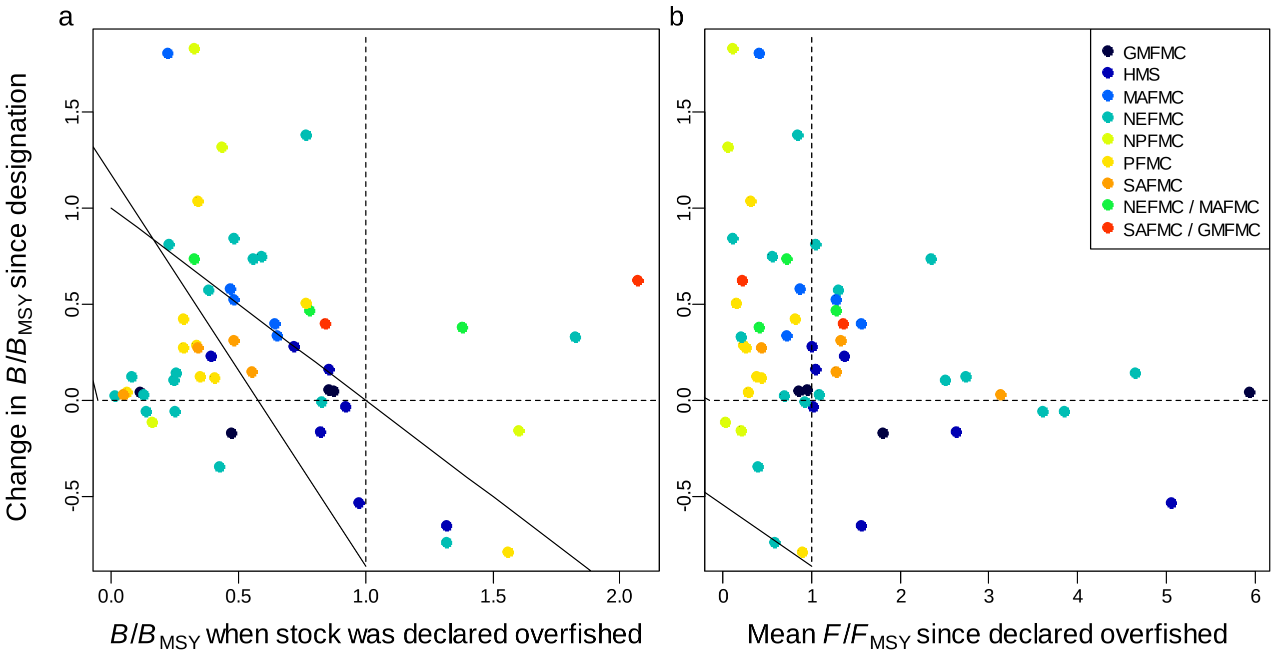

FIGURE 3.18: Change in relative biomass B/B_{MSY} since overfished designation versus B/B_{MSY} when the stock was declared overfished (left panel), and as a function of average F/F_{MSY} since overfished designation (right panel). The change in biomass was calculated as the difference in biomass between the year of the last assessment and the year the stock was declared overfished, scaled relative to B_{MSY}. The solid line in the left panel indicates the change that would have been required to achieve rebuilding (when B/B_{MSY} was < 1). Each dot corresponds to a stock, colored by region.

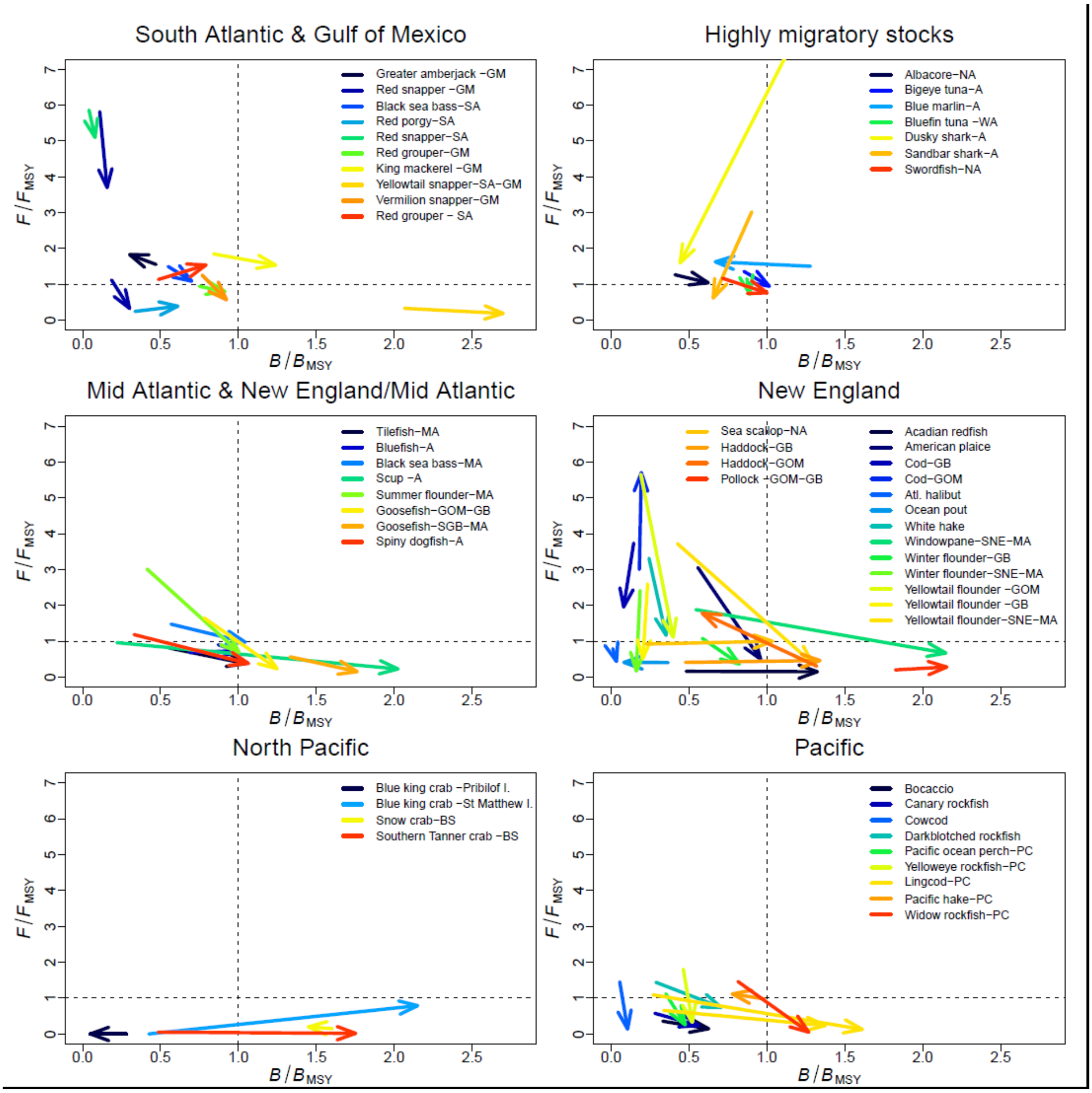

FIGURE 3.19: Change in status (F/F_{MSY} and B/B_{MSY}) from being declared overfished to the present. Results are shown for the subset of 55 stocks included in the evaluation, by region.

Although there is clearly considerable (regional) variation in trends, the general direction of change is for reduced fishing mortality (evidenced by arrows that indicate movement from the upper left quadrant to the lower left) and increased biomass (indicated by arrows from the lower left to the lower right quadrant). In addition, the majority of cases in which rebuilding or at least an increase in biomass did occur (arrows indicating displacement to the right), that increase occurred when fishing mortality was below F_{MSY}. Overall, most of the 55 stocks conform to classical fisheries theory in that biomass goes up when fishing mortality is reduced.

Conclusions

Most U.S. fisheries have been managed using simple constant exploitation rate strategies, whereby the Overfishing Limit (OFL) is determined by applying a fixed target fishing mortality rate (F) to predictions of exploitable biomass for the following years and the Acceptable Biological Catch (ABC) is then computed by reducing the OFL to account for scientific uncertainty. Historically, few fisheries have sustained constant exploitation rates over time, partly because F increases when fisheries are developing. Even after F_{MSY} has been reached, F often continues to increase, as illustrated in Figure 3.1a. In fact, the majority of the stocks which were declared overfished and for which quantitative assessments are available (see Appendix C) exhibit this inverse relationship between F and B as biomass declines below B_{MSY}, until the stock reaches the point where it is declared overfished. Fishing mortality may continue to increase beyond F_{MSY} if exploitable biomass is consistently overestimated (e.g., Figure 3.16) or if there is a time lag between decreasing biomass and implementation of reductions in catch limits. In theory, constant exploitation rate strategies and harvest control rules that decrease F when biomass drops below some threshold could achieve desired rates of rebuilding, but the historical trends observed for the majority of these stocks did not follow the trajectories expected under such management strategies.

The designation of a stock as overfished and the subsequent required implementation of a Rebuilding Plan have, in most cases, resulted in a change away from this pattern of increasing F with decreasing B and have prompted a reduction in F. The evaluation of F and B trends conducted based on a subset of 55 stocks assessed by quantitative methods indicates that F was reduced below F_{MSY} in 23 of the 36 stocks that were being subject to overfishing at the time of overfished designation. Stocks now estimated to have been below their MSST when declared overfished (35 of the 55 stocks) have generally increased in biomass when F was reduced or kept below F_{MSY} (21 stocks) and may be either rebuilding (5 stocks, i.e. 14%) or have already rebuilt (10 stocks, i.e. 29%). Of the 20 stocks estimated to still be overfished, 11 stocks (31%) had fishing mortalities well below F_{MSY} in the last year included in the assessment and many of them are showing significant rebuilding progress, while 9 stocks (26%) continued to be subject to overfishing. Overall, stocks that rebuilt or whose biomass increased appreciably were, in almost all cases, experiencing fishing mortalities below F_{MSY}. These general conclusions about stock responses are similar to those of other studies of rebuilding overexploited populations (e.g., Milazzo, 2012; Neubauer *et al.*, 2013; Sewell *et al.*, 2013), although the percentage of stocks that fall in each response category would of course depend on the specific collection of stocks considered and when the analysis was conducted. In addition, the fraction of U.S. stocks that rebuilt reported above (29%) is somewhat lower than in other studies because stocks now estimated to have not been overfished at the time of overfished designation were not included in the analysis.

The reasons for a stock not rebuilding in a timely manner are varied. First, target exploitation rates are selected so that there is at least a 50% probability of achieving rebuilding within the specified time period. Under such a criterion, even if everything went according to plan, only half the stocks would be expected to recover within the selected time horizon. Second, F for some stocks has continued to be high, still exceeding F_{MSY}, in spite of the implementation of Rebuilding Plans with target Fs equal to 75% F_{MSY} or less. The failure of Rebuilding Plans to achieve adequate reductions in F reflects implementation problems due to ineffective input controls and lack of accountability measures (e.g., as in New England before 2004), difficulties to lower fishing mortality of species caught as bycatch of other fisheries, or errors in the estimates of stock size leading to the setting of ACLs that are too high. Nature is variable and changing, leading to high uncertainty in the stock assessments and population forecasts used to set ACLs even in situations in which there are abundant data; furthermore, assessments can be consistently biased as in the case of retrospective patterns. Third, there may be delays or difficulties in reducing target exploitation rates due to litigation and delays in adopting Rebuilding Plans.

The requirement for a Rebuilding Plan is triggered by the designation of a stock as overfished. While this legal requirement was effective in forcing corrective management measures and achieving needed reductions in F in the majority of the cases, the reliance on the determination of stock status relative to a specific overfishing stock threshold (as opposed to a gradual reduction in F when the stock becomes depleted) is problematic. The review of changes in stock status associated with assessment updates presented in this chapter indicates that this determination of stock status has a relatively high probability of being wrong, given the uncertainties inherent in specifying a threshold for action and in determining whether the stock has dropped below that threshold. This dependence on an uncertain classification of stock status creates disjointedness in the management response mechanism needed to maintain fisheries at sustainable levels, and amplifies the instabilities caused by changes in stock status associated with successive assessment updates. Management strategies that incorporate a smoother response to changes in stock biomass, as used in other jurisdictions (discussed below), are likely to be more robust to errors in determination of stock status and re-evaluation of reference points.

INTERNATIONAL PERSPECTIVES

Introduction

Fisheries management, including efforts to rebuild overfished fish stocks, is arguably more advanced in the U.S. than in other countries or international organizations. The SFA mandated rebuilding of overfished stocks to B_{MSY} six years before most nations committed to a comparable objective at the 2002 World Summit on Sustainable Development (United Nations, 2002). The latter committed to "… Maintain or restore stocks to levels that can produce the maximum sustainable yield with the aim of achieving these goals for depleted stocks on an urgent basis and where possible not later than 2015…" (United Nations, 2002).

Since 2002, several countries and jurisdictions have established or strengthened policies to prevent overfishing and rebuild overfished or depleted stocks. For example, the European Commission (EC) reformed its Common Fishery Policy in 2002 with the overarching objective of ensuring exploitation of living marine resources that provides sustainable economic, environmental and social conditions (Article 2 of European Union, 2002). Australia established its Commonwealth Harvest Strategy Policy in 2007 (DAFF, 2007) and New Zealand finalized its Harvest Strategy Standard the following year (New Zealand Ministry of Fisheries, 2008). Canada similarly implemented the 2009 Sustainable Fisheries Framework (Fisheries and Oceans Canada, 2009). In 2012, the Organization for Economic Co-operation and Development (OECD) published a report, which profiled seven member nations, including the U.S., Australia, Canada, Korea, New Zealand, Norway, and Turkey. These profiles demonstrate the range of approaches that have been implemented throughout the world.

The fishery management frameworks in Australia, New Zealand, and Canada have important similarities to the U.S. approach based on the MSFCMA and NS1G. They all aim to maintain or rebuild stocks to B_{MSY} or larger. The frameworks limit fishing mortality to F_{MSY} and require a reduction in fishing mortality as stock size decreases below B_{MSY}, although they differ in how much below B_{MSY} the stock must be to trigger a reduction, and by how much fishing mortality is reduced. However, the frameworks in these three countries either have no time limit on rebuilding depleted or overfished stocks, or they have greater flexibility than in the U.S.

New Zealand

Management of fish stocks in New Zealand is based on the 1996 Fisheries Management Act, which requires that the Total Allowable Catch (TAC) be set to maintain the stock at or above a level that can produce the MSY or to enable the size of any stock which is presently below that level to be increased. The sizes of stocks that are below B_{MSY} should be increased "within a period appropriate to the stock, having regard to the biological characteristics of the stock and any environmental conditions affecting the stock." As such, the 1996 Fisheries Management Act does not impose hard limits on the year by when depleted stocks are to be rebuilt. However, as noted below, such limits are included in the more recently adopted Harvest Strategy Standard.

New Zealand's Harvest Strategy Standard (New Zealand Ministry of Fisheries, 2008; 2011a) defines the target level for a New Zealand fish stock as "the desired biomass level or fishing mortality rate, or catch or proxies for each of these." This Standard requires the definition of (a) a target level about which a fishery or stock should fluctuate, (b) a soft limit that triggers a requirement for a formal, time-constrained rebuilding plan, and (c) a hard limit below which fisheries should be considered for closure. Stocks which have fallen below the soft limit "should be rebuilt back to at least the target level in a time frame between T_{MIN} and $2 * T_{MIN}$ with an acceptable probability", where T_{MIN} is defined as in the U.S. The New Zealand Harvest Strategy Standard contains guidelines for the information that should be provided for stocks between the soft limit and the target (New Zealand Ministry of Fisheries, 2011a), but there is no pre-specified rebuilding time frame for them. The default soft limit is 50% of B_{MSY} or $0.2B_0$, whichever is higher, and the default hard limit is 25% of B_{MSY} or $0.1B_0$, whichever is higher.

Australia

The Commonwealth Fisheries Harvest Strategy Policy adopted by the Australian government (DAFF, 2007) provides a framework for the management of fisheries which includes the definition of explicit default target and limit reference points and harvest control rules. Reference points used as targets are those related to maximum economic yield (MEY). The biomass associated with MEY, B_{MEY}, is usually set at the default proxy of $1.2B_{MSY}$ and the default proxy for B_{MSY} is 0.40 of the unfished level. Harvest control rules call for a progressive reduction of fishing mortality from F_{MEY} to zero as biomass drops from B_{MSY} to $B_{LIM} \geq 1/2\ B_{MSY}$. A risk criterion has been established requiring that there is less than a 10% chance of the stock falling below B_{LIM} per generation time under application of the harvest strategy. A rebuilding strategy, possibly including additional conservation measures, has to be developed for stocks that fall below B_{LIM}. However, rebuilding management responses may be part of the "normal" harvest strategy, which specifies measures as a function of estimates of stock size. As far as setting a rebuilding time frame, the Australian legislation recognises "that there are likely to be a number of alternative time paths to rebuild a stock", and that "the optimal time path to rebuild a stock has an economic component." Still, "typical recovery times are defined as the minimum of: (1) the mean generation time plus 10 years, or (2) three times the mean generation time (where the mean generation time is defined as the average age of a reproductively mature animal in an unexploited population)."

A variety of harvesting strategies that do not require the use of model-based *F* or *B* reference points have been developed for data-poor or low-value stocks. These may involve the use of empirical reference points and indicators (e.g., catch rates, age/size composition), effort controls and area closures. Additional precaution may be implemented in order to achieve performance that is consistent with that of the model-based harvest rules.

The adoption of the Harvest Strategy Policy at the national level has been accompanied by increased use of spatial management, with 38% of Australia's exclusive economic zone currently within Marine Parks 13, a close to 30% reduction in fishing capacity through a government-funded buy-back program (Vieira *et al.,* 2010), and significant investment to deter foreign illegal fishing[14] (Sainsbury, personal communication).

Canada

Canada's fisheries are primarily governed by the Fisheries Act. However, the Fisheries Act itself does not include specifics on rebuilding. Instead, the Act authorizes Fisheries and Oceans Canada to manage rebuilding through Integrated Fisheries Management Plans. The United Nations *Agreement on Straddling and Highly Migratory Fish Stocks* (UNFA), which went into effect in 2001, commits Canada to use the "precautionary approach"- in managing both its migratory and domestic stocks. As a result, in 2006 Canada developed a fishery decision-making framework incorporating the Precautionary Approach[15], which classifies stock biomass as critical, cautious, or healthy. The amount harvested must be progressively reduced once a fish stock has fallen out of the healthy category and into the cautious zone. The reference point that separates healthy from cautious is known as the "Upper Stock Reference" or USR. The USR may be the target biomass, but targets can also be set higher.

European Union

Fisheries management in the European Union (EU) is comparable to that in Canada, New Zealand, Australia, and the U.S. in many respects, but its fisheries are not performing as well in terms of stock status. Fisheries of the EU are managed by the European Commission according to the Common Fisheries Policy (CFP), most recently reformed in 2002. The

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[13] Details of Australian system of marine reserves provided in http://www.environment.gov.au/marinereserves/overview.html

[14] http://www.aic.gov.au/publications/current%20series/rpp/100-120/rpp109/08.html

[15] http://www.dfo-mpo.gc.ca/fm-gp/peches-fisheries/fish-ren-peche/sff-cpd/precaution-eng.htm

Policy calls for application of the precautionary approach to protect and conserve living aquatic resources, provide sustainable exploitation and minimize impacts of fishing on marine ecosystems. It also calls for progressive implementation of an ecosystem-based approach to fisheries management. However, the lack of operational guidance for fisheries management and a clear priority for preventing overfishing is a serious problem with the Policy (Sissenwine and Symes, 2007). The European Commission (EC) also concluded that "No priority is set for these objectives and, while direct references are made to adopting a precautionary and an ecosystem approach, it is not clear how this relates to economic and social conditions. There are no clear indicators and yardsticks that could provide more concrete guidance or to help measure policy achievements" (European Commission, 2009). The EC review reported that 88% of European stocks were being fished beyond MSY, but the situation has improved since the report was issued.

In 2006, the EC adopted a policy for "Implementing sustainability in EU fisheries through maximum sustainable yield" to fulfill its WSSD commitment to maintain or restore stocks to levels that can produce MSY by 2015 (European Union, 2006). According to the policy, the EC interprets its commitment to restore stocks to the MSY level by 2015 as meaning reducing fishing mortality to or below the F_{MSY} level. Unlike the U.S. and many other countries, the EC policy does "… not seek to manage biomass levels" (European Union, 2006, Section 3.3). It argues that biomass targets are highly uncertain because of environmental variability. ICES began giving catch advice according to an MSY approach in 2010 (ICES, 2010a). The ICES MSY approach does use a biomass reference point as a trigger for reducing fishing mortality, but it does not require an estimate of B_{MSY} because such estimates are considered to be dynamic and uncertain, particularly for fisheries that have been overfished for so long that B_{MSY} is an extrapolation outside the range of stock observations.

In 2009 the EC launched a public process to discuss the way the EU managed their fisheries. The debate focused on a suite of issues that included fleet overcapacity, inadequate policy guidance for decision making and implementation, political will around issues of enforcement, and the roles and responsibilities of the fishing industry. This process resulted in a new CFP agreement to be implemented in early 2014, which involves a commitment to ending overfishing by 2015 where possible, and at the latest by 2020, for all fish stocks.

Comparison of stock status statistics

Though data are difficult to compare, recent assessments indicate that stock status in the U.S. is similar to that found in Canada, New Zealand and Australia. In 2012 the U.S. reported that 20% of assessed stocks were overfished while 13% were undergoing overfishing (Table 3.1). For stocks with known status in 2011, Canada reported that 12% were harvested above approved levels and, in terms of biomass, 60% were healthy, 26% required caution (i.e., a progressive reduction in removal rate is required) and 14% were in the critical zone (i.e., below a limit reference point where removals are kept to the lowest possible level)[16]. For New Zealand, the percentage of depleted stocks (comparable to "overfished" in the U.S.) decreased slightly from 19% to 17% from 2009 to 2012, and the percentage of stocks subject to overfishing decreased from 25% to 18% during this period

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[16] Data provided by Environment Canada, available at http://www.ec.gc.ca/indicateurs-indicators/default.asp?lang=en&n=1BCD421B-1

(Mace, 2012, personal communication). Finally, of these four countries, Australia reported the lowest percentages of stocks classified as overfished (12%) and undergoing overfishing (6%), for stocks whose status could be determined as of 2011 (Woodhams *et al.*, 2012).

The proportion of European stocks subject to overfishing is much larger than in the U.S., Canada, Australia or New Zealand, although it has dropped substantially over time (Table 3.5). However, 80% of stocks in the Mediterranean and Black Seas are reported as still undergoing overfishing in 2012 (European Commission, 2012).

The United Nations Food and Agriculture Organization (FAO) reported that about 30% of the world's fish stocks of known status are overexploited (which corresponds to overfishing in the U.S.) (FAO, 2012). Thus, the U.S., Canada, New Zealand and Australia are performing somewhat better than Europe and than the worldwide average.

TABLE 3.5: Percent of European stocks with overfishing relative to F_{MSY}. SOURCE: European Union, 2012.

2005	2006	2007	2008	2009	2010	2011	2012
94%	91%	94%	88%	86%	72%	63%	47%

FINDINGS

3.1: Over the period 1997-2011, 85 stocks or stock complexes were at some point declared to be overfished or approaching an overfished state, and were therefore subject to the requirement to be placed under a Rebuilding Plan. Rebuilding Plans were implemented for 79 of those stocks.

3.2: Analysis of the annual reports to Congress indicates that 42 of these 85 stocks are no longer classified as overfished: 11 are rebuilding and 31 were subsequently designated as rebuilt, one of which is currently considered of undefined status. Four additional stocks that were declared rebuilt became overfished again and one is approaching overfishing.

3.3: The 10-year rule determined the target year for rebuilding for 31 of the 70 stocks for which Rebuilding Plans with a defined rebuilding time frame have been implemented.

3.4: Target fishing mortalities have generally been lower than 75% F_{MSY}. *In some regions, target fishing mortalities are substantially lower than this, and rebuilding time frames chosen in those regions are much shorter than the maximum specified by the National Standard 1 Guidelines. More extreme reductions in target fishing mortalities have been effected in situations in which rebuilding progress was slower than anticipated when the Rebuilding Plan was adopted and the target year for rebuilding was approaching.*

3.5: Due to the uncertainty in stock assessments, the perceived status of fish stocks relative to overfished status in any particular year can change substantially as more data become available and assessment methods are changed over time. According to the most recent assessments available, there is a substantial probability of:

- *Classifying stocks as overfished and requiring rebuilding plans when later assessments indicate that the stocks were not below the minimum stock size threshold (MSST).*

- *Classifying stocks as rebuilt when the updated assessments indicate that the stock was not overfished.*

The probability of classifying stocks that are overfished as healthy cannot be quantified from the data available, but may also be high.

3.6: Estimated trends for 55 stocks declared overfished and assessed by quantitative methods indicate that fishing mortality has generally been reduced, and stock biomass has generally increased, for stocks that were placed under a rebuilding plan. Of the 35 stocks now estimated to have been below their MSST *when declared overfished, 10 have rebuilt, 5 are rebuilding* ($MSST < B < B_{\mathrm{MSY}}$) *and 6 more have increased in size. Of the remaining 20 stocks estimated to still be overfished, 11 had fishing mortalities well below* F_{MSY} *in the last year included in the assessment and are therefore expected to rebuild if low fishing mortalities are sustained. Stocks that rebuilt or whose biomass increased appreciably were, in almost all cases, experiencing fishing mortalities below* F_{MSY}.

3.7: Some stocks continue to be subject to overfishing despite fishing targets being set at or below F_{MSY} *with the intent of rebuilding within the maximum time frame. Retrospective patterns in the assessments leading to overestimation of stock size have contributed to this continuing overfishing.*

3.8: Although rebuilding plans have target years for recovery to B_{MSY}*, the rate at which stocks rebuild is probabilistic such that some stocks will rebuild before the target year while others will rebuild after the target year or not rebuild until environmental conditions improve, even if the rebuilding plan is implemented as intended, fishing mortalities are close to the targets, and targets are based on robust stock assessments.*

3.9: Over half of the nation's 478 stocks and stock complexes identified in the stock status reports to Congress are either unassessed or unknown with regards to their status as overfished or overfishing.

3.10: Countries that have legal or policy mandates for ending overfishing and rebuilding stocks appear to have better stock status on average than other countries.

4

TECHNICAL CONSIDERATIONS IN DEVELOPING REBUILDING PLANS

INTRODUCTION

The MSFCMA requires that fisheries be managed to achieve optimum yield (OY), which, under the current specification of the law, is considered to be the maximum sustainable yield (MSY) as reduced by ecological, economic and social factors[1] (Chapter 2). Consequently, the concept of MSY is key to the implementation of the MSFCMA, as it is for fisheries management in general worldwide (United Nations, 2002). According to theory, a population (quantified in terms of biomass, measured as the number of individuals multiplied by their average weight) can produce an annual surplus in biomass that can be harvested sustainably. Over the range of population biomasses, there exists a level at which the average sustainable surplus and associated harvest is maximized. Broadly, if a population is allowed to grow until it reaches its environmental carrying capacity, surplus production will cease and the population will grow no further. Production may also be negligible if the population is driven to critically low levels, at which it would be considered collapsed. Theoretically, between these maximum and minimum population levels, a surplus in production can be harvested on a regular basis while the population is sustainably maintained. The maximum surplus production thus determines the MSY, which occurs at some intermediate population size (B_{MSY}). The fishing mortality corresponding to this MSY, F_{MSY}, is the rate of population removal by the fishery that will maintain the average population size at B_{MSY}. The function that relates surplus production to fishing mortality (F) and population biomass (B) is referred to as the production function. It has been recognized for some time that trying to harvest at MSY on an ongoing basis is fraught with risk given the variability and uncertainties that exist in nature, science, and management (Larkin, 1977). Nevertheless, the concept can be a useful one in establishing population sizes and harvest rates that a population can sustain and be productive, given its life-history characteristics and dynamics.

ACCOUNTING FOR UNCERTAINTY IN FISHERIES REBUILDING

The ability to implement MSY-based fisheries management depends upon the characteristics of the stocks and the fisheries, the quality and quantity of data available

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[1] 16 U.S.C. § 1801(b)(4); 16 U.S.C. § 1802(33).

for both, the production model chosen, and the frequency at which assessments occur so that management can act adaptively. MSY concepts were designed for, and work best for, data- and knowledge-rich fisheries, which are generally those stocks that are of greatest economic value, productivity, and volume. The high valued Alaskan fisheries, which constitute over half of the annual landings in the U.S. by weight (NMFS, 2012), exemplify a number of success stories in this context (Chapter 3). In contrast, most stocks in the Caribbean and Western Pacific are considered “data-poor” and do not have enough information to estimate MSY or apply MSY-based control rules.

MSY, B_{MSY} and F_{MSY} can be influenced by a number of factors:

- *MSY depends on fishing practices.* F_{MSY} is usually treated as single value under the assumption that the relative fishing mortalities-at-age are constant (that is time-invariant fishing selectivity is assumed). Depending on the fishing gear or combination of gears used, it might be possible to increase the long-term sustainable yield by reducing the proportion of small fish captured (e.g., with larger trawl mesh size). The fact that F_{MSY} depends on selectivity can lead to counter intuitive outcomes when management measures are applied through adjustments in gear selectivity. For example, if mesh size is changed to protect smaller fish and increase potential long-term yield, the B_{MSY} associated with the resulting new selectivity pattern could also be larger. This could lead to an overfished condition without overfishing having occurred, if current biomass is less than the new, larger B_{MSY}.
- *MSY depends on environmental conditions.* MSY is derived from a production function, which depends on biological processes, including reproduction, growth and natural mortality. These processes are all affected by environmental conditions, which vary in space and time, such that MSY also changes. B_{MSY} is therefore usually interpreted as the long-term average biomass level associated with F_{MSY}. This approach is probably satisfactory when there are interannual or relatively short-term fluctuations in environmental conditions. However, a production function or an estimate of B_{MSY} based on a past average may not be a useful representation of the future in cases where gradual environmental changes occur over longer time periods (e.g., decadal) or when permanent changes in average conditions occur, such as phase shifts (see Chapter 5).
- *MSY depends on ecological interactions.* Biological processes for one species (e.g., growth and natural mortality) are affected by changes in the abundance of interacting species, which may also be subject to management. Competition for food and predator-prey interactions makes it unlikely that all species can be maintained simultaneously at the B_{MSY} values calculated for each species individually. Just as the production function for a single species depends on fishing practices that influence the size composition of fish in the population, multispecies production depends on fishing practices that influence the species composition of the community. Multispecies production functions have been calculated for communities of fish (e.g., Brown *et al*., 1976), and were considered in the setting of a multispecies total allowable catch for the International Commission for the North Atlantic Fisheries, ICNAF (Halliday and Pinhorn,

1985; O'Boyle, 1985). The impact of ecological interactions on rebuilding is discussed further in Chapter 5.

- *MSY depends on technical interactions.* When a fishery aimed at one species generates fishing mortality on other species, which occurs both in mixed stock fisheries and in the inevitable situations of bycatch, it is referred to as "technical interaction." It is generally not possible to apply desired fishing mortality rates to all species simultaneously when there are technical interactions, so the maximum total average catch that can be taken from a community of interacting species is lower than if there were no technical interactions. The target fishing mortality for some stocks may need to be reduced below (perhaps substantially below) F_{MSY} if the mix of stocks caught includes some unproductive (or rebuilding) stocks, given the requirement to avoid overfishing of any stocks. The practice of reducing the target fishing mortality for productive stocks to avoid overfishing unproductive stocks is common, and is referred to as "weak stock management". For example, catches of yellowtail rockfish off the U.S. were reduced substantially given the need to reduce harvest of widow rockfish when the latter was declared overfished. However, weak stock management, and technical interactions in general, can lead to marked economic and social impacts, as discussed in Chapter 6.

Finally, the MSY concept relates to biological yield (number, weight, or volume of fish). It does not take account of the value of the fish, the cost of catching the fish, the distribution of benefits from fishing, or the socio-economic impacts of fisheries management measures. Economic analogs of MSY exist (e.g., Maximum Economic Yield, as defined in Chapter 2), but these are not currently used for status determination in the U. S. (See Chapter 6 for details on the socio-economic dimensions of rebuilding.)

Setting reference points and targets for rebuilding

As outlined in Chapter 2, the National Standard 1 Guidelines (NS1G) specify that overfishing is occurring when fishing mortality is above the Fishing Mortality Maximum Threshold, which is equal or less than F_{MSY}. A stock is considered overfished when stock biomass drops below the Minimum Stock Size Threshold (MSST)[2]. MSST is commonly set at ½ B_{MSY}, but guidance has been provided indicating that other values might be selected. Restrepo *et al.* (1998), for example, advocated MSST= $(1-M)B_{\mathrm{MSY}}$, where M is the instantaneous rate of natural mortality. The current NS1G say that MSST should be a biomass level from which a stock will recover to B_{MSY} in 10 years if $F=F_{\mathrm{MSY}}$. Measures must be put in place to reduce mortality when overfishing (i.e., $F > F_{\mathrm{MSY}}$) is estimated to occur. Once the overfished threshold is impinged upon (i.e., $B <$ MSST), a fishing mortality that would allow rebuilding, F_{REBUILD}, is usually determined with the goal of fishing at low enough levels to allow biomass to grow to B_{MSY} with 50% probability or greater within a specified time period. Ultimately the target fishing mortality used to calculate the Accepted Biological Catches (ABC) (F_{ACL}) may be lower than this, as is the case in New England where F_{ACL} is set to F_{REBUILD} or 75% F_{MSY} whichever is less (see

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[2] 50 C.F.R. 600.310 (e)(iv)(2)(B)-(E) (2009).

Chapter 3). There exists considerable variability in how these reference points are defined and how population biomass is monitored relative to the B_{MSY} reference point.

Three general approaches have been used to derive biological reference points using population dynamics models (Sissenwine and Shepherd, 1987). These are the classical production modeling approach (Graham, 1935; Schaefer, 1954; 1957; Pella and Tomlinson, 1969), the yield-per-recruit approach (Thompson and Bell, 1934; Beverton and Holt, 1957), and the spawner-recruit approach (Ricker, 1954; Beverton and Holt, 1957). The classical production model follows changes in population biomass relative to changes in catch to estimate the intrinsic rate of population growth and the carrying capacity, which can be used to determine F_{MSY} and B_{MSY}. The yield-per-recruit model follows the changes in the biomass of a cohort of fish through time due to mortality and individual growth to determine the fishing mortality that maximizes the lifetime yield for that cohort (F_{MAX}). The estimates of F_{MAX} and other Fs derived through yield-per-recruit analyses, such as for example $F_{0.1}$, have often been used as proxies for F_{MSY} when there is insufficient information to support one of the other two approaches. When age information is available, a spawner-recruit model might be formulated and spawning stock levels that maximize the catch (the product of yield-per-recruit and recruits) can be determined. Thus, the F and B that maximize production in this context are again interpreted as F_{MSY} and B_{MSY}. However, even in these situations, the relationship between number of recruits and measures of reproductive potential may be so uncertain that proxies (such as $F_{35\%}$ or $F_{40\%}$, defined as the fishing mortality that reduces the average spawning biomass-per-recruit to 35% or 40% of the unfished level) might be used instead (see for example the work by Clark, 1991; 2002). Consequently, there are several approaches to deriving biological reference points depending on the information available. In the many cases where information about production is limited or unavailable, reference points may still be calculated, albeit with increased uncertainty.

In practice, F_{MSY} is more robustly estimated and has a firmer foundation for implementation than B_{MSY}, especially in situations where reference points are used in a rebuilding context. The value of F_{MSY} varies directly with the slope of the stock-recruitment or stock-production relationship. Overfished stocks have data at low abundance that may allow this slope, and hence $F_{\mathrm{MSY},}$ to be estimated with greater confidence than stocks without data at low abundance. In contrast, the biomass reference points B_{MSY} and the unfished biomass, B_0, depend on the strength of density dependence, which determines the curvature of the stock-production relationship. The degree of curvature is not well determined from data at low abundance. Therefore the estimates of B_{MSY} are expected to change (and hopefully improve) as stocks rebuild. Even if B_{MSY} were not used as a target, biomass reference points are usually still used to specify harvest control rules, as discussed in the section below. Here the choice of biomass threshold may be less critical to the performance of the harvest control rule provided that F is reduced smoothly when biomass falls below the threshold.

The problem with estimating B_{MSY} from stock-recruit data is that it requires fitting a compensatory stock-recruitment function to estimate the average recruitment corresponding to B_{MSY} so that it can be multiplied by the spawning stock biomass per recruit at F_{MSY}. Even for stocks that have been reduced to low abundance, the stock-recruitment relationship may remain uncertain. This can occur when the relationship between recruits and spawning stock appears to be a random scatter of points. Here there would be no evidence that recruitment decreases with decreases in spawning stock biomass. In this situation, F_{MAX} may be the best proxy for F_{MSY}, although $F_{0.1}$ might be more precautionary. In the case where there is a scatter of points that shows at least some evidence of decreasing recruitment with decreasing spawning stock biomass, an estimate of the slope at the origin is possible. If the slope corresponds to an F that is less than F_{MAX}, one might use that estimate for F_{MSY}, otherwise F_{MAX} may still be the best proxy for F_{MSY}. In these cases of uncertain stock-recruitment relationships, it is common to turn to F proxies such as $F_{35\%}$ or $F_{40\%}$, as discussed in the paragraph above. Regardless of whether a direct F_{MSY} estimate or proxy is used, assumptions about the distribution of recruitment are required to estimate B_{MSY}. The choice of proxy is also uncertain as proxies may have been derived using generic life histories, which may be inadequate for the stock in question. There is a trade-off between choosing an uncertain reference point or a potentially biased proxy. More years of data over a greater range of stock abundance should allow these biological reference points to be estimated with more precision.

Potentially very rudimentary and *ad hoc* reference points are used in data-poor situations where even the most basic productivity information is lacking. For example, management actions may be based on changes in fishery-dependent or -independent catch per unit effort (CPUE) indices over time, relative to some reference level (as is done for the skate complex in New England and coral reef fish in the Western Pacific). Some have suggested using other metrics such as changes in average length as a proxy for mortality (Brodziak *et al.*, 2012) or defining overfishing in terms of declines in growth, recruitment, economic value, or ecosystem integrity (Russ, 1991). Although relating these metrics directly to B_{MSY} is not always possible, establishing such benchmarks for management action in data-poor situations should facilitate keeping the fishery at some sustainable level, although that level may not be optimal, or necessarily equate to the legal equivalent of B_{MSY}.

Given the wide range of information available to estimate reference points, it is not surprising that the operational definitions for overfishing and overfished vary among regions of the U.S. and even among stock categories within regions (Table 4.1). This variation can lead to inconsistencies among regions. For example, a stock of Pacific Ocean perch (*Sebastus alutus*) would be declared overfished if its biomass was less than half of the estimate of $B_{35\%}$ by the North Pacific Fishery Management Council (NPFMC), but if this biomass was below 25% of the unfished equilibrium biomass, B_0, by the Pacific Fishery Management Council (PFMC). Consequently, the nature of the fishery, the condition of the environment, the quality of the data, and even the background and experience of the scientists and managers will play a role in determining if a stock needs rebuilding, how and to what level it should be rebuilt, and the mechanisms by which progress towards rebuilding will be monitored.

The problem of accounting for scientific uncertainty in formulating management advice is more complicated than simply choosing how to specify the reference points. For many, if not most, stocks there may be multiple plausible representations of the dynamics and productivity of a stock with little scientific basis for choosing among them, even when classical estimation methods are applicable and a given choice for the reference point is clear (e.g., F_{MSY} based on a stock-recruitment relationship). For example, the fit of either a Beverton-Holt or Ricker stock-recruitment model to the data may appear to be equally plausible, but the biological reference points estimated with the two models may be dramatically different (see for example Myers *et al.*, 1994 and Barrowman and Myers 2000). Unfortunately, the choice of how the dynamics are represented may be critically important for determining the trade-offs between short-term and long-term benefits from the fishery and risks to the stock. In situations like this, scientists are more often now communicating the uncertainty to managers and policy makers outlining the implications of each plausible model under the alternative rebuilding strategies, usually in the form of some decision table (see for example, Lane and Stephenson, 1998; MacCall, 1999). The bridge between best use of science and accounting for risk in decision making will continue to develop with necessary input from all parties involved.

Harvest control rules

F_{REBUILD} is defined as the fishing mortality rate that would allow the stock to recover with 50% or greater probability (the probability is a management choice) to B_{MSY} within the allotted recovery time period (i.e., by T_{TARGET}). F_{REBUILD} is typically determined through simulations conditioned upon a population-dynamics model and according to a harvest control rule (HCR), which may call for changes in F as a function of stock status and possibly other variables. Ideally, the HCRs are established by the Regional Fisheries Management Councils (RFMCs) during the development of the Fishery Management Plan, and this Plan is implemented, hopefully, prior to any need for stock rebuilding. HCRs should specify the strategy for maintaining a sustainable fishery, and also establish what actions to take if a stock is determined to be overfished or if overfishing is determined to be taking place. The HCRs might also include rules that differ according to whether the stock is healthy, just below B_{MSY}, overfished, or in a rebuilding phase (Punt, 2003; Punt and Ralston, 2007), although it would be best if they were constructed in such a way so as to not create any discontinuities, as discussed below. Simulations that use HCRs, and are based on the assessment model used or a reasonable approximation of it, can then be used to predict what is likely to happen under different management scenarios when exploring options for the target year for rebuilding. Most modern simulation methods allow quantification of the uncertainty associated with the projections, such that the probability of the stock being above or below B_{MSY} in any future year can be calculated. However, even if the projections are relatively accurate, the population is still, by definition, only expected to reach the target biomass level *half of the time* if the plan chosen is based on a 50% rebuilding probability.

The current "10-year rule", established by the NS1G to set the maximum rebuilding time, leads to a discontinuity because a small change in information (or model assumptions) can lead to a major change in the maximum permissible time to rebuild to B_{MSY} (T_{MAX}). There are similar discontinuities in the HCRs used to determine ACLs depending on whether or not a stock is under a Rebuilding Plan. Figure 4.1 provides illustrative examples of how and when discontinuities can occur and be avoided (based loosely on control rules used by the NPFMC and the PFMC for groundfish stocks). Figure 4.1(a) shows the HCRs for stocks whose sizes are larger than MSST (taken to be $0.5B_{\text{MSY}}$ in this example) and therefore are not subject to a Rebuilding Plan. Fishing mortality is maintained at a rate that is less than F_{MSY} when biomass is larger than B_{MSY} to ensure that overfishing does not take place with some chosen probability, and declines if stock size is smaller than B_{MSY}. However, if the stock had been declared to be overfished and has not rebuilt, ACLs would be based on a HCR such as that in Figure 4.1(b). The change from the "normal FMP" HCR in Figure 4.1(a) to the rebuilding HCR in Figure 4.1(b) can lead to a marked change in allowable fishing mortality and hence catch. Paradoxically, rebuilding HCRs can result in an increase in removals compared to what would have been expected under the "normal" HCR, as can be seen by extending the dashed diagonal line in Figure 4.1(a) to intersect the lower horizontal line in Figure 4.1(b) at some stock sizes. In addition, once the stock has rebuilt to B_{MSY}, the fishing mortality will increase from the rebuilding fishing mortality ($0.2F_{\mathrm{MSY}}$ in Figure 4.1(b)) to the fishing mortality under the "normal" HCR. Thus, the target fishing mortality for a stock whose size is between MSST and B_{MSY} depends on whether the stock is governed by a normal FMP HCRs (if it has not been overfished) or by a Rebuilding Plan. Such discontinuities resulting from implementation of the NS1G exemplify the advantages of polices that adjust fishing mortalities gradually and smoothly as a function of changes in stock size regardless of whether or not the stock has been declared overfished.

In general, the ability to satisfy management goals while avoiding unnecessary disruptions in the operation of a fishery is best achieved when there are few or no discontinuities. For example, a control rule such as that in Figure 4.1(c) would permit rebuilding to B_{MSY} while avoiding discontinuous changes in fishing mortality. However, the time to rebuild to B_{MSY} would not necessarily match that expected under the existing rules for determining $\mathrm{T}_{\mathrm{MAX}}$, although the values for the parameters of the HCR could be chosen with this intent in mind. An additional advantage of having a single, continuous HCR is that it reduces the dependence on the application of status determination criteria (overfished versus not overfished), which may be highly uncertain and prone to changes with successive assessment updates, as shown in Chapter 3. The recent re-categorization of the yellowtail flounder stock from southern New England as rebuilt provides an example of one such highly uncertain decision[3]. Status determination for this stock depended on which of two recruitment scenarios proposed was accepted. One scenario involved a reduction in stock productivity since about 1990, leading to an estimate of B_{MSY}= 2,995 metric tons (mt) and to the conclusion that the stock had rebuilt. A much larger B_{MSY}= 22,615 mt, which would imply that the stock is still overfished, was estimated under the alternative recruitment scenario, which attributed the low recent recruitments to the current small size of the spawning stock and predicted higher recruitments for spawning stock biomasses larger than 4,319 metric tons. The Stock Assessment Review Committee (SARC, the external peer review body in New England and the Mid-Atlantic charged with reviewing benchmark assessments) concluded that the evidence was 60:40 in favor of the productivity change and the stock was reclassified as rebuilt. The SSC decided that ABCs and management in general should be based on a change in productivity and that in this case 30 years of low productivity was enough evidence that something differed from the long-term average of the entire time series

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[3] 54th SAW Assessment Summary Report available at http://www.nefsc.noaa.gov/publications/crd/crd1214/crd1214.pdf.

(approximately 70 years). The justification for lowering the rebuilding target in this case is very weak. As discussed in Chapter 5, this Committee found no evidence of a decrease in recruits per unit of spawning biomass for this stock.

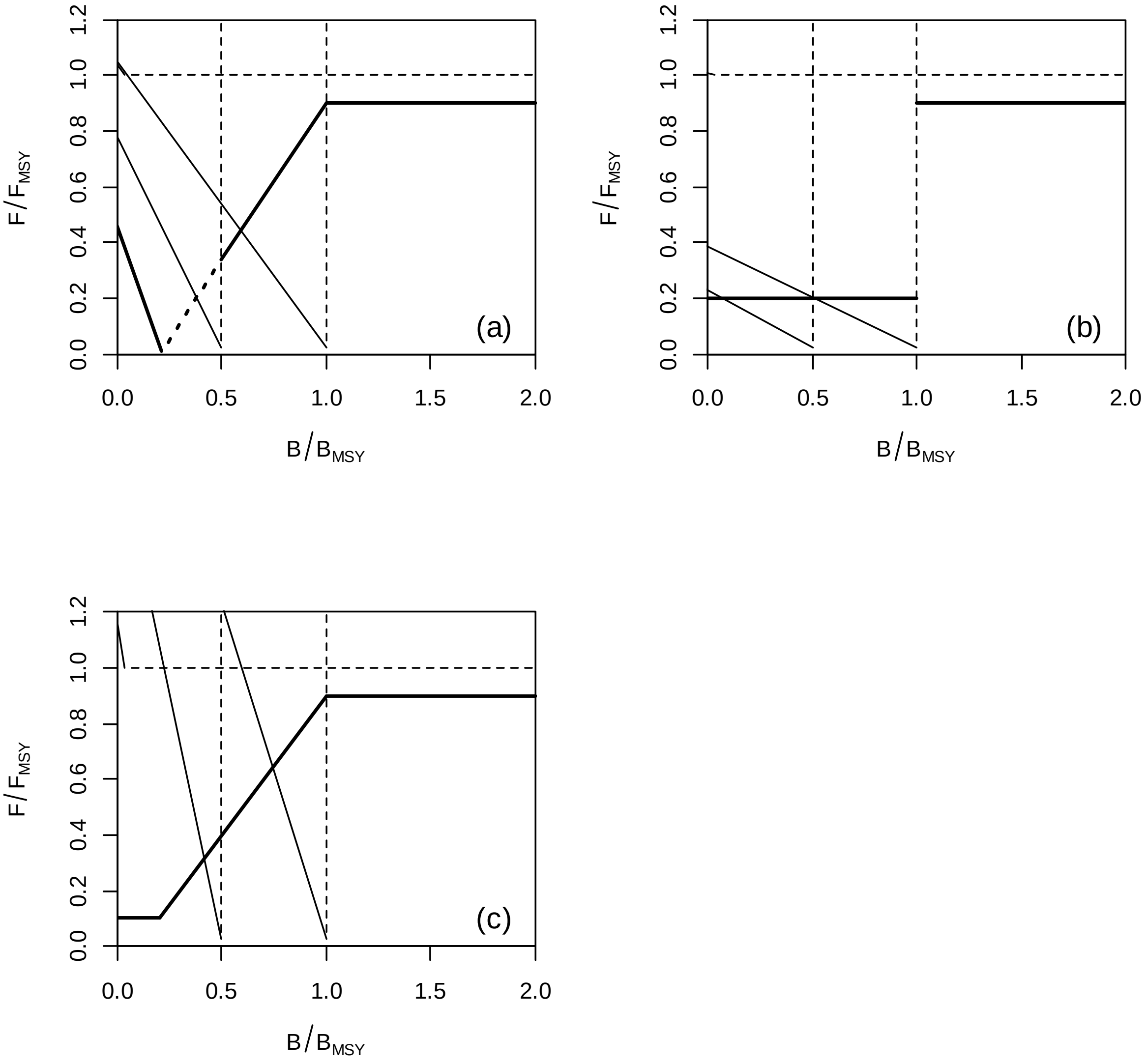

FIGURE 4.1: Illustrative harvest control rule (HCR) for a stock for which MSST = ½ B_{MSY}. (a) a HCR for stocks that have not been declared overfished, (b) a HCR for stocks that have been declared overfished, illustrating an abrupt discontinuity in allowable fishing mortality and (c) a HCR that does not have any discontinuities. Note that the diagonal dashed extension of the rule segment in (a) is not used in practice because stocks below MSST are managed under a Rebuilding Plan.

Probability of meeting rebuilding deadlines

Determining the probability of meeting rebuilding deadlines is difficult, and subject to much uncertainty given variability in the ecosystem, the dynamics of the exploited fish populations, the fishery and the data collected. Stock projections are generally based on the most recent stock assessment, which provides estimates of stock size relative to B_{MSY} (or its proxy), the age structure of the population, mean generation time, productivity and fishing selectivity parameters, all of which are imprecise. A model of the relationship between spawning-stock size and future recruitments is fitted to the historical estimates as part of the stock assessment and used to project the population forward using many stochastic realizations, such as those shown in Figure 4.2. Alternatively when model fits are considered unreliable, stochastic simulations are based on re-sampling historical recruitment estimates from a period of time considered applicable to the projection period. A better approach would be to re-sample the recruits per spawners ratio (R/S) since R must be conditional on S regardless of how noisy the data may be.

Most stock assessments and associated projections do not include all the relevant sources of uncertainty (Punt *et al.*, 2012). If uncertainty is included at all, it may only include those components of variation that can most directly be quantified. Hence, bootstrap or finite-difference approximations are often used to characterize uncertainty in the biomass estimates resulting only from variation seen in the data. Alternatively, Markov Chain Monte Carlo methods can be used to further include uncertainty in key assumed parameters, such as the rate of natural mortality. However, even these methods do not often include model uncertainty, management uncertainty, or the process uncertainty associated with changes in the environment, the ecosystem or even the population (see the analysis of Taylor, 2011, for an example of how environmental variables might be considered in the decision-making process). Consequently, the variation shown in projections underestimates the true level of uncertainty, and the expectations associated with rebuilding timelines should be tempered given these considerations. That being said, quantification of the relative uncertainty of predictions is useful, as is the motivation to decrease the uncertainty of the stock assessment provided by such calculations, for example through improved data collection and model design. Some sources of uncertainty, such as whether recruitment will be better or worse than expected in a specific future year and the form of the stock-recruitment relationship are more difficult to resolve.

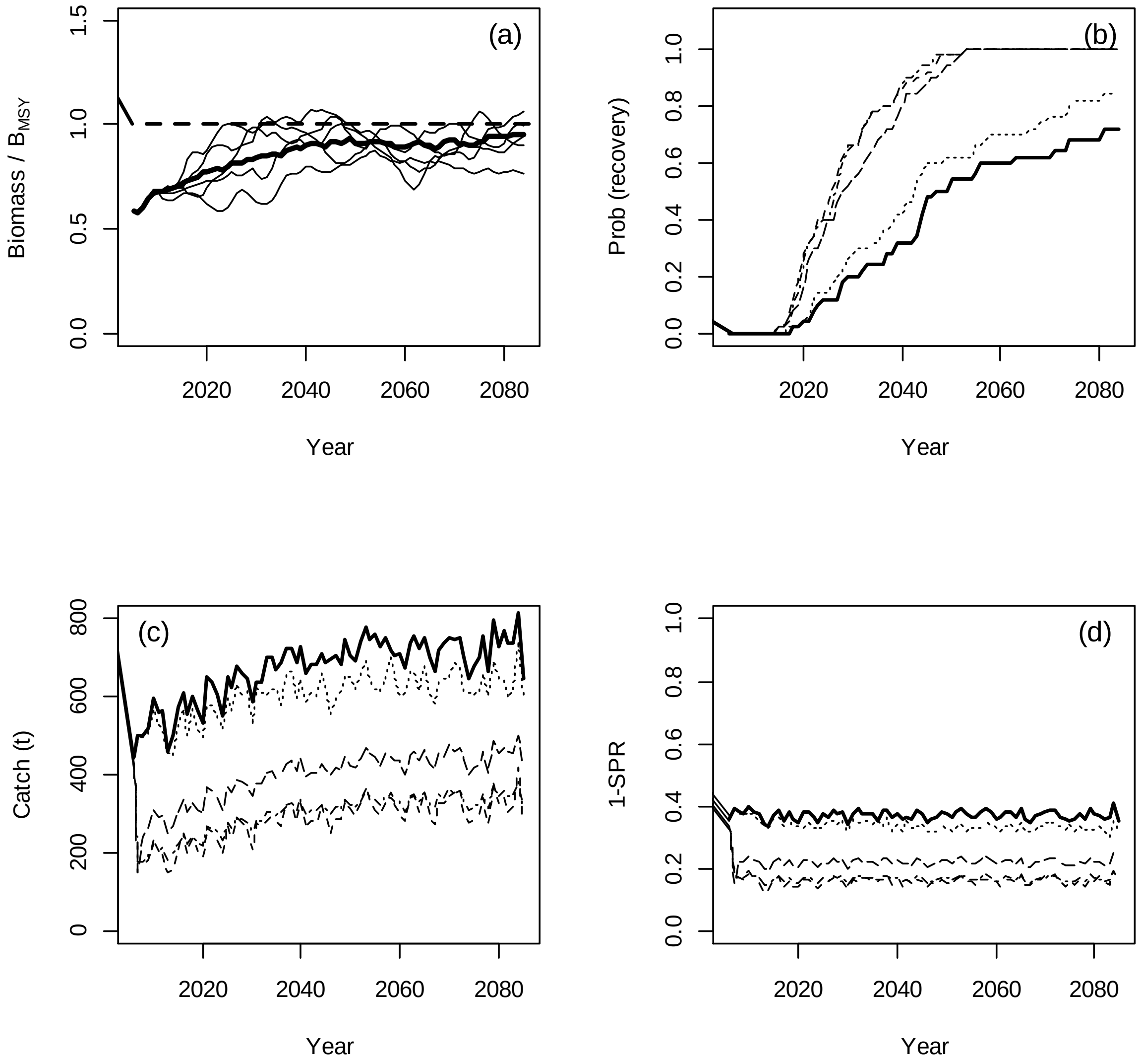

FIGURE 4.2: (a) Multiple stochastic time-trajectories of biomass relative to B_{MSY} (light lines) and the median of these stochastic projections (solid line) for one example management strategy, (b) the probability of recovery to B_{MSY} for five management strategies, (c) the expected catch for five management strategies, and (d) the fishing intensity for five management strategies. Fishing intensity is measured as the complement of Spawning stock biomass Per Recruit (1-SPR). The solid lines in (b) – (d) correspond to the management strategy in (a).

Alternative approaches for conducting assessments and rebuilding analyses

Best assessment models

The standard approach used in most regions for adjusting catch limits or other management regulations, whether or not a stock is under a rebuilding plan, involves the use of a single "best" estimate of current or projected stock size. Often a range of stock assessment models of varying levels of complexity or several alternative configurations of a standard stock assessment model are applied, and the "best" of these is selected using some formal model selection criteria, examination of residuals, or consideration of the relative plausibility of each model based on assessment by a review group (Butterworth, 2007).

Using a single best model for estimating stock levels, setting biological reference points, assessing stock status, and conducting rebuilding projections is the most common approach for formulating management advice in the U.S. This approach has its advantages. One practical advantage is that the use of a single projection model implies that only one set of values of model parameters (and their corresponding measures of uncertainty) is needed in developing projections and evaluating management options. Given that many alternative rebuilding management strategies might be explored, this single reference model greatly simplifies the process of interpretation of outputs and reduces the time needed to evaluate scenarios. One major scientific advantage is that the strengths and weaknesses of the selected model become well explored and understood in the process of reaching a consensus on a single best model during for example, a SARC, SEDAR (SouthEast Data Assessment and Review) or STAR (Stock Assessment Review) process. Of course, developing and using several approaches (including alternative models, but also systematically tracking various data indices such as trends in CPUEs, sizes at age in surveys and catch, and life history parameters) in an ongoing manner during the assessment process is both recommended and highly informative (National Research Council, 1998a).

In contrast, multiple models might be used, and either the outcomes conditioned on each of the models being valid could be put forward to managers separately in a decision table (Walters, 1986; MacCall, 1999), highlighting the relative benefits and risks under the alternative perceived states of nature, or the results of the models might be averaged to provide a weighted best estimate of the current state of the system and of the weighted performance statistics of evaluated harvesting strategies (e.g., Brodziak and Legault, 2005). Model averaging may create a practical advantage by arriving at a single outcome which can be agreed by a council SSC and hence used for council decision making, when there are multiple plausible models and no consensus on the best characterization of the state of nature. This is a common situation especially when there are many parties involved in the science and management decision process. Model averaging can be advantageous if the models represent a balanced perspective of alternative states of nature, and when the uncertainty associated with model choice can be carried forward in an integrated way. In other situations, model averaging may facilitate compromise that allows decision making to move forward when parties cannot agree. However, one must avoid smoothing over inconsistent results in the process.

Furthermore, model averaging prematurely may hinder the deeper understanding arrived at through debate and consensus building in trying to understand the consequences of alternative characterizations of states of natures and identifying the associated risks, especially if the models under consideration are highly contradictory (Schnute and Hilborn, 1993).

An advantage of considering multiple models is that this is more likely to reflect the actual uncertainty in the system and to allow examination of the consequences associated with the various management decisions under the alternative states of nature. This approach was recently used in New England for Gulf of Maine cod. (NMFS SARC 55, see also Punt, 2013). A practical disadvantage of this approach is that simulations need to be developed for all the combinations of candidate management actions and possible states of nature, and management actions here refer to decision rules applied consistently over time (as opposed to, for example, single ACLs). Although this approach has been applied around the U.S., the number of combinations of models and management options can grow rapidly making the delivery of a timely and interpretable set of outcomes more challenging to accomplish.

Management strategy evaluations (MSE)

An alternative to the best-assessment approach is the “management procedure” approach or Management Strategy Evaluation (MSE), which is increasingly being used to specify management actions for commercially-exploited fish and invertebrate populations worldwide (Punt, 2006; Butterworth, 2007). Management procedures are combinations of pre-defined data used as input to calculate catch quotas (or other regulations), assessment methods or algorithms used to process the data (which may or may not include an assessment model), and harvest control rules. The fundamental difference between a management procedure and a standard “best-assessment” approach to setting quotas is that the former involves a fully-specified feedback decision rule that has been simulation-tested across a wide range of scenarios; in the latter the rule used in practice to set the quotas changes whenever best-model assumptions change. In the section on case studies we present two examples of rebuilding plans developed using the management procedure approach: those for the New Zealand red rock lobster (*Jasus edwardsii*) and the southern bluefin tuna (*Thunnus maccoyii*).

The performance of alternative management procedures are explored using computer simulation under the wide range of population dynamics models available for characterizing both populations and ecosystems. These systematic, computer-generated thought experiments allow the testing of management strategies and rebuilding plans (e.g. Punt and Ralston, 2003) before implementation, and can be used to identify different paths for achieving a set of management goals. Generally, the model scenarios used in applications of MSE focus on the various sources of scientific and management uncertainty that affect population or ecosystem projections, and future catch rates. Conceivably the models and analyses could be extended to address questions of cost-benefit trade-offs in a socio-economic context (see for example discussions in Chapter 6). While the NS1G approach to treating scientific and management uncertainty aims to reduce the risk of overfishing occurring, it is unknown if, and how much, long-term potential yield is sacrificed. There is likely to be a trade-off between risk and yield. Management Strategy Evaluation should be used to quantify the trade-off, and to help develop strategies that are robust to the main uncertainties identified while providing fishery benefits in line with specified management objectives and legal mandates.

MSE may identify HCRs or management strategies and associated rebuilding strategies that go beyond and perhaps outperform those formulated under the classical MSY perspective, including simple rules based on trends of abundance indicators, or spatial harvesting strategies (e.g., rotation, spatial closures). Management procedures can be divided into those that are “empirical” and those that are “model-based”. Empirical management procedures specify management actions directly from data collected from the fishery without using an intervening assessment model (e.g., De Oliveira and Butterworth, 2004). They may just adjust regulations in response to trends in fishery indicators, or they may also involve some empirical target or threshold. Model-based management procedures, on the other hand, commonly employ simpler models than the models used for standard assessments, to facilitate quickly exploring a variety of management options without greatly increasing the computational burden of the evaluation process. It is generally believed that empirical management procedures are more responsive to rapid changes in monitoring data, but at the expense of higher variation in catches (Butterworth and Punt, 1999; Cox and Kronlund, 2008; Punt *et al.*, 2012).

A quantitative and rigorous approach to evaluating models and developing robust control rules can be extremely helpful when multiple interpretations of the data are possible and there is no consensus on a single best model. The southern bluefin tuna case study described later in the chapter provides an example in which a lack of scientific consensus and management impasse was resolved by incorporating multiple models in the evaluation of alternative rebuilding strategies. There are, however, some caveats associated with the implementation of the analyses and models that form the basis of the MSE approach. First, these analyses can be computer-intensive and take time. Second, while the aim is to identify a fully specified decision rule to calculate the harvest controls (e.g., the ACL) directly from new and historical data, the approach should always be coupled with ongoing monitoring and periodic in-depth stock assessments to check that observed trends are within the range of possibilities considered in the simulations (Butterworth, 2008).

Interim reviews/monitoring/assessments and adaptive management options

Rebuilding Plans have to be reviewed at least every 2nd year (as outlined in Chapter 2). There is a wide range of interpretations for what constitutes a review of a Rebuilding Plan. The review can range from comparing catches expected under the Rebuilding Plan with those actually taking place, to reviewing and updating stock assessments, which might result in updated estimates of B_{MSY} (or its proxy), and consequently may cause a reevaluation of stock status relative to B_{MSY}, as explained in Chapter 3. Thus, the evaluated status of the stock may change for a variety of reasons that have little to do with changes in the ecological condition of the stock. The frequency with which stock assessments are updated sets the pace for all adaptive management decisions including possible corrective actions to management arrangements. However, the frequency of assessments, and the associated peer review process whereby the analyses and assessment results are vetted in an open forum, differs markedly across regions. In the North Pacific, assessments for many of the major species are done on an annual basis (e.g., hake, sablefish, yelloweye rockfish) with SSC peer review and more in-depth peer reviews only conducted when there is a significant change in the data or models. In New England, benchmark assessments are more often done on a three to four year cycle with full peer review, but update assessments were not typically provided during the interim period until recently. When assessments are not updated every year, the ABCs are calculated based on model projections from the terminal year of the last stock assessment conducted. The uncertainty around projected stock sizes increases with the number of years projected, contributing to implementation errors. Furthermore, the use of infrequent assessments, and therefore the reliance on longer projections to set the ABCs, tends to amplify the impact of the estimation errors in cases of severe retrospective patterns (see Chapter 3). Simulation analyses conducted by the New England Groundfish Augmented Plan Development Team[4] indicate that the magnitude of the retrospective bias (i.e., the difference between the projected mean and most current stock size estimate for a given year) tends to increase with the number of years projected (Figure 4.3). The process that defines the frequency and use of New England stock assessments is currently under review. The NEFMC will likely modify the process in the next few years to create a two tiered approach associated with a timelier process for identifying and providing benchmark and update assessments. Part of the difference between regions is historical, but part is circumstantial. Some regions show greater biological productivity and others greater species diversity. Some regions have higher human population concentrations near their shores and consequently exhibit greater human impact and influence. Programmatically, the number of assessment scientists relative to the number of stocks that are regularly assessed varies by region, although a shortage of human resources limits the capacity to fully carry out the needed analyses in all regions. Some have more litigation reflecting that assessments are more contentious in some regions (thus the review processes are more onerous and involved). And fisheries in some regions are so diverse, so spread out, and so data limited, that conducting an assessment at all is nearly impossible. All of this, of course, affects how stock status is evaluated and reevaluated, how benchmarks and thresholds are specified and updated, and the timeliness with which management actions are reexamined.

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[4] Details of the simulation analyses are provided at http://www.nefmc.org/tech/cte_mtg_docs/120824/a_110802_APDT_report.pdf

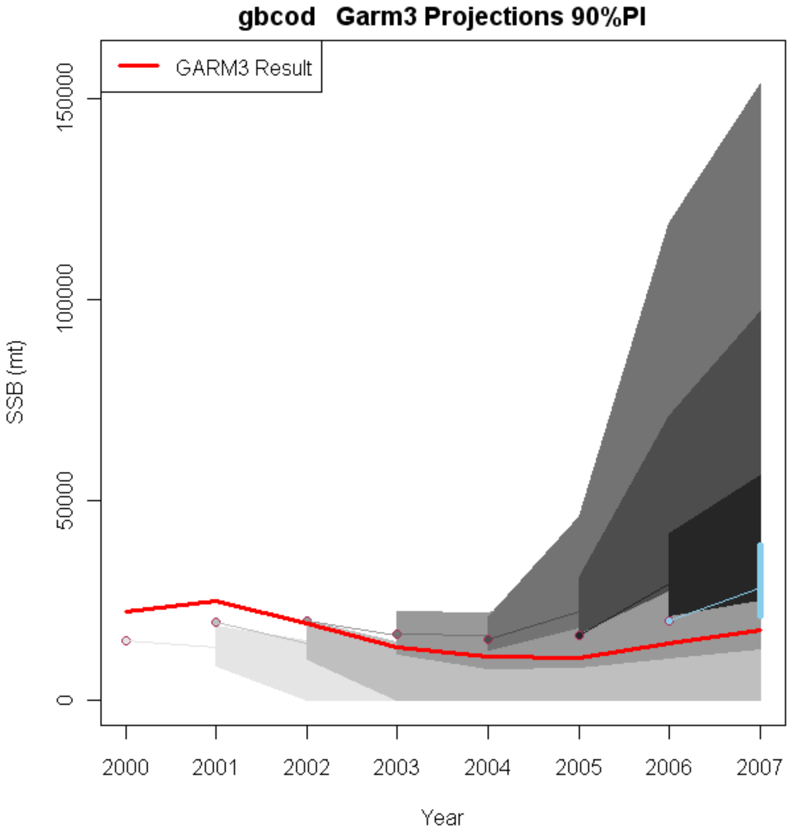

FIGURE 4.3: Spawning stock biomass (SSB) of Georges Bank cod projected from a series of retrospective stock assessments (terminal years 2000 through 2006). Analysis conducted by the New England Groundfish Augmented Plan Development Team to evaluate performance of projections by comparing the distribution of projections from each retrospective model (shown using different colors) to the estimates from the 2007 assessment (red line), taken as "the truth"[5]. The upper and lower limits are the 5th and 95th percentiles of the bootstrapped projections. A line connects the terminal year model estimate from each retrospective assessment (indicated by a dot) with the median value of the bootstraps in the following year. (Source: Liz Brook, personal communication).

Results from updated stock assessments are compared with projections to evaluate progress towards Rebuilding targets and thus to guide adaptive management decisions. However, as mentioned earlier, rebuilding fishing mortality rates are only expected to result in rebuilding within the selected time frame 50% of the time. A particular stock may not be found to be rebuilding at the expected rate even if fishing mortality has been decreased by the appropriate amount, and even if the methods for evaluating stock status do not change because, for example, recruitment may be lower than projected in the rebuilding analysis (e.g., Georges Bank yellowtail flounder). The uncertainty in these systems is great and while a biomass target and the evaluation of a range of timelines for

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[5] Details of the simulation analyses are provided at http://www.nefmc.org/tech/cte_mtg_docs/120824/a_110802_APDT_report.pdf

rebuilding are essential to formulating options and to planning, once a specified timeline is chosen, the outcome will be variable, and rebuilding may be faster or slower than expected. Attempting to adjust the F_{ACL} to try to achieve rebuilding by the specified timeline will meet with increasing lack of flexibility as T_{TARGET} approaches. In the extreme (e.g., the case of Southern New England/ Mid-Atlantic winter flounder) rebuilding may not be achievable even by setting F_{ACL} to zero. The PFMC has agreed to conduct an MSE to evaluate alternative rules for revising Rebuilding Plans.

Mixed Stock Fisheries

Fish species do not live in isolation; they represent components of a community that exhibit a range of interactions and varying degrees of overlap in space and time. The influence of ecosystem biological interactions on the exploited populations and their fisheries are discussed in Chapter 5. Here we highlight some of the technical issues associated with what is often referred to as “technical interaction”. Sparre and Venema (1998) define three types of interactions that exist in a multi-species, multi-fleet system. Biological interactions are between fish stocks and with other biological components of the system. Economic interactions are between fleets (e.g., competition). Technical interactions refer to situations in which fishing mortality is caused to one stock when fishing occurs on another[6]. This may be because several fish species are being harvested together (true multi-species fishery) or when fish are caught incidentally as bycatch. Data collection, assessment science, setting biological reference points, regulation and management can become much more complicated when technical interactions exist, especially in the context of single-species approaches to assessment and management, which is the current dominant paradigm. The impact of technical interactions involving overfished and rebuilding stocks has led to losses in yield for healthy stocks, given that the “Mixed Stock Exception” as it has been written has not been invoked. This loss in yield is expected given that F_{MSY} is a limit reference point, but is exacerbated when unproductive stocks are placed under Rebuilding Plans. For example, the valuable sea scallop (*Placopecten magellanicus*) fishery is closed when the bycatch of yellowtail flounder (*Limanda ferruginea*) exceeds the yellowtail flounder TAC (see Gedamke *et al.*, 2005 for one example of this).

Obviously, the mixed-stock problem has implications for those less productive or more vulnerable stocks as well, as an F that is reasonable for the targeted stock may cause fishing mortalities in excess of F_{MSY} for the bycaught species (several examples of this are provided by Milazzo, 2012 in the context of rebuilding). And while efforts are perennially made to modify gear to avoid or allow escapement of non-target species as well as to identify and close areas which will act as spatial refugia or to create bycatch hot-spot maps as is often done with industry participation to avoid premature closures, there will always remain the unintended removal of certain non-target species.

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[6] Here one might distinguish between technical interactions that are really due to technical issues and those that appear like technical interactions but can be minimized by changes in fishing behavior. See, for example, the SeaState program in Alaska that makes use of behavioral changes to reduce what might have been considered a technical interaction.

Time and area closures based on the overlap in distribution of the different stocks and the biology of the species in question, and gear and behavior modifications (Dunn *et al.,* 2013) could be used to keep F below F_{MSY} for all species. However, even using these techniques it may not be possible to achieve this goal without reducing F for most species in a multispecies fishery to well below target levels. If using F_{MSY} as a limit reference point for all species in a mixed-stock fishery is problematic (i.e., results in too much loss in yield and subsequent loss of social and economic benefits), what alternatives might we consider? Essentially, it is necessary to consider the risks associated with fishing some stocks above F_{MSY}. Higher levels of fishing mortality may be allowed over some time periods or in some areas only. At the extreme, the F that drives a component population to threatened or endangered status (referred to here as F_{THREAT}) would clearly be undesirable, and ultimately unacceptable under the U.S. Endangered Species Act. But in order to make the mixed-stock exception operational, it may be useful to consider F values intermediate to F_{MSY} and F_{THREAT}. For example, a reasonable upper limit on F might be F_{MSST}, the F that corresponds to the overfished threshold at a theoretical equilibrium, even though fishing at this level will increase the likelihood of stocks becoming overfished.

Ecosystem modeling (Latour *et al.,* 2003) and MSE (Ives *et al.*, 2013) would need to be used to evaluate the concomitant gains and risks associated with alternative harvest strategies, considering the fishery as a whole (i.e., the trade-offs between risks and increase in fishing opportunities associated with not having to reduce F to F_{MSY} for all stocks). One strategy could be to identify “major target” species for which F should remain below F_{MSY} (or which need to be rebuilt within a given period) and within this context identify the benefits of allowing F to exceed F_{MSY} for "non-major" stocks. This approach to mixed-stock fisheries could also be applied to rebuilding of "non-major" stocks; the time to rebuild such stocks (or whether the stocks are rebuilt to MSST or all the way to B_{MSY}) could be adjusted given the estimated impacts on fisheries for major stocks.

The consequences of applying single-species harvest strategies to multispecies fisheries are well established (e.g. Clark, 1990 and references therein). One might even argue that most fisheries take place in a multispecies setting. The move towards Ecosystem Based Fisheries Management (e.g., Pikitch *et al.*, 2004) will certainly cause scientists and managers to be confronted more often by this issue. For example, the use of multi-species production or age-structured models often gives rise to exploitation rates that are applicable to species complexes or even entire ecosystems (Mueter and Megrey, 2006). Implicit in these situations, is the differential rate of exploitation across individual species within the complex, which will need to be addressed biologically and in the context of the law. In any event, some mechanism for monitoring the status of the less productive components of mixed-species complexes will also need to be established. Note that much of this discussion on multi-species fisheries applies equally to multiple stocks of a single species. The challenge here of course is the added complexity of identification of substocks within a species. More generally, it is recognized that the trade-offs in managing mixed stocks are not easily addressed. A concern is that a management structure that allows F to exceed F_{MSY} for some components of the system, even if restricted to limited areas or time periods, could easily be abused. However, the mixed-stock issue is one that needs focused attention and careful consideration in order to increase net benefits from fisheries.

Data-poor and Knowledge-limited stocks

The 2006 amendments to the MSFCA are most readily implemented in the context of data-rich stocks, for which reliable catch and survey information are available and quantitative stock assessments can be conducted. Fortunately, a great majority of the Nation's most valuable fisheries, the focus of much of this report, have sufficient information for developing MSY-based control rules, which have proven effective for rebuilding some overfished stocks and for ending overfishing (Chapter 3).

The directives of the MSFCMA, however, must also be applied to stocks for which there is little or no information available, which is problematic. In the context of the MSFCMA, "data-poor" stocks are defined as those for which reliable MSY-based reference levels are unavailable. However, the limitations on data and knowledge vary widely both within and among management regions. What is considered data-poor in Alaska could be considered data-rich in the Caribbean. The nature of this variability and how RFMCs are dealing with it will be discussed here.

There are many data limited stocks, for which it is unrealistic to calculate an annual catch with a specified probability of preventing overfishing. A variety of *ad hoc* methods have been developed, but their performance is largely unevaluated and it is not known if they are more or less precautionary than strategies applied for stocks where probabilistic modeling is possible. It would be contrary to the precautionary approach if having less data and scientific information resulted in a higher catch being acceptable.

Some stocks are truly data limited and no attempt at an analytical assessment can be made. For other stocks, information may be available, but the assessments are not reliable because of lack of contrast in the data, inconsistencies in the findings, or biases in the resulting estimates (e.g., silver hake in New England). Such stocks might be considered "knowledge-limited", as data exist for these stocks, but the information is spotty or inconsistent. Even the situation discussed earlier about decision makers being faced with multiple plausible models might also be viewed as knowledge-limited. The MSY-oriented components of the NS1G are difficult to apply in both data-limited and knowledge-limited situations, but this is seldom acknowledged (Adkison, 2007). In reality, all stock assessments are knowledge-limited to some degree, and therefore it is more appropriate to consider fish stocks on a continuum of data and knowledge availability, from almost no information at one extreme to high-quality data and knowledge at the other (Figure 4.4). The suitability of different management approaches depends in part on where the stocks and fisheries lie on this continuum.

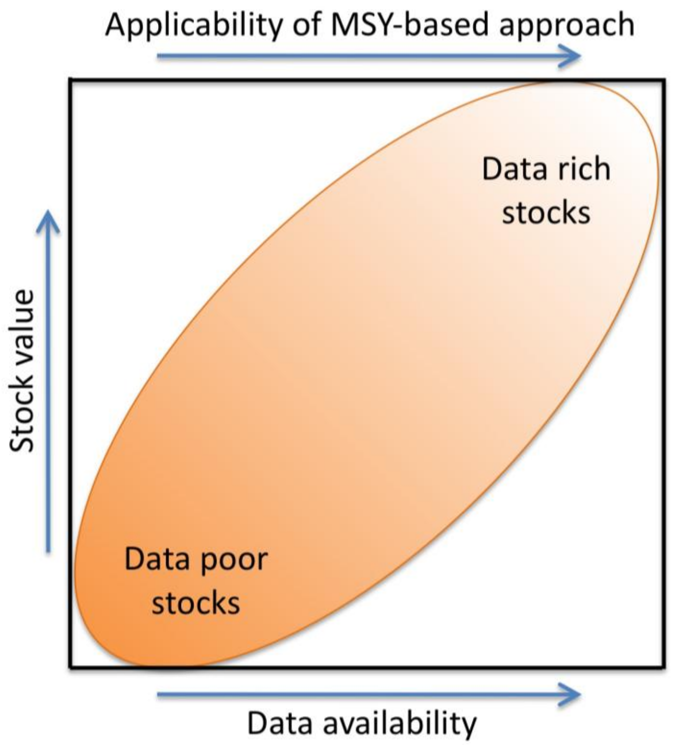

FIGURE 4.4: Conceptual diagram showing where stocks lie on three theoretical gradients: data availability (often inversely correlated with uncertainty), individual stock value (ex-vessel landings), and the relative applicability of current MSY-based control rules for rebuilding.

When rich sources of data and knowledge are available (e.g., many of the high-value stocks in Alaska), annually updated stock assessments are the norm, which are typically based on long-term catch histories and survey data, resulting in relatively accurate biological production estimates, and allowing the evaluation of projections under various management scenarios, as discussed earlier. Focus on these high-valued stocks is understandable, but has led to unintended consequences for smaller, data-limited stocks.

At the low end of the continuum in Figure 4.4, stocks are typically smaller in scale and value (e.g., many of those in the Caribbean), and are assessed with less frequency and rigor, or have never been assessed at all. However, many of the lower-valued stocks contribute importantly to the ecosystem (e.g., forage fish), or as components of multi-species fisheries (e.g., a coral reef fishery). Most coral-reef fisheries coincide with areas of high biodiversity (e.g., the Caribbean), where fishermen typically do not venture far from their local ports and target many different species simultaneously. In these small-scale, multi-species fisheries, local socio-economic conditions may depend more on overall ecosystem health than on any individual species or stock (Cinner *et al.*, 2011). In data-poor situations, therefore, other scientifically sound alternative paradigms for assessment and management may be more appropriate, that incorporate socio-economic and ecosystem considerations holistically in selecting control rules and for judging the success of rebuilding (see below).

Problems of data-poor fisheries are not insignificant in scope. Over half of the stocks or stock complexes managed in the U.S. have overfishing thresholds that are not defined, not applicable, or their overfishing status is unknown (NMFS, 2012). Data-limitation is potentially solvable. Indeed, one of the recommendations to come out of the report prepared by Restrepo *et al.* (1998) was to set as a first priority to gather the data needed to bring a stock and its assessment up to data-moderate levels. But this solution may not be realistic in many situations due to costs or the difficulties in sampling small-scale fishing operations. Knowledge-limitation, on the other hand, such as inconclusive assessments or systematic biases, may or may not be solvable by collecting more data.

The next section of the report focuses on how rebuilding plans have addressed the management of data-poor stocks within the context of the MSFCSA. Some of these methods allow determination of MSY-based control rules and ACLs without analytical stock assessments. For stocks lowest on the data-poor spectrum, alternative paradigms might be more appropriate (Bentley and Stokes 2009). Some of these methods and frameworks are reviewed and discussed in the following sections.

Methods for developing Rebuilding Plans for data-poor fisheries

Conceptually, Honey *et al.* (2010) defined data-poor methods as those that could be used to develop qualitative or quantitative control rules, without the guidance of a full stock assessment. There are several recent reviews of data-poor approaches and methods (e.g., Restrepo *et al.*, 1998; Maunder *et al.*, 2006; McCall *et al.*, 2009; Honey *et al.*, 2010; Berkson *et al.*, 2011; Dorn *et al.*, 2011; McGillard *et al.*, 2011; Punt *et al.*, 2011; ICES 2012; Brodziak *et al.*, 2012). Berkson *et al.* (2011) offer a tiered approach for setting ABCs based on a gradient of information availability, and they offer a variety of approaches for various tiers. Control rules within FMPs (e.g., Table 4.2, which illustrates the tiered system) are used by the South Atlantic FMC.

The problem of data-limited stocks is also potentially complicated by the requirement for accountability measures. This is particularly a problem for fisheries where there is a substantial lag in the availability of catch information such that within-year closure of the fishery is not feasible (e.g., many recreational fisheries). One reason that the catch may exceed the ACL is that stock size is larger than anticipated when the ACL was set. In this case, accountability measures, such as reducing the ACL the next year or closing more areas to fishing (as proposed for the Caribbean), result in a reduction in fishing mortality, stock growth and greater likelihood of exceeding the ACL again, triggering more accountability measures. Ironically, if stock size is smaller than anticipated, the ACL is less likely to be exceeded and a catch limit that is too high is likely to be maintained. Applying accountability measures without ascertaining the reason an ACL is exceeded is potentially destabilizing, particularly for data-limited stocks.

Conceivably, one could also consider recommending other forms of fishery management (e.g., marine protected areas or effort limits) that are more robust to uncertainty, except that the NS1G interpret the MSFCMA as requiring an annual catch limit except under limited circumstances (discussed above).

Alternative paradigms

There are many cases for which data and knowledge are too limited to develop robust MSY control rules and achieve the expectations of NS1G. A recent SEDAR data evaluation review, for example, concluded that ". . . despite several attempts, no acceptable quantitative assessments have been developed for Caribbean stocks because data to support traditional stock assessment methods simply do not exist for the species considered so far" (SEDAR, 2009, p.3).

The philosophy for managing stocks without quantitative stock assessments is often to invoke the precautionary approach (FAO, 2005). This approach includes developing management measures that incorporate ecosystem-level and space-time-based harvest controls such as Marine Protected Areas (MPAs), designed to protect essential habitat (i.e., breeding, nursery and feeding grounds) by excluding anthropogenic impacts at critical places and times. The value of MPAs depends in part on the value of the protected habitat. In the Caribbean, multi-species spawning aggregations for large predatory reef fishes (e.g., groupers and snappers) occur along shelf edges and reef promontories, particularly in association with vertical structures (Koenig *et al.,* 2000; Heyman and Wright, 2011; Coleman *et al.,* 2011; Kobara *et al.,* 2013). Such sites have been protected as part of an alternative strategy to rebuild overfished grouper and snapper stocks in the Caribbean and the South Atlantic (see case studies below).

Without data, however, it is nearly impossible to monitor or evaluate the outcomes of some of these rebuilding efforts. Biomass usually increases within well-enforced MPAs (e.g., Aburto-Oropeza *et al.,* 2011), and MPAs can contribute to local fish populations through both emigration and larval export (Harrison *et al.,* 2012). However, the overall effect of MPAs on rebuilding entire stocks is difficult to assess. The relationship between MPAs and stock rebuilding represents an important area for future research.

While management measures contained in the 2006 reauthorization of the MSFCMA primarily rely on output controls (i.e. ACLs/TACs) an alternative approach is to use input control rules (e.g., effort controls), iteratively and adaptively, as stocks increase or decrease. Another approach that could be used to manage and monitor fisheries without the need for full stock assessments involves the use of marine reserves as "reference" ecosystems, where unfished biomass and age structure can be compared to exploited portions of stocks. Density-ratio control rules have been proposed as a way to use a comparisons between biomass within marine reserves versus biomass outside reserves, to develop control rules, without the need for a full stock assessments (Babcock and McCall 2011; McGillard *et al.,* 2011). However, these control rules have not been applied in practice.

Challenges and unintended consequences with implementing the MSY paradigm

MSY-based biological reference points present conceptually reasonable management thresholds for information-rich stocks. Empirical evidence on directional changes or responses of such stocks to fisheries management actions is generally consistent with conventional fisheries models. However, the focus on explicitly-defined reference point estimates often overstates the degree of accuracy with which the stocks can be assessed. This uncertainty increases particularly for less valued and less-studied stocks (Figure 4.4). Stocks interact with each other and with other components of the ecosystem (see Chapter 5), which leads to several scientific and technical challenges that should be considered. An ecosystem focus is needed to incorporate these interactions, but that will present challenges too (see Chapter 5).

The problem of data-poor stocks potentially complicates the mixed stock problem. Less abundant stocks that exhibit lower productivity and limit the catch of more abundant stocks with a higher potential yield and greater value are often data limited (e.g., ocean pout in New England). While it is recognized that technical and biological interactions mean that it is generally not possible to simultaneously achieve MSY for all stocks in a fishery, the presence of stocks that are under Rebuilding Plans is likely to exacerbate this constraint (see also Chapters 2, 5, and 6). Thus, the benefits from investing in data collection and research on stocks of high economic value and high potential yield may be undermined by uncertainty in the assessments of these less abundant stocks. These interactions can lead to a refocusing of priorities to stocks that have received less attention in the past and to ecosystem-based approaches to fisheries management.

Case studies

Canary rockfish

Canary rockfish (*Sebastes pinniger*) has been exploited off the U.S. west coast extensively since during WW II due to increased demand for protein at that time. More recently, canary rockfish has been caught in most commercial and recreational groundfish fisheries over the entire U.S. west coast and is taken as bycatch of fisheries targeting other species. The proxy for B_{MSY} for this stock was set to 40% of B_0 as stipulated by the PFMC Scientific and Statistical Committee, SSC. MSST, which is 25% of B_0 for groundfish stocks, is hence 62.5% of B_{MSY}. The stock was declared overfished by the NMFS in January 2000 after stock assessments for the populations north and south of $40^{\circ}10'$N found that the spawning biomass was below MSST (Crone *et al.*, 1999; Williams *et al.*, 1999) [Figure 4.5a]. A Rebuilding Plan for canary rockfish was adopted by the PFMC based on a rebuilding analysis that was parameterized using the results of the 2002 assessment, which treated the resource as a single coast-wide population (Methot and Piner, 2002). The PFMC chose a probability of recovery to the proxy for B_{MSY} of 60% by $T_{\mathrm{MAX}} = 2076$, and hence selected a harvest strategy with a fishing mortality rate of 0.022 yr^{-1}, corresponding to 0.36 F_{MSY}. This schedule resulted in a target year for rebuilding of 2074. The management measures selected to limit catches of canary rockfish included reduced landing limits on co-occurring species, establishing extensive time/area closures, and restricting the use of trawl nets equipped with large footropes (PFMC, 2011b). Bag limits and, if necessary closed areas, have been used to limit catches in recreational fisheries. Management measures implemented prior to the

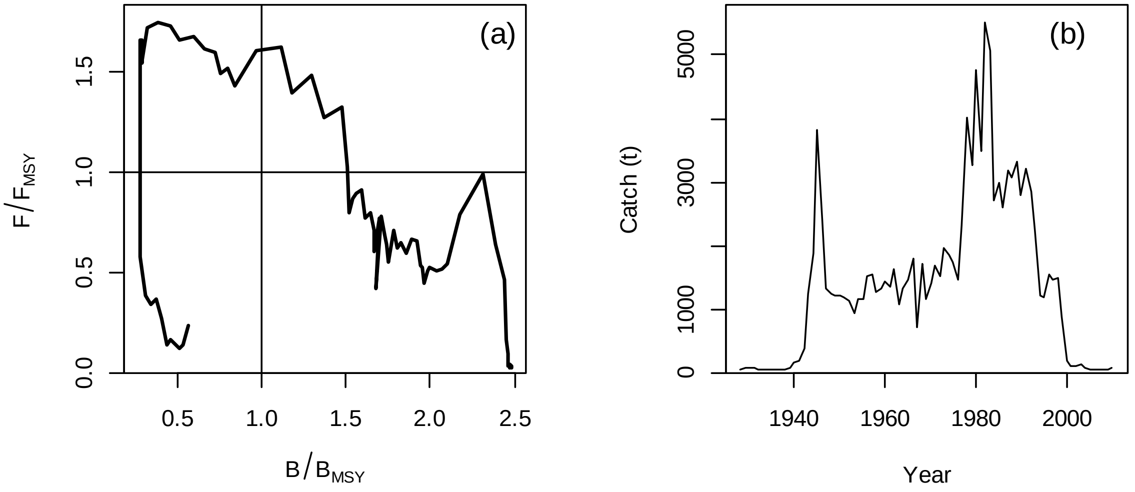

FIGURE 4.5: Phase plot (a) and annual catches of canary rockfish (b).

adoption of the Rebuilding Plan led to a large reduction in the catch of canary rockfish (from 899 tons in 1999 to 200 tons in 2000; Wallace and Cope, 2011; Figure 4.4b).

The assessments after 2002 have been based on essentially the same specifications. Nevertheless, changes and additions to the data have led to changes to the estimates of B_0, B_{MSY}, current biomass, and consequently rebuilding parameters (Table 4.3). For example, the estimate of B_0 and hence the proxy for B_{MSY} has changed over the last four assessments, ranging from 35,600 mt from the 2007 assessment to 26,000 mt from the 2009 assessment (Table 4.3). The phase-plot is relatively consistent over assessments (Figure 4.6) although there are noteworthy changes in how much below B_{MSY} the stock was depleted. All four stock assessments exhibited an inverse relationship between F and B, even after the stock dropped below B_{MSY}. The harvest strategy adopted by the PFMC in 2006 (Amendment 16-4 to the Groundfish Management Plan) had a $\mathrm{T}_{\mathrm{TARGET}}$ of 2063. Even though the target exploitation rate ($\mathrm{SPR}_{\mathrm{TARGET}}$ in Table 4.3) has not changed, the change to quantities such as B_0 has led to a reduction in $\mathrm{T}_{\mathrm{MIN}}$ (from 2048 to 2027) and $\mathrm{T}_{\mathrm{TARGET}}$ (from 2063 to 2030) based on the most recent (2011) assessment.

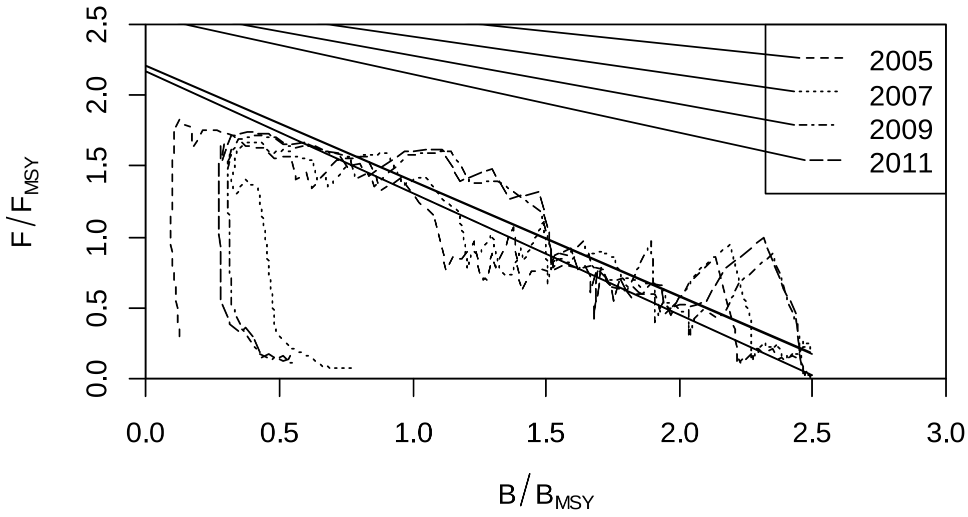

FIGURE 4.6: Change over time of the relationship between F/F_{MSY} and B/B_{MSY}.

TABLE 4.3: Changes over time in rebuilding parameters for canary rockfish

Parameter	2006 Amendment 16-4	2007 Rebuilding analysis	2009 Rebuilding Analysis	2011 Rebuilding Analysis
B_0 (mt)	34,155	32,561	25,993	27,846
Stock-recruitment steepness[1]		0.511	0.511	0.511
B_{MSY} proxy	13,662	13,024	10,397	11,138
B_{2007} (mt)		10,544		
B_{2009} (mt)			6,170	
B_{2009} (mt)				6,459
T_{MIN}	2048	2019	2024	2027
Mean generation time	23	22	22	23
T_{MAX}	2071	2041	2046	2050
T_{TARGET}	2063	2021	2027	2030
SPR_{TARGET}	88.7%	88.7%	88.7%	88.7%

SOURCE: Stewart [2009]; Wallace [2011].
1 – Fixed rather than estimated

New Zealand Rock Lobster

Spiny red rock lobster stocks off New Zealand are managed in 10 quota management areas (Figure 4.7). The management advice for four of these management areas (CRA3, CRA4, CRA7 and CRA8) is based on the application of management procedures while management advice for two other stocks (CRA1 and CRA2) relies on the results of stock assessments and projections. A management procedure is currently under development for rock lobster in management area CRA5. The management procedure for rock lobster in CRAs 7 and 8 was developed when the stocks in these management areas were assessed to be depleted to below the target level. The first comparison of alternative management procedures was conducted by Starr *et al*. (1997) when the stocks in these management areas were assessed to be one-third of B_{MSY}. The original management procedure adjusted the catch limit depending on how well catch rate compared to that expected under a rebuilding strategy. This management procedure has been refined several times, most recently during 2007 (NZ Ministry of Fisheries, 2011a), when separate management procedures were developed for these two management areas (previously a single management procedure was applied to both areas). The management procedures for CRA7 and CRA8 involve determining the TAC for a year based on the catch-rate for the previous year where the function relating the catch-rate to the TAC is piecewise linear (NZ Ministry of Fisheries, 2011a).

The management procedures for rock lobster stocks off New Zealand do not explicitly include estimates of biomass or B_{MSY}, but are rather based on catch-rate relative to desired levels. This is most obviously the case for the current management procedure for CRA4, which sets the TAC proportional to the current catch rate divided by a target catch rate raised to the power 1.4. The choice of 1.4 was made to achieve a reasonable trade-off between risk and catch. The management procedures are constructed with the intent to rebuild stocks that are below target levels, but there is no pre-specified rate of recovery or time to recovery. The management procedures include maximum allowable levels of change in TAC as well as the minimum amount of change in the TAC from the management procedure that will lead to a change in actual TAC (i.e., a recommended change in TAC of 1% will be ignored). The constraints are imposed to increase stability and avoid disruption of the fishing industry. The management procedures implemented for CRA7 and CRA8 have been successful at allowing the stocks to recover (as indicated by trends in catch-rate), but it is not clear if the stocks are at B_{MSY}.

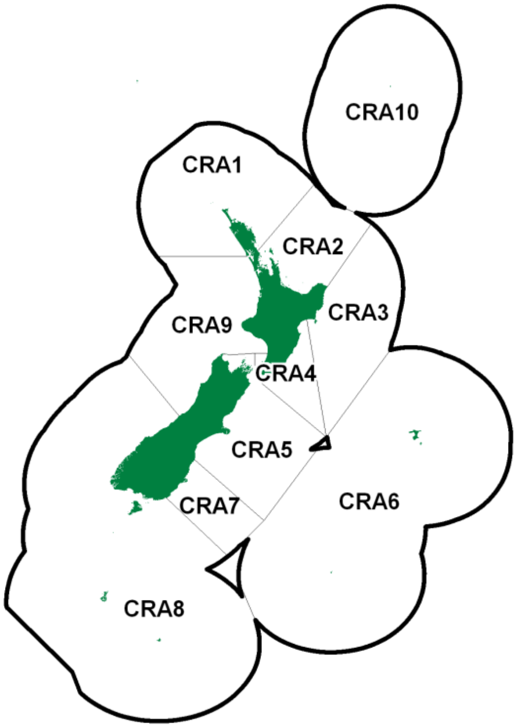

FIGURE 4.7: Management areas for spiny red rock lobster off New Zealand. (Source: Ministry for Primary Industries (2012). Fisheries Assessment Plenary, November 2012: stock assessments and yield estimates. Compiled by the Fisheries Science Group, Ministry for Primary Industries, Wellington, New Zealand. 531 p., available at http://fs.fish.govt.nz/)

Southern Bluefin Tuna

Management of southern bluefin tuna (SBT) epitomizes the challenges faced by regional fisheries management organizations charged with regulation of international high-seas fisheries. SBT is a highly-priced, large, long-lived and late maturing temperate tuna species, distributed throughout the southern hemisphere mainly in waters between 30°S and 50°S, but only rarely in the eastern Pacific. The fishery for SBT expanded rapidly during the late 1950s, reaching 80,000 mt in the early 1960s. Heavy fishing led to a continued decline of the spawning stock biomass, now estimated to be at about 5% of its unfished level (Figure 4.8) (CCSBT, 2011). During the 1980s, Australia, Japan and New Zealand – the main fishing nations at the time– voluntarily agreed to substantially reduce catches. This trilateral agreement was later formalized with the creation of the Commission for the Conservation of Southern Bluefin Tuna (CCSBT) in 1994. Korea, Taiwan and Indonesia, the other principal fishing nations for SBT, became members in 2001, 2002 and 2008, respectively.

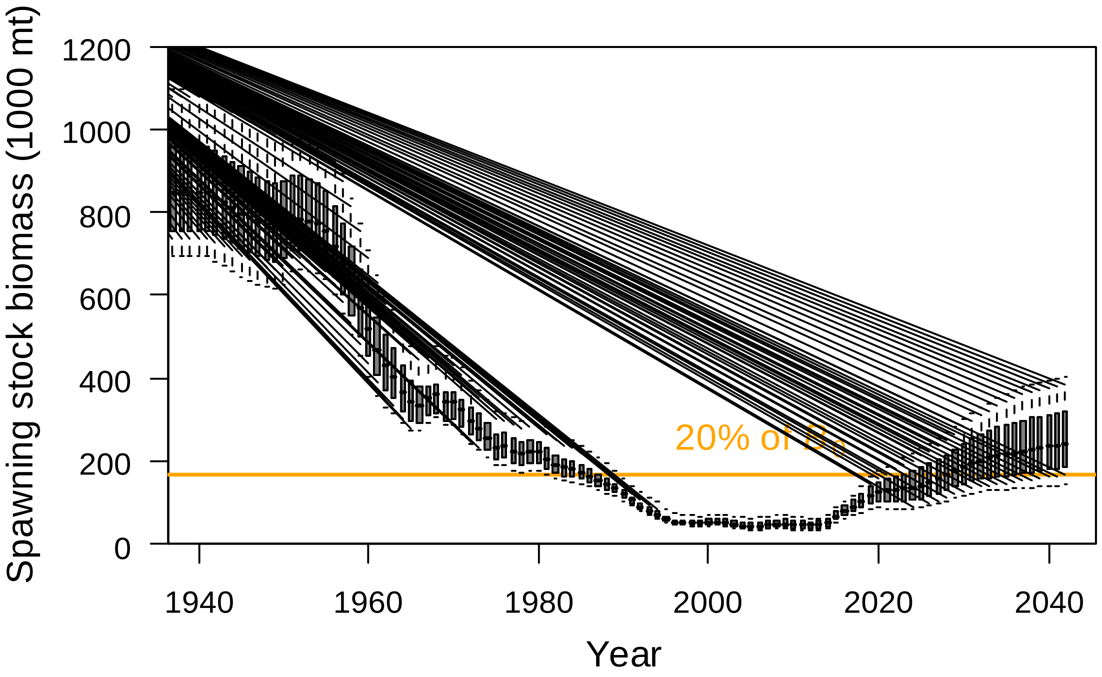

FIGURE 4.8: Distribution of historical and projected spawning stock biomass of southern bluefin tuna predicted with a reference set of population dynamics models with total allowable catches for 2012 and beyond set by the management procedure adopted by CCSBT. Horizontal line shows the interim rebuilding target to be achieved by 2035 with a 70% probability.

While further stock declines were halted after the CCSBT was established, the spawning stock did not recover, and CCSBT could not control the expansion of catches by non-members during the 1990s. A history of highly contentious assessments and widely diverging views among member countries about future prospects for the stock, and the need for further quota reductions resulted in lack of official adoption of total allowable catches (TAC) and "management paralysis". The impasse culminated in international litigation in 1999 over a proposal for experimental fishing (Polacheck, 2002).

The scientific advisory process was restructured in 2000, and CCSBT approved a multi-year plan to design a management procedure to rebuild the stock. The development of the management procedure involved member scientists proposing candidate rules for setting annual catch limits, and testing them with the same simulation models and data, and pre-agreed testing rules. Completion of the design took four years, but allegations of substantial under-reporting of historical catches in 2005 forced re-consideration of models on which testing was based. The CCSBT agreed to reduce the total allowable catch by close to 26% and new measures were put in place to control catches. A second round of testing culminated in 2011 with the CCSBT's adoption of the "Bali management procedure" to guide the setting of the global SBT TAC for 2012 and beyond. According to the assessment conducted in 2011, there is a positive outlook for the SBT stock, given that fishing mortality has been reduced to below F_{MSY} (CCSBT, 2011) and a management procedure designed to adjust TACs in response to future indicators of stock status has been adopted.

The implementation of the plan to design a management procedure changed the nature of the scientific debate. It re-directed scientists' attention away from irreconcilable arguments about what constituted the "best stock assessment" towards discussion of testing protocols and hypotheses to include in stock projections. The approach involved the selection of a weighted set of operating models deemed to represent the most important uncertainties. In its most current version[7], this so-called "reference set" is composed of 320 models, which represent alternative hypotheses about (i) the population dynamics (e.g., productivity of the stock-recruitment relationship, natural mortality parameters), (ii) interpretations of the fishery data (e.g., CPUE), and the level of under-reporting of historical caches and its impact on longline CPUE. A wide range of harvest control rules were evaluated using the reference set, and a series of "robustness tests" representing hypothetical situations and worst-case scenarios (e.g., recruitment failure, regime shifts, etc.). Some candidate management procedures were based on empirical rules that adjusted catch limits as a function of trends in longline CPUE and estimates from an aerial survey used to index recruitment; others involved population dynamics models fitted to the same data. The adopted Bali procedure uses a simple population model, and was designed to respond to trends in estimated biomass, and to how far biomass and recruitment are from selected respective thresholds (CCSBT, 2011[8]).

Management goals established initially by the CCSBT included restoring the spawning stock biomass to the 1980 level by 2020. This goal proved to be unrealistic in the light of simulation results. Given uncertainties about long-term future SBT dynamics, the CCSBT opted for an interim rebuilding target of 20% of the unfished spawning biomass (SSB_0) to be achieved in the medium term. $B_{MSY,}$ which is estimated to occur at 0.24 SSB_0 (95% confidence interval is 0.15-0.31 of SSB_0) is still a long-term rebuilding target (CCSBT, 2009). Parameters of a selected subset of candidate management procedures were adjusted to meet the interim rebuilding target at a range of time frames and with different probabilities, as specified by the CCSBT. Also, the changes in total allowable catches in each TAC update were constrained so that they did not exceed certain values. Trade-offs between several performance statistics related to trends in catches (expected catches and year-to-year variability) and risks to the stock were quantified, and the Bali procedure was recommended as "the winner". Finally, the CCSBT selected 2035 as the target year for rebuilding to the interim biomass target with a 70% probability. This time frame is a bit less than the minimum time for rebuilding this stock with 50% probability (~10 years, according to zero-catch projections based on the reference set) plus the mean generation time, which is close to 17 years.

In addition to the Bali procedure, the CCSBT adopted a set of meta-rules to decide if exceptional circumstances which fall out of the bounds considered in the testing scenarios have arisen, or if new information has become available, that merit re-

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[7] Details of the models are provided in the Report of the Sixteenth Meeting of the Scientific Committee (CCSBT, 2011)

[8] Technical specifications of the Bali procedure are available at http://www.ccsbt.org/userfiles/file/docs_english/general/MP_Specifications.pdf

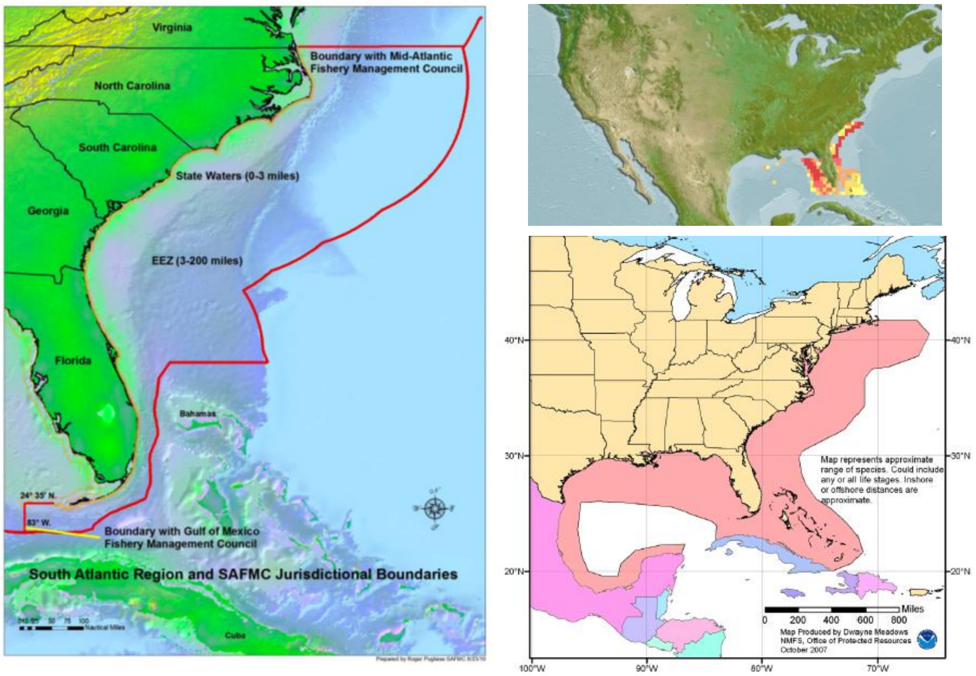
evaluation of performance. This is critically important because despite the efforts to incorporate a realistic range of uncertainties in the models, surprises often happen. The meta-rules allow management to continue to operate while alternative courses of action are evaluated.

Speckled Hind and Warsaw Grouper in the South Atlantic Region

National Marine Fisheries Service presently considers Speckled Hind, *Epinephelus drummondhayi* (SH) and Warsaw grouper, *Epinephelus nigritus* (WG) as undergoing overfishing but their status with respect to biomass is unknown. They are both listed by NOAA as Species of Concern, by the American Fisheries Society as endangered, and by IUCN as critically endangered. The range of these species is within, but extends beyond the jurisdiction of, the South Atlantic Fishery Management Council (SAFMC). SH falls under at least two council jurisdictions and WG range is shared by the Caribbean, Mid Atlantic, Gulf and Caribbean FMC jurisdictions (Figure 4.9).

SH once supported important commercial and recreational fisheries in the SE but in spite of various regulatory measures, abundance has declined and overfishing continues (Ziskin *et al.*, 2011). Commercial landings peaked in the South Atlantic in 1984 at 14.8 mt (NMFS data) but have dwindled to less than 1 mt since 1995.

The Snapper Grouper Fishery Management Plan (FMP) (SAFMC 1983) was the first ever FMP for the SE region. In this data-poor situation, SAFMC took a holistic,



Map represents approximate range of species. Could include any or all life stages. Inshore or offshore distances are approximate.

Map Produced by Dwayne Meadows NMFS, Office of Protected Resources October 2007

FIGURE 4.9: Distribution range for speckled hind (top right) and Warsaw grouper (bottom right) are centered within the jurisdiction of the SAFMC (left) but both species are shared by other council jurisdictions. (Images are from Fishbase 2011; SAFMC 2011b).

ecosystem-based approach and considered the snapper grouper complex as a mixed stock fishery for species that shared general habitat requirements and life history characteristics, and that were largely harvested by the same group of fishermen using a limited set of gears.

Exploitation continued but SH dropped in relative abundance as other species were more heavily exploited. Snapper and grouper stocks were assessed (Huntsman *et al.*, 1991) and revealed overfishing was occurring for both these species, which triggered the regulatory response and development of rebuilding plans. Amendment #6 to the FMP (SAFMC 1993) prohibited sale of SH and WG and established commercial and recreational trip limits at one fish/vessel for these species. Amendment #10 to the Snapper Grouper FMP established Essential Fishery Habitat and Habitats of Particular Concern (EFH and EFH-HAPC) for the snapper grouper complex in the SA region (SAFMC, 1998). Amendment #11 (SAFMC, 1999) established MSY proxies between 30 and 50% static SPR (Spawning Potential Ratio) for all species in the complex. Rebuilding timeframes for all overfished groupers were set at < 15 years (year 1 = 1991). SH and WG were assessed as being overfished with static SPR = 8-13% and 6-14% for SH and WG, respectively.

Commercial sale of SH and WG were prohibited 1994, the fishery became limited entry in 2006, but stocks continued to decline. Under pressure from various conservation stakeholders, the SAFMC enacted Amendment #17B (SAFMC 2010), which set SH and WG ACLs to zero and prohibited fishing for snappers and groupers below 73 m depth throughout the SE region. The closure, dubbed the “240 foot closure”, had significant negative socioeconomic consequences and triggered a backlash from commercial and recreational fishers. It was based on a misunderstanding of the habitat for these two species – an assessment that was made without consultation with fishers who, when queried, indicated that the species were often encountered in waters shallower than 240 feet. However, many other commercially important snapper grouper species, which are not considered overfished, commonly exist in waters deeper than 240 feet. In 2012, however, after intense pressure from the fishing community, the council enacted Regulatory Amendment 11 (SAFMC, 2012b), which repealed the 240-foot closure.

The South Atlantic Fishery Management Council also developed and established a network of eight Marine Protected Areas (MPAs) in Amendment 14 to the SG FMP (SAFMC, 2007). The primary purpose of these MPAs was to aid “in the recovery of overfished stocks and to ensure the persistence of healthy fish stocks, fisheries, and associated habitats.” In particular, the MPAs were designed to protect deep water snapper and grouper species. In parallel, the Council also developed a Fishery Ecosystem Plan for the South Atlantic region (SAFMC, 2009), which was later amended to allow Fishery Management Plans (FMPs) “to respond to ecosystem issues that may go across fisheries as opposed to single species management for these issues” (SAFMC, 2011b). Furthermore, the plan amended the Snapper Grouper FMP to designate new EFH and EFH-HAPC in the South Atlantic.

Most recently, the Council brought together an MPA Expert Workgroup to evaluate the existing MPAs as well as to propose new reserves that were likely to provide protection for SH and WG. The participants included patriarch commercial fishermen as well as scientists with expertise on spawning aggregations. The group proposed some reconfigurations and several new reserves along the continental shelf edge, designed to protect spawning aggregation sites for WG, SH, and other associated grouper and snapper species (SAFMC, 2012a; SAFMC, 2013). Several of the new areas selected by fishermen, were selected independently by scientists, based on geomorphology and the finding was reinforced by observations of high concentrations of fish found in spawning condition in these locations during MARMAP surveys (NOAA Fisheries Service, 2013).

The history of SH and WG management in the South Atlantic offers these lessons:

1) Socio-economic considerations are often applied in an inconsistent and *ad hoc* manner (e.g. closing the fishery of snappers and groupers below 240 feet) and have significant socio-economic impacts, but are not necessarily based on best available data.
2) Alternative management paradigms, including spatially explicit, ecosystem-based regulations might be better suited for managing data-poor reef fisheries than control rules based on F_{MSY}.
3) Marine Protected Areas that protect reef fish spawning aggregations might contribute to the management of many reef species.

Caribbean Groupers

The Caribbean Fishery Management Council's (CFMC) has jurisdiction over the Federal Waters of the U.S. Caribbean, including Puerto Rico and the U.S. Virgin Islands of St. Thomas, St. John and St. Croix (Figure 4.10). Each island is physically isolated from the others and has unique cultural identity, as well as distinctive physical environments and associated biota. Commercial Caribbean reef-fish fisheries are highly diverse comprising roughly 350 species, 180 of which are commonly harvested in Puerto Rico and the U.S. Virgin Islands. The initial FMP for the Reef Fish Fishery included 64 of these species, which make up the bulk of the commercial and recreational harvest (CFMC, 1985). These small-scale fisheries are conducted by a wide diversity of small operators, in small boats, who target many different species opportunistically with various gear types (even within a single day fishing) (CFMC 1985), and land their catch at a number of landing sites on each of the three main islands (Carr and Heyman, 2012). These factors pose challenges to data collection, assessment and management of Caribbean fisheries, problems that are compounded by low institutional and governance capacity relative to other fisheries.

The value of the reef fisheries in the U.S. Virgin Islands and Puerto Rico is low compared to many of the industrial fisheries in other regions. Nonetheless, the reef-fish complex is extremely valuable for the local communities, providing employment, income in commercial and sport fisheries, protein, recreation, and supporting social customs and cultural identity. Total reef-fish landings were estimated at 7.5 million lbs. in 1982, with a total value of $ 8.7 million (CFMC, 1985), employing roughly 2,000 commercial fishermen and 12,000 recreational boats. According to the best available data, and corroborated by interviewed fishermen, landings began to decline by the early 1980s. CPUE for the trap fishery in Puerto Rico also had declined by 57%. Still, in this time period, groupers made up roughly 23% of the shallow-water reef-fish landings (CFMC

1985). Fishing for high-trophic level groupers and snappers continued into the 1990's including intensive harvest on spawning aggregations. As groupers and snappers were fished down, fishers targeted lower trophic-level species, such as parrotfish and grunts, which now dominate landings.

The CFMC manages 179 fish stocks under four FMPs. Caribbean Grouper Unit 4 is managed under the Caribbean Reef Fish Fisheries Management Unit (FMU) and includes red grouper (*Epinephelus morio,* misty *(Hyporthodus mystacinus*), yellowedge (*H. flavolimbatus),* tiger (*Mycteroperca tigris*), yellowfin (*M. venenosa*) and black (*M. bonaci*) groupers. In 2005, the CFMC designated the complex as overfished and undergoing overfishing, commencing a 10-year rebuilding plan (CFMC, 2005). Overfished was defined as a biomass level below 20% of the spawning biomass per recruit that would occur in the absence of fishing. Yellowfin grouper was used as the indicator species for the unit, although attempts to assess it were unsuccessful (SEDAR 14, 2007). Amendment 5 to the FMP established reference points (MSY, OY), status determination criteria (MSST and MFMT); and annual catch limits (ACLs) and accountability measures (AMs) to prevent overfishing, but the development of acceptable quantitative assessments has been frustrated by lack of reliable data (CFMC, 2009). Even landings data are problematic due to under-reporting and approximate adjustment factors are estimated based on all available data and fishermen's opinions (CFMC, 2011). Yet management of these data-poor stocks is held to the same standards as any other stock in the nation.

All of the species of Caribbean Grouper Unit 4 aggregate to spawn and most have been documented in multi-species spawning aggregation sites. **S**pawning aggregations of groupers and snappers are predictable in space and time, and are highly vulnerable to fishing pressure (Coleman *et al.*, 1996; 2000). Following a precautionary management approach and recognizing severe limitations on data, enforcement and monitoring capacity, the CFMC has taken a pro-active approach closing several areas containing spawning aggregations in the region, in an effort to reduce fishing mortality. These closures include one for red hind near St. Thomas (CFMC, 1996), a site known to harbor a mutton snapper spawning aggregation off the southwestern tip of St. Croix, seasonally closed in 1993-1994 (Kojis *et al.*, 2009), seasonal and area closures at Grammanik Bank south of St. Thomas, Lang Bank east of St. Croix and several others. The most recent closure, Bajo de Sico, Puerto Rico, was suggested by fishermen for its importance as a multi-species spawning aggregation site for groupers (red hind, Nassau, and yellowfin grouper) (CFMC, 2012).

Protection of a red hind spawning aggregation in the Virgin Islands lead to a 400 fold increase in red hind biomass aggregating at the site in only four years (Beets and Friedlander 1999), and to the recovery of other species including yellowfin grouper and Nassau grouper (Kadison *et al.*, 2009). Recovery has also been documented at Riley's Hump, a multi-species spawning aggregation site in the Florida Keys National Marine Sanctuary (Burton *et al*., 2005). Nassau grouper have recovered after the protection of their spawning aggregations in the Cayman Islands (Heppel *et al.*, 2012).

While affording seasonal protection is valuable, data from other locations that are less impacted by fishing suggest that multi-species spawning sites have a predictable geomorphology (Heyman and Kobara 2012; Kobara *et al.*, 2013) and are used for spawning throughout the year and would benefit from year-round protection (Coleman *et al.*, 1996; 2000; Claro and Lindemen, 2003; Heyman and Kjerfve, 2008; Heyman, 2011; Whaylen *et al.*, 2004). Closures could be complemented by monitoring programs designed to evaluate the status of all of the stocks that use the spawning sites. Such programs could be implemented efficiently via collaborative programs with fishermen using both fishery dependent and independent data. Networks of MPAs that encompass multi-species spawning aggregation sites can serve as source sites for the recovery of grouper and snapper throughout their geographic range, and at the same time be the basis for marine ecotourism, a non-consumptive use of the resource that would contribute to local socio-economic conditions (Heyman *et al.*, 2010).

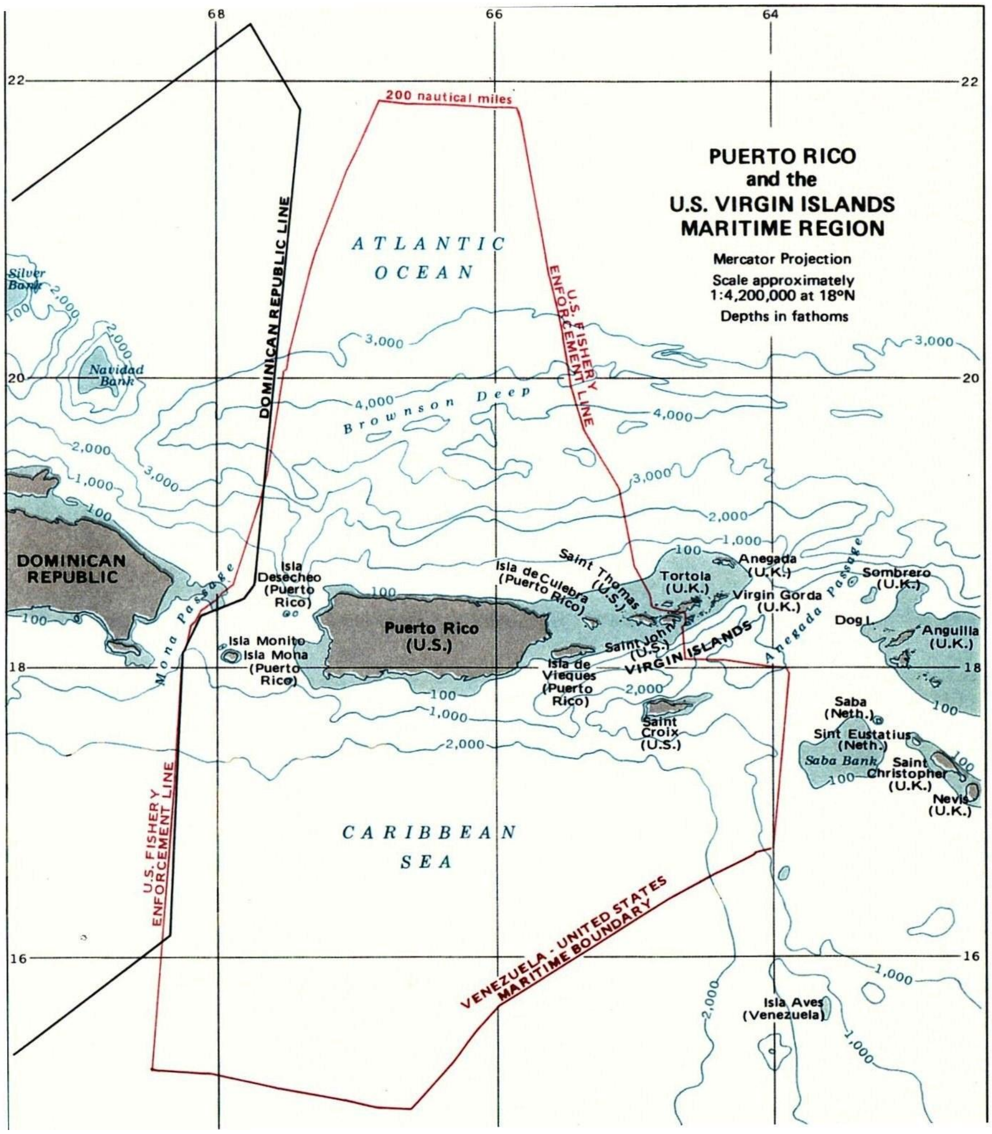


FIGURE 4.10: Caribbean Fishery Management Council jurisdiction includes the tropical waters around Puerto Rico and the US Virgin Islands.

FINDINGS

4.1: *Fish stocks can be considered on a continuum of data and knowledge availability. As a result, there is variation around the nation in definitions for overfishing and being in an overfished state resulting from differences in data availability, standards, and practices among regions.*

4.2: *There is a discontinuity and potentially large decrease in target fishing mortality as a result of the distinction made between fishery management plans for a stock that has not been declared overfished and for rebuilding the same stock as required to accommodate the MSFCMA. Alternative harvest control rules that gradually reduce fishing mortality as estimated stock size falls below* B_{MSY} *could result in a lower likelihood of a stock becoming overfished, as well as providing for rebuilding if necessary.*

4.3: *Although rebuilding plans need to be reviewed every 2nd year, these reviews do not always involve updating quantitative stock assessments. Furthermore, the frequency of assessments varies within and among regions, from stocks that have never been assessed to stocks that are assessed annually. More frequent assessments might lead to more frequent but less extreme changes in ACLs or other "course corrections", and closer adherence to fishing mortality targets.*

4.4: *Estimating fishing mortality reference points seems to be more robust to uncertainty than estimating biomass reference points.*

4.5: *The impact of technical interactions involving overfished and rebuilding stocks has led to loss in yield for healthy stocks, given that the "Mixed Stock Exception" has not been invoked in part due to the narrow range of situations to which it applies under the MSFCMA. This loss in yield is expected given that* F_{MSY} *is a limit reference point but is exacerbated when unproductive stocks are placed under rebuilding plans. Technical interactions also make it more difficult to rebuild stocks that are caught as bycatch of fisheries targeting other species. The operational feasibility of the mixed-stock exception should be reconsidered subject to assurances that bycaught species are not driven to unacceptably low abundance or become threatened.*

4.6: *In the face of the high uncertainty involved in population projections, the emphasis placed on achieving a biomass rebuilding threshold in a defined time frame may call for severe reductions in target fishing mortality (well below* F_{MSY}*) when a stock's rebuilding is slower than expected.*

4.7: *When information is limited, there may be alternative, scientifically-sound strategies that could lead to better management results than the* B_{MSY} *target oriented approach prescribed by the NS1G. In the case of data-poor stocks for which analytical assessments are not available, and catch limits are therefore difficult to establish, empirical rebuilding strategies that rely on input controls to reduce fishing mortality may be more effective and defensible than strategies based on annual catch limits and* B_{MSY} *targets.*

4.8: The use of fully specified harvesting strategies that have been tested across a range of plausible models provides one approach, which has been proven useful, for dealing with the biological and implementation uncertainties.

TABLE 4.1: Summary of the overfishing and overfished definitions for various selected categories of fish and invertebrate species by council.

(a) New England Fishery Management Council

Species Group	Overfished Definition	Overfishing Definition
Northeast Multispecies		
GOM Cod	$B < \frac{1}{2} B_{MSY}$	$F > F_{40\%}$
GBK Cod	$B < \frac{1}{2} B_{MSY}$	$F > F_{MSY}$
GOM and GBK Haddock	$B < \frac{1}{2} B_{MSY}$	$F > F_{MSY}$
American Plaice	$B < \frac{1}{2} B_{MSY}$	$F > F_{MSY}$
Witch Flounder	$B < \frac{1}{2} B_{MSY}$	$F > F_{40\%}$
GOM and GBK Winter Flounder	$B < \frac{1}{2} B_{MSY}$	$F > F_{MSY}$
CC/GOM and GBK Yellowtail Fldr	$B < \frac{1}{2} B_{MSY}$	$F > F_{MSY}$
SNE/MA Yellowtail Fldr	$B < \frac{1}{2} B_{35\%}$	$F > F_{40\%}$
Other Species[1]	$B < \frac{1}{2} B_{MSY}$	$F > F_{MSY}$
Scallops	$B < \frac{1}{2} B_{MSY}$	$F > MFMT$[2]
Monkfish	$B < \frac{1}{2} B_{MSY}$	$F > MFMT$[3]
Small Mesh Multispecies		
Silver Hake	$B < \frac{1}{2} B_{MSYproxy}$	$F > F0.1$
Northern Red Hake	3 yr mean $CPUE_{survey} < 1.6$ kg/tow	$F > F_{MSYProxy}$
Southern Red Hake	$CPUE_{survey}$[4] < 25th percentile of $CPUE_{survey}$ series	NA
Offshore Hake	$CPUE_{survey}$[4] < 25th percentile of $CPUE_{survey}$ series	NA
Dogfish	$B < \frac{1}{2} B_{MSYproxy}$[5]	$F > MFMT$[6]
Red Crab	$B < \frac{1}{2} B_{MSY}$ or Average $CPUE < CPUE_{base}$ for 3 yrs $CPUE_{survey}$ or Average $CPUE < CPUE_{threshold}$ in 1 yr	$F > F_{MSY}$[7]
Skates		
Winter and little	$CPUE_{survey}$[4] < 75th percentile of $CPUE_{survey}$ series	3 yr $CPUE_{average}$[8] > 20%
Barndoor	$CPUE_{survey}$[4] < ½ 0.81kg/tow	3 yr $CPUE_{average}$[8] > 30%
Thorny	$CPUE_{survey}$[4] < 75th percentile of $CPUE_{survey}$ series	3 yr $CPUE_{average}$[8] > 20%

Smooth and clearnose	$CPUE_{survey}$[4] $<$ 75th *percentile of* $CPUE_{survey}$ *series*	3 yr $CPUE_{average}$[8] > 30%
Rosette	$CPUE_{survey}$[4] $<$ 75th *percentile of* $CPUE_{survey}$ *series*	3 yr $CPUE_{average}$[8] > 60%

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1 – Pollock, redfish, white hake, Atlantic halibut, ocean pout, windowpane flounder, SNE/MA winter flounder, wolfish, herring.
2 – MFMT is based on a stochastic YPR model.
3 – MFMT is based on an F_{MSY} proxy, F_{MAX}.
4 – Overfished status is determined based on current survey CPUE measured as mean weight per tow when compared to a percentile of the CPUE from a specified time series of stable CPUEs .
5 – B_{MSY} proxy set to SSB_{MAX}.
6 – MFMT is based on an F_{MSY} proxy, *Frep*.
7 – Fmsy defined as average landings/average survey index during stable period.
8 – Overfishing determined as a decline of X% or more in the 3 year moving average of CPUE measured as mean weight per tow in the Autumn survey.

(b) Mid-Atlantic Fishery Management Council

Species Group	Overfished Definition	Overfishing Definition
Squid, Butterfish, Mackerel	Unknown	Unknown
Bluefish	$B < ½ B_{MSY}$	$F > F_{MSY}$
Spiny Dogfish	$B < 100{,}000$ *mt*	$F > F_{MSY}$
Summer Flounder	$B < ½ B_{MSY}$	$F > F_{MSY}$
Scup	*Spring survey < 2.77 kg/tow*	$F > F_{MSY}$
Black Sea Bass	Unknown	$F > F_{MAX}$
Surfclam	$B < ½ B_{MSY}$	$F > F_{MSY}$
Ocean Quahog	$B < 977{,}000$ *mt*	$F > F_{25\%}$
Tilefish	$B < ½ B_{MSY}$	$F > F_{MSY}$
Monkfish	$B < ½ B_{MSY}$	$F > F_{MAX}$

(c) South Atlantic Fishery Management Council

Species Group	Overfished Definition	Overfishing Definition
Shrimp	$B < ½ B_{MSY}$	$F > F_{MSY}$
Spiny Lobster[1]	*NA*	*NA*
Black Sea Bass	$B < (1-M)B_{MSY}$	$F > F_{MSY}$
Red Porgy	$B < (1-M)B_{MSY}$	$F > F_{MSY}$
Red Snapper	$B < (1-M)B_{MSY}$	$F > F_{MSY}$
Snowy Grouper	$B < MSST$	$F > F_{MSY}$
King Mackerel[1]	$B < (1-M)B_{30\%}$	$F > F_{30\%}$
Yellow Snapper	*NA*	*NA*
Red Grouper	$B < ½ B_{MSY}$	$F > F_{MSY}$

1 – Managed jointly with GMFMC.

(d) Gulf of Mexico Fishery Management Council

Species Group	Overfished Definition	Overfishing Definition
Shrimp	$B < ½\ B_{MSY}$	$F > F_{MSY}$
Spiny Lobster[1]	*NA*	*NA*
King Mackerel[1]	$B < (1\text{-}M)B_{MSY}$	$F > F_{MSY}$
Spanish Mackerel[1]	$B < (1\text{-}M)B_{MSY}$	$F > F_{MSY}$
Cobia	*NA*	*NA*
Gray Triggerfish	$B < (1\text{-}M)B_{30\%}$	$F > F_{30\%}$
Greater Amberjack	$B < (1\text{-}M)B_{MSY}$	$F > MFMT$
Red Snapper	$B < (1\text{-}M)B_{26\%}$	$F > F_{26\%SPR}$
Gag	$B < (1\text{-}M)B_{MSY}$	$F > F_{MAX}$
Red Grouper	$B < (1\text{-}M)B_{MSY}$	$F > F_{MSY}$

1 – Managed jointly with SAFMC.

(e) Caribbean Fishery Management Council

Species Group	Overfished Definition	Overfishing Definition
Queen Conch	*NA*	*NA*
Caribbean Grouper	*NA*	*NA*
Spiny Lobster	*NA*	*NA*

(f) Pacific Fishery Management Council

Species Group	Overfished Definition	Overfishing Definition
Groundfish		
Rockfish and elasmobranches	$B < ¼\ B_0$	$F > F_{50\%}$
Roundfish	$B < ¼\ B_0$	$F > F_{45\%}$
Flatfish	$B < 1/8\ B_0$	$F > F_{30\%}$
Salmon	$B < ½\ B_{MSY}$ *or* $¾\ B_{MSY}{}^{5}$	$F > MFMT^{4}$
High migratory species[1]	$B < (1\text{-}M)\ B_{MSY}$ *for* $M \leq ½$ $B < ½\ B_{MSY}$ *for* $M > ½$	$F > F_{MSY}$
Coastal Pelagic Species	$B^{1+} < 150{,}000$ *mt (Pacific sardine)*[2] $B^{1+} < 18{,}200$ *mt (Pacific mackerel)* [2] *Catch > 31,000 mt (Jack mackerel)*[3] *Catch > 9,750 mt (northern anchovy northern pop)* [3] *Catch > 25,000 mt (northern anchovy southern pop)* *Egg escapement-per-recruit < 30% unfished (market squid)* [3]	*Catch > OFL*

1 – defaults; used unless an RFMO develops alternative definitions.
2 – B^{1+} is the biomass of animals aged 1 and older
3 – There are monitored species in the CPS Fishery Management Plan.

4 – MFMT (Maximum Fishing Mortality Threshold) is less than or equal to F_{MSY} where F_{MSY} is either estimated or for Chinook salmon assumed to be 0.78.
5 – Biomass is a three-year geometric mean of annual spawning escapement B_{MSY} is estimates variously for each salmon stock.

(g) North Pacific Fishery Management Council

Species Group	Overfished Definition	Overfishing Definition
Groundfish		
Tiers 1 & 2	$B < ½ B_{MSY}$	*Catch > OFL*
Tier 3	$B < ½ B_{35\%}$	*Catch > OFL*
Tierd 4-6	*N/A*	*Catch > OFL*
Bering and Aleutian Islands crab		
Tiers 1 & 2	$B < ½ B_{MSY}$	*Catch > OFL*
Tier 3	$B < ½ B_{35\%}$	*Catch > OFL*
Tier 4	$B < ½ B_{MSYProxy}$[1]	*Catch > OFL*
Tier 5	*NA*	
Scallops	*NA*	*Catch >OFL*

1 – An average biomass selected by NPFMC SSC as reflecting when the stock was at B_{MSY}.

(h) Western Pacific Fishery Management Council

Species Group	Overfished Definition	Overfishing Definition
Pelagics	$B <$ max (1-M, ½) B_{MSY}	$F > F_{MSY}$ *if* $B>$ max (1-M, ½) B_{MSY} $F > F_{MSY}\, B/$ max (1-M, ½) B_{MSY} *if* $B>$ max (1-M, ½) B_{MSY}
Bottom Fish	$B <$ max (1-M, ½) B_{MSY}	$F > F_{MSY}$ *if* $B>$ max (1-M, ½) B_{MSY} $F > F_{MSY}\, B/$ max (1-M, ½) B_{MSY} *if* $B>$ max (1-M, ½) B_{MSY}
Crustaceans	*NA*	*NA*
Precious Corals	*NA*	*NA*
Corals	$Effort>Effort_{MSY}$	$CPUE<CPUE_{MSY}$

Table 4.2: Tiers of stocks as used to define OFLs and ABCs to stocks with various levels of data and/or knowledge limitations (SAFMC, 2011a)

Tier 1 – Assessed stocks: Whenever possible, ABC recommendations should conform to an ABC control rule that is based on the probability of overfishing (i.e., P* approach)

Tier 2 - Depletion based stock reduction analysis (DB-SRA): If the information necessary to implement the Council's approved ABC control rule is not available (e.g., MSY reference points, projected stock size, distribution of OFL, etc.), then the basis of the ABC should be explicit about what aspects of the derivation were based on expert judgment.
- Requires full history of landings and other life history info for the stock
- Gives a pdf of OFL. Could apply P* or other risk/p level to derive ABC

Tier 3 - Depletion-corrected average catch (DCAC) (MacCall, 2009): If components of the ABC control rule cannot be provided, a provisional ABC should be based on alternative approaches, but deviation from the control rule should be justified.
- Requires less data than 2nd tier
- Provides provisional sustainable catch.

Tier 4- Catch data only: Difficult to prescribe.
-Requires judgment and careful consideration of all available sources, which may vary greatly between stocks falling in this tier

5

ECOSYSTEM CONSIDERATIONS

Introduction

An ecosystem approach to fisheries management is “one that is geographically specified, adaptive, takes account of ecosystem knowledge and uncertainties, considers multiple external influences, and strives to balance diverse societal objectives” (Sissenwine and Murawski, 2004). The ecosystem approach recognizes both the effects of fisheries on the ecosystem and the effects of ecosystem state and variability on the fisheries (Crowder *et al.*, 2008). Much has been written about what elements should be included in such an approach since the U.S., together with many other countries, embraced an ecosystem approach to fisheries management (Pikitch *et al.*, 2004; Murawski, 2007).

Section 207 of the Sustainable Fisheries Act (SFA) provided initial guidance on inclusion of ecosystem principles in the management of the Nation’s fisheries.[1] This guidance was included based on the perceived realization that traditional management measures were insufficient to ensure sustainable fisheries. The basic premise is that fished stocks form essential components of complex marine ecosystems that must be well understood in order to manage them. Guidelines recognize that there exist multiple interdependent relationships among stocks, their fisheries, and the ecosystem in which they reside. This section of the SFA mandated formation of the Ecosystems Advisory Panel to the National Marine Fisheries Service, which was tasked to review the progress towards incorporation of ecosystem principles in FMPs.[2] The Panel report (NOAA, 1999) specified the need to better account for and minimize by-catch and discard of fish, identify essential fish habitat and take measures to protect it, and determine the effects of fishing on the environment.

While explicitly considering factors beyond the single-species dynamics is clearly a sound objective, the details of how to do this are still a subject of ongoing research, and a variety of approaches are being pursued. Much work is being conducted by the U.S. Fisheries Science Centers to incorporate ecosystem considerations into fisheries management (Hollowed *et al.*, 2011). These approaches include incorporating indices of environmental and biological conditions into stock assessment projections (Hare *et al.*, 2010), using multi-species and food-web models to assess the effects of harvesting strategies (Link *et al.*, 2011), and investigating new forecasting methods (Deyle *et al.*,

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[1] Sustainable Fisheries Act, Pub. L. No. 104-297 § 207, 110 Stat. 3559, 3621 (1996).
[2] Sustainable Fisheries Act, Pub. L. No. 104-297 § 207, 110 Stat. 3559, 3621 (1996).

2013). These models could be used to support rebuilding efforts but no comprehensive review of these models is provided here.

In the context of fish stock rebuilding, changes in the ecosystem can alter rebuilding rates and the target biomass level to which an overfished species should be rebuilt. The most direct ecosystem consideration for rebuilding is that "MSY stock size (B_{MSY}) means the long-term average of the stock or stock complex…that would be achieved by fishing at F_{MSY}," where MSY is defined "under prevailing ecological, environmental conditions."[3] The challenge is to define these MSY-based reference points, given background levels of variability and to adjust them as necessary, in response to ecosystem changes. This chapter is structured around the main ecosystem considerations for fisheries management, with sections on climate, habitat, and ecological interactions. We then discuss the possibility of incorporating ecosystem factors into rebuilding plans, and end with findings and conclusions.

Assumptions about population structure and dynamics

We start by considering ecosystem effects on population dynamics that are implicit rather than explicit. Density-dependent processes are expected to reduce the population growth rate as depleted populations rebuild. Density dependence is integral to production models and is also included in rebuilding projections when a stock-recruitment model is used. Individual growth rates can also vary with population density and the amount of available food (see discussion on Ecological interactions below). As an example of changing reproductive potential, the growth rate of Georges Bank haddock (*Melanogrammus aeglefinus*) declined following recruitment of the very large 1999 and 2003 year classes (Brodziak *et al.*, 2008). Density-dependent growth is typically not considered in population projections but, where it occurs, it affects the calculation of biological reference points such as B_{MSY} and F_{MSY}.

Rebuilding plans require projections of stock dynamics into the future, which are sensitive to assumptions about the future state of the ecosystem. The population models commonly used to project stock rebuilding are generally single-species (no interactions among species). They assume that historical conditions in the ecosystem (including variability) will continue into the future (stationarity assumption), and the biomass reference points are calculated under stable equilibrium assumptions. These assumptions of single species, stable variability, and stable equilibrium may oversimplify the problem; recent observations suggest more complex *dynamical* behavior operating in fishery ecosystems (Box 5.1).

In particular, natural populations can exhibit dynamical behaviors broadly described as nonlinear. These include multiple equilibria (regime shifts), limit cycles, and chaotic dynamics (May, 1973). More generally, nonlinear dynamics simply means that population behavior depends on ecosystem state. Indeed, state-dependence is how nonlinearity is measured in ecological time series (S-maps, Sugihara, 1994); it implies that ecosystem effects must be studied synergistically, not one factor at time. Nonlinear

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[3] 50 C,F.R. 600.310 (e)(i) (2009).

dynamics can be driven by fishing, environmental variability, or trophic interactions (Steele and Henderson, 1984; Anderson *et al.*, 2008). There is growing evidence of nonlinear dynamics in fish populations (Dixon et. al., 1999; Glaser *et al.*, 2011), as well as a growing consensus that ecosystem and multi-species effects are important. The existence of nonlinear dynamics has profound implications for the way we should think about fisheries ecosystems, how we model fish populations, and ultimately our expectations for stock rebuilding.

BOX 5.1
A Simplifying Paradigm for Complex Ecosystem Dynamics

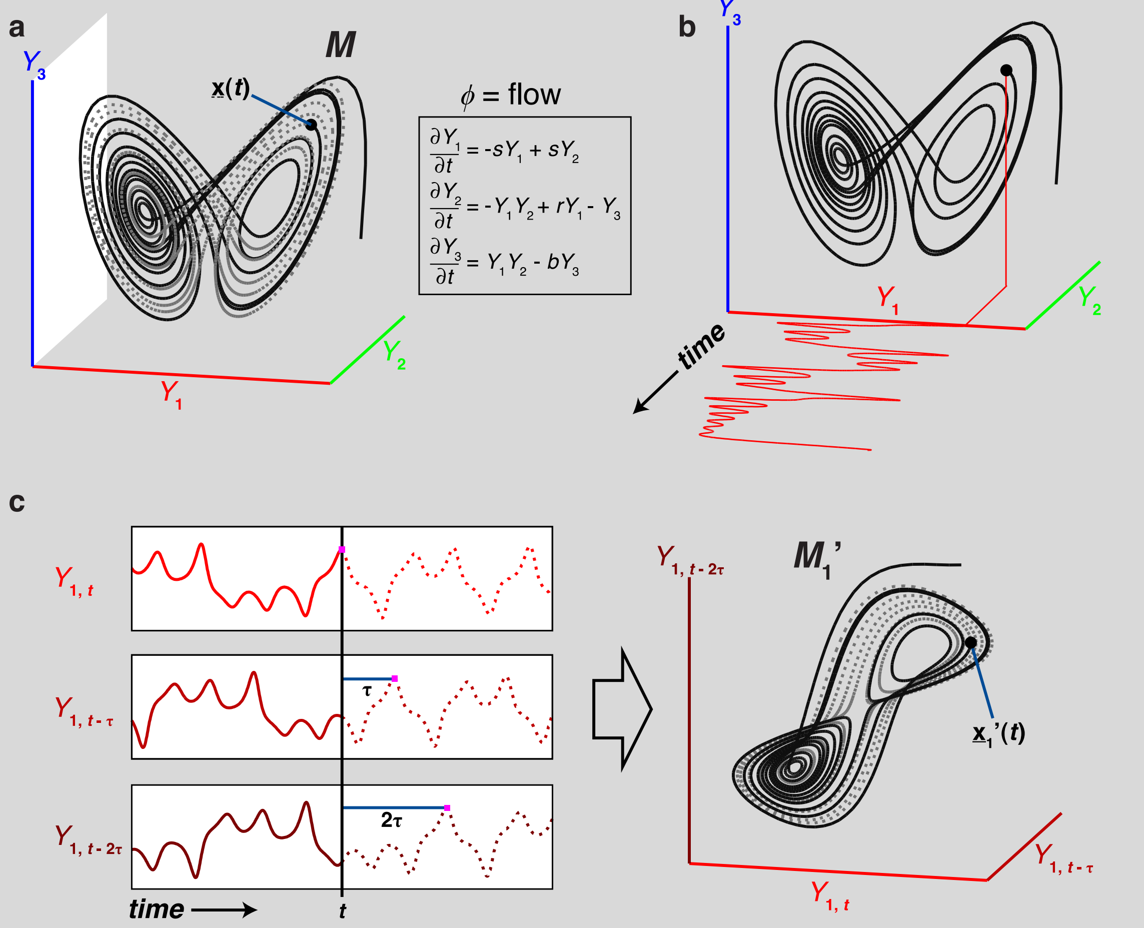
Population models used in ecology and fisheries, derived in the middle of the last century, have stability and equilibrium as foundational concepts. Although mathematically tractable and well defined, "stability" and the implication of "equilibrium" in ecological systems began to give way in the 1980s and 1990s to growing evidence that "change" rather than "constancy" is the rule, and that nonlinear instability, thresholds and chaos can be ubiquitous in nature (Sugihara and May, 1990; Grenfell, 1998; Scheffer and Carpenter, 2003). Nonetheless, models with stable equilibria and MSY remain the benchmark for fisheries assessment and management (Quinn and Collie, 2005). These equilibria are the basis for calculating the biological reference points that are used to determine fish population status and the targets for rebuilding overfished stocks.

A new paradigm explicitly recognizes the growing evidence from field measurements on natural populations that nonlinear complexity and instability are ubiquitous, and that a stable equilibrium (even multiple equilibria) though simple and manageable, is likely to be the idealized exception rather than the rule. This new paradigm views fisheries as *dynamic*, complex systems of inter-dependent variables, represented by simple non-parametric models that depend on a data-driven construct, rather than being built around assumptions of stationary variability and stable equilibrium.

The Simplifying Approach

Complex systems are typically modeled using differential (or difference) equations that describe the transition through time between different states of the system. Each *state* is represented as a vector of state variables $\underline{x}(t)$ (e.g., abundances of foxes, rabbits, and grasses, temperature, stock levels, prey levels, etc.), and the set of all states that a dynamic system transitions through forms a geometric construct known as an attractor *manifold*, ***M***. The manifold describes how ecosystem state variables relate to each other through time – a dynamic version of Huchinson's n-dimensional niche. If there are rules governing ecosystem changes (i.e., if ecosystems are not *purely* random) there is an attractor manifold to be uncovered (Box 5.1, Figure 1) (Deyle and Sugihara, 2011). Attractor manifolds determine (express) relationships among variables, and can be obtained simply by re-plotting the time series data. Constructing manifolds empirically from ecological time series is the basis of the approach. Box 5.1 Figure 1 illustrates the following three core ideas:

Box 1 Figure



Box 5.1, Figure 1. **(a)** The Lorenz butterfly attractor example. The attractor manifold ***M*** is the set of states that the system progresses through. x(*t*) is the state of the system at time *t*, and the dynamics are defined by the Lorenz equations. (**b**) A time series is simply a projection of the system states from ***M*** to a coordinate axis (Y_1 is a state variable of the system). The manifold can be constructed from the component time series**.** **(c)** Following Takens Theorem (Takens, 1981), lags of the time series $\{Y_1\}$ can act as coordinate axes to construct a shadow manifold ***M***$_1$', which maps 1:1 to the original manifold ***M*** (the visual similarity between ***M***$_1$' and ***M*** is apparent). These shadow manifolds can be used for ecosystem-based prediction, identifying causal variables, and much else. (Reproduced from Sugihara *et al.,* 2012; see also the supplemental animations).

1) Nonlinear State Dependence (Panel a). If there is an attractor manifold ***M***, that is not flat (a hyperplane) relationships between variables will depend on system state (e.g., Y_1 and Y_3 are positively correlated at some times and negatively associated at other times). Baltic Sea fisheries, for example, exhibit radically different dynamic control regimes (top-down versus bottom- up) depending on the threshold abundance of planktivores, causing the correlations between fish and zooplankton to change sign (Cassini *et al.,* 2009). Thus, if a fishery exhibits nonlinear state dependence, fish populations, fishing pressure, and environmental effects should be considered together (Deyle *et al.,* 2013).

2) Time Series as Observation Functions (Panel b). A time series $\{Y_i\}$ is a projection of the dynamics occurring on ***M*** (panel b). More generally, the Y_i are *observation functions* of the dynamics on ***M***. The Y_i may be fundamental coordinates or they may be any function (e.g., rotations or linear combinations of the original Cartesian coordinates) that maps points in ***M*** to

time series observations. The key insight is that ecological time series can appear complex because they are projections into one dimension of dynamics occurring in higher dimensions.

3) State Space Reconstruction and Takens Theorem (Panel c). If all the variables and equations governing an ecosystem were known we could construct the attractor manifold by direct simulation. In fact, it is possible to reconstruct the manifold empirically, if we only had time series for all the variables. This manifold would be an empirical expression of all of the dynamic relationships among variables observed in the data. However, in practice we may only have time series information about one species. A key result from dynamical system theory — the Takens embedding theorem — proves that one can reconstruct the dynamical attractor for a system from data in the form of lagged samples of just one variable, such as Y_i. Thus, *state space reconstruction (SSR)* is a method to recover an approximation of $\boldsymbol{M}$ from time series. This is illustrated in panel c, where the shadow manifold, $\boldsymbol{M}_1$' is constructed using lags of time series $\{Y_1\}$. The reconstruction captures the essential topology and dynamics of the original system. Further refinements include: (1) multivariate reconstructions that are more mechanistic (Dixon *et al.,* 1999; Deyle and Sugihara, 2011), (2) identifying and incorporating stochastic environmental forcing, and (3) exploring environmental scenarios (Deyle *et al.,* 2013), among other things.

This new paradigm has several important implications for fisheries science in general and for stock rebuilding in particular. The non-equilibrium nature of these models challenges the basis of calculating biological reference points, particularly biomass reference points such as unfished stock size (B_0) and B_{MSY}. Although ecosystems may have tens to thousands of interacting variables, their essential dynamics at any time may involve relatively few key variables, or dimensions. Several studies indicate that the relevant ecosystem dimension for certain fish species is often relatively low—involving from three to eight dimensions—and they also demonstrate relatively high predictability (Dixon *et al.,* 1999; Hsieh *et al.,* 2005). While forecasting skill may be high in the short term, the ability to make medium and long-term forecasts is limited by unstable dynamics and forcing by a stochastic environment (Glaser *et al.,* 2013).

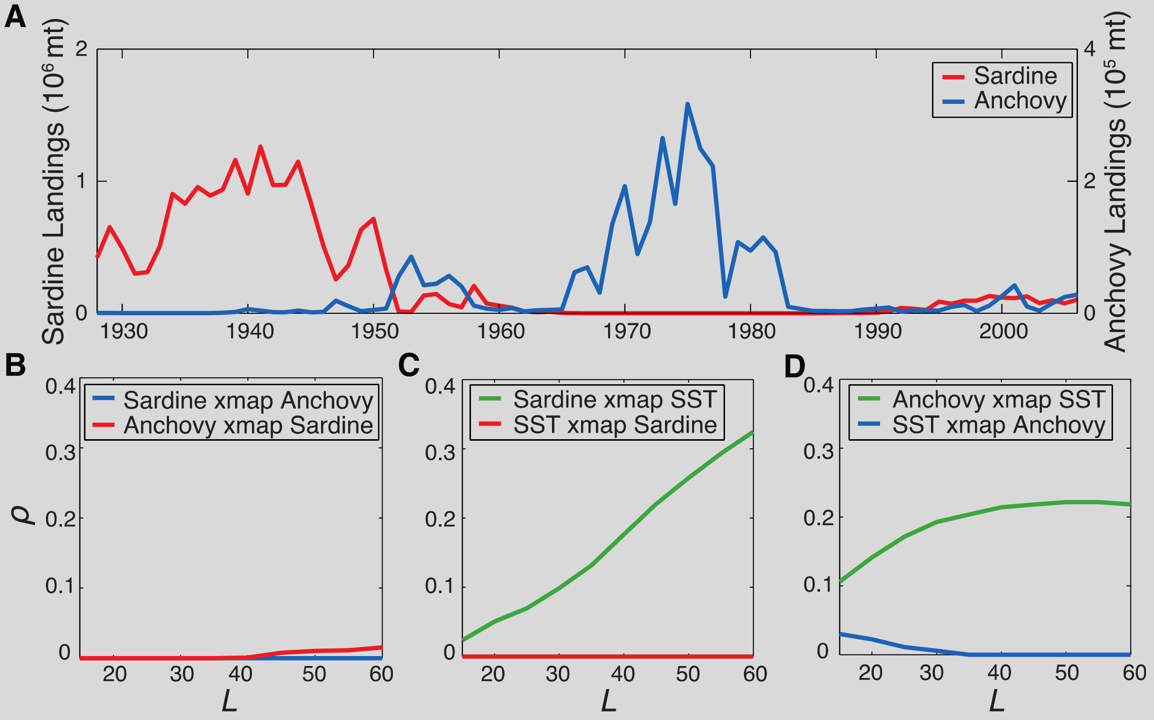
Although addressing nonlinear dynamics and complex ecosystems can appear to be daunting, new empirically-based methods that use time series data for reconstructing complex ecosystem dynamics may provide a practical simplification for understanding the role of interspecific interactions and environmental influences on population dynamics (Sugihara and May, 1990; Sugihara *et al.,* 2012). This empirical time-series-based paradigm emphasizes dynamics rather than static statistical relationships, and thereby avoids issues related to so-called “mirage correlations” that appear then disappear through time (Planque *et al.,* 2010) (Box 5.2). This approach is currently being examined by several NMFS science centers (SWFC, NEFSC, AFSC), and may provide guidance for a next-generation dynamic management paradigm based on forecasting production (Deyle *et al.,* 2013).

BOX 5.2
Mirage Correlation the Bane of Ecosystem Science

While we are all familiar with Berkeley's 1710 dictum "correlation does not imply causation." Less well appreciated is that in nonlinear systems the converse "causation does not imply correlation" is also true. Ecosystems are particularly perverse on this issue by exhibiting mirage correlations – associations among variables that spontaneously come and go or even switch sign (Sugihara *et al.,* 2012). This common behavior of nonlinear systems can create conceptual sand traps that distract research effort, continually causing us to rethink relationships we thought we understood. This is particularly relevant for investigating causative environmental factors (e.g., ocean temperatures) affecting fish production.

As a case in point, the alternation of Pacific sardine, *Sardinops sagax*, and northern anchovy, *Engraulis mordax*, in the California Current ecosystem is perhaps the most classic example of trying to understand ecosystem effects on pelagic fish populations. Jacobson and MacCall (1995) found a significant relationship between sardine recruitment success and sea surface temperature (SST) using a generalized additive model (GAM). Based on this relationship, the Pacific Fishery Management Council modified the sardine management plan to afford extra protection when SST is unfavorable (PFMC, 1998). However, a weakness of the GAM approach is that it does not readily accommodate the interacting effects of explanatory variables. The state-dependence of recruitment-environment relationships suggests that methods besides static linear correlation analysis are required.

Put simply, the problem has been the use of *static* linear methods to investigate a nonlinear *dynamical* system. In trying to understand environmental factors driving nonlinear fishery ecosystems we are interested in how variables affect each other dynamically (causally). Following ideas presented in Box 5.1, convergent cross mapping (CCM) is a recent tool that leverages the idea of Takens theorem that variables in a dynamic system share information about each other (Sugihara *et al.,* 2012). Thus, if two variables are dynamically connected (influencing each other's time series) it is possible to predict states of one from the other, and CCM tests for this. In the sardine-anchovy example, application of CCM showed that SST affected both sardine and anchovy, but there was no interaction between sardine and anchovy (Box 5.2, Figure 1).



Box 5.2, Figure 1. Detecting dynamic causation in a pelagic fishery. California landings of Pacific sardine (*Sardinos sagax*) and northern anchovy (*Engraulis mordax*) (A), and Convergent Cross Mapping (CCM) of (or lack thereof) of sardine versus anchovy (B), sardine versus SST (Scripps Pier) (C), and anchovy versus SST (Newport Pier) (D). The ability of CCM to predict the states of the forcing variable from the manifold of the forced variable is measured by the correlation coefficient between the predicted and observed states (ρ), which increases with time series length *L,* as points in the manifold become closer. This analysis shows that sardines and anchovies do not interact with each other and that both are forced by temperature. Reproduced from Sugihara *et al.* (2012). “Sardine xmap Anchovy” means cross mapping Anchovy from Sardine, etc.

Climate changes and shifting baselines

Environmental variability affects fish population dynamics on temporal scales ranging from interannual to decadal and millennial (Cushing, 1982). According to the National Standard 1 Guidelines (NS1G), “If environmental changes affect the long-term reproductive potential of the stock or stock complex, one or more components of the SDC must be respecified.”[4] The guidelines require a high standard for changing SDC so that this provision will not undermine the statutes that mandate ending overfishing.

As an example of changing reproductive potential, recruitment of Atlantic croaker (*Micropogonias undulatus*) depends on over-wintering temperature in the estuaries where juveniles rear. Hare *et al.* (2010) fit a temperature-dependent stock-recruitment model to data for Atlantic croaker that greatly reduces the unexplained recruitment variability (Figure 5.1). This statistical relationship allows biological reference points such as B_{MSY}

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[4] 50 C.F.R. 600.310 (e)(2)(iii)(B) (2012).

to be estimated with more precision, as well as levels of population abundance and sustainable harvest to be projected under assumed future temperature conditions. Similar models with environment-dependent stock-recruitment relationships are being formulated for species with rebuilding plans, such as winter flounder, *Psuedopleuronectes americanus* (NEFSC, 2011). However, these models may fail under environmental conditions other than those used to fit the model; alternate approaches may be needed (Box 5.2).

“Long-term environmental changes affect both the short-term size of the stock or stock complex and the long-term reproductive potential of the stock or stock complex.”[5] With climate change, environmental conditions into the future are assumed to change in a smooth progression from decade to decade. A common response of coastal and marine finfish species to climate change is a shift in their geographic distribution so they maintain themselves in preferred temperature conditions. Fish species may shift their geographic distributions poleward or to deeper water (Nye *et al.*, 2009). As species distributions shift, it may become necessary to change stock boundaries and definitions.

Biological reference points such as B_{MSY}, which are based on demographic parameters, become moving targets with changing climate. Productivity may increase as in the Atlantic croaker example, or it may decrease as seems to be the case of winter flounder at its southern range. The effect of climate is expected in cases such as winter flounder, but the relationship between climate and productivity is not well enough

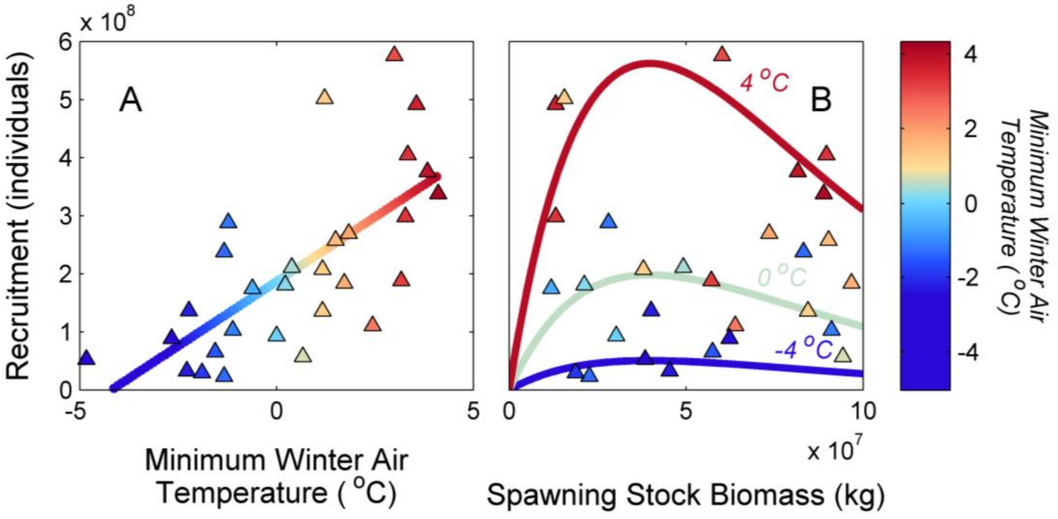

FIGURE 5.1: Relationship between Atlantic croaker (*Micropogonias undulatus*) recruitment, temperature, and spawning stock size. **A.** Relationship between recruitment and minimum winter air temperature in Virginia, USA. **B.** Environment-dependent stock–recruitment relationship illustrated at three temperature levels, -4, 0, and +4ºC. Reproduced with permission from Hare *et al.* (2010).

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[5] 50 C.F.R. 600.310(e)(2)(iii)

understood to make the types of long-term projections that are required for rebuilding programs (NEFSC, 2011). However, with better understanding of environmental trends, and the coupling of the environment and fishing to stock production it is possible to evaluate management strategies under climate-change scenarios (e.g., see Ianelli *et al.*, 2011; Punt, 2011).

A primary determinant of productivity in marine fish populations is the per capita recruitment rate (Myers *et al.*, 1999). The ratio of recruits per spawner can therefore be used to identify persistent shifts in productivity. Acadian redfish (*Sebastes faciatus*) had a period of increased productivity from 1980 to 2000, which contributed to its rebuilding (Appendix C, Figure.1). In contrast, five stocks from the New England and Mid-Atlantic regions experienced persistent downward shifts in recruits per spawner during the 1990s and 2000s: Mid-Atlantic black sea bass, Gulf of Maine haddock, scup, summer flounder, and Southern New England/Mid Atlantic (SNE/MA) winter flounder (Appendix C). These shifts help to explain the declining biomass of Gulf of Maine haddock (Figure 3.18) and lack of rebuilding of SNE/MA winter flounder (Appendix C, Figure 58), despite the target fishing mortality (F_{ACL}) being reduced to as close as possible to zero since 2009, resulting in $F_{\mathrm{ACT}} = 0.24\ F_{\mathrm{MSY}}$ in 2012 (Figure 3.7). The classification of stock status as overfished or not, and in turn the requirement for a Rebuilding Plan, is uncertain when there is an appearance of a change in productivity regime, as exemplified by the recent change in status of the SNE/MA yellowtail flounder (see Chapter 4). The per capita recruitment of SNE/MA yellowtail flounder has varied around its median level, apart from the very strong 1987 year class (Appendix C, Figure 63). The decline in recruitment of this stock was mainly due to low mature biomass, not a shift in productivity. This interpretation does not support the recent decision to lower the rebuilding target for this stock by favoring a hypothesis of regime shift (see Chapter 4).

Decadal regime shifts occur on time frames intermediate between short-term (i.e., internannual) and longer-term changes. MSY-based reference points and rebuilding targets can be specified for the current regime, while recognizing the possibility that the ecosystem may switch to a different regime within the time period covered by the Rebuilding Plan. The challenge with regime shifts is that they are difficult predict or characterize beyond recognizing that some rapid large-scale, system-wide change occurred. Alaska crab stocks were at high abundance levels during the 1960s and 70s, and then suffered steep declines in the early 1980s (Kruse *et al.*, 2010). While overharvesting contributed to these declines, several stocks have not recovered, even with low or no harvesting. Following the implementation of Rebuilding Plans in 2000, the St. Matthew blue king crab stock recovered beyond B_{MSY}, but the Pribilof Islands blue king crab stock has not rebuilt even though all fishery-related mortality has been essentially zero for a decade (Figure 5.2). While the mechanisms for lack of rebuilding are unclear (temperature, predation, lack of large males), the Bering Sea and the Gulf of Alaska appear to have shifted from a regime conducive to crab productivity to a regime more favorable to walleye pollock and other groundfish species.

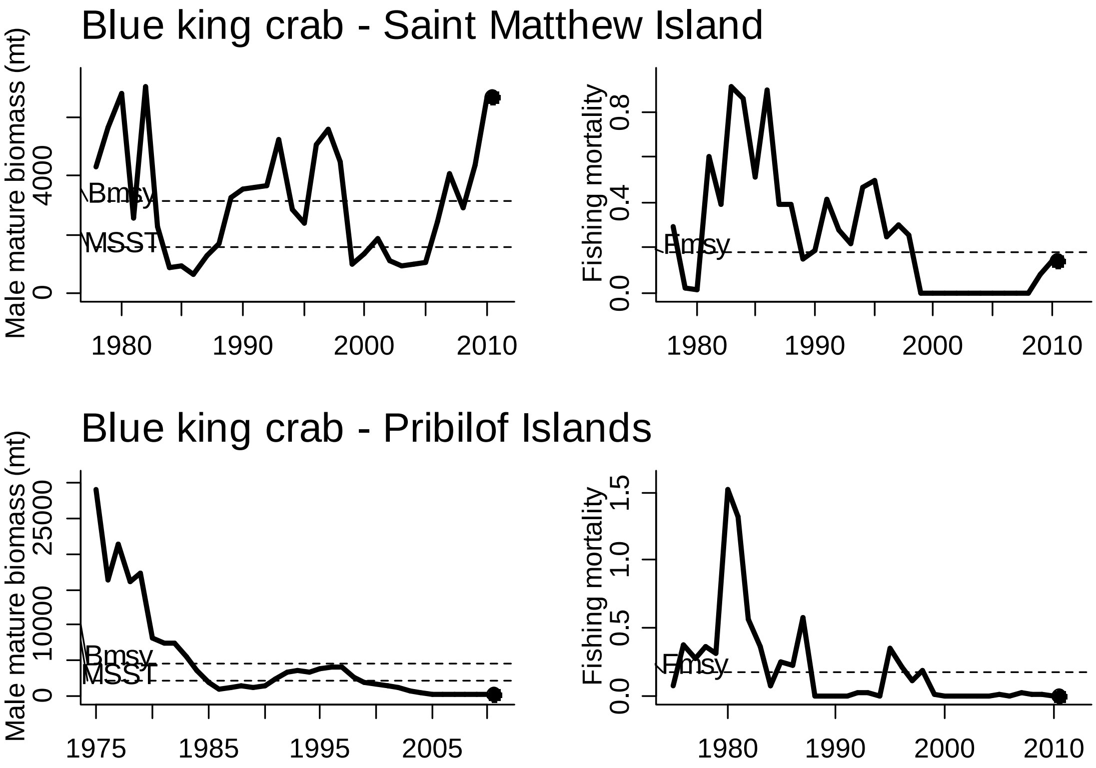

FIGURE 5.2: Biomass and fishing mortality rate of two blue king crab stocks in the Bering Sea: St. Matthew Island and the Pribilof Islands. (From Appendix C, Figures 10 and 11)

Regime shifts in marine fish stocks appear to be quite common (Vert Pre *et al.*, 2013). Such regime shifts create a dilemma for developing and assessing the performance of Rebuilding Plans. Conditioning rebuilding targets on an earlier, high-abundance regime could make rebuilding unattainable under the new prevailing environmental conditions. This scenario can lead to reductions in F well below F_{MSY} to reach a target that is too high to meet in a fixed time frame, and to constraints on other species in a mixed-stock fishery (see Chapter 4). Conversely, species rebuilding targets based on a recent period of low productivity could forgo larger potential harvests if stocks could, in fact, rebuild to their earlier high abundance levels. An additional challenge is to know whether hysteresis or delayed recovery may be operating. Though progress is being made in detecting regime shifts with theoretical models (Scheffer *et al.*, 2009) and experimental lakes (Carpenter and Brock, 2006), it is extremely difficult in practice with relatively short time series of empirical data.

Harvest strategies are needed that perform reasonably well under the alternative hypotheses considered plausible. Constant harvest rate policies, in which the harvest rate corresponds to the average productivity, have been shown to perform well in some simulated cases by allowing biomass to track changes in productivity (Parma, 1990; Walters and Parma, 1996). However, such policies may increase risks in the face of persistent low-productivity regimes. The commonly used "hockey stick" control rules respond to regime shifts by reducing fishing mortality at low biomass levels (Spencer and

Collie, 1997). Alternatively, the harvest rate can be adjusted dynamically in response to measured changes in stock productivity (Collie *et al.*, 2012). Harvest control rules that account for regime shifts in recruitment have been investigated for some species that have had rebuilding plans, such as snow crab in the eastern Bering Sea (Szuwalski and Punt, 2013).

While it is common to attribute population declines exclusively to fishing or exclusively to the environment, in most cases, the observed stock dynamics are probably a combination of fishing and the environment. Furthermore, fishing and environmental effects may interact in ways that are non-additive (Hsieh *et al.*, 2008; Deyle *et al.*, 2013). Planque *et al.* (2010) reviewed many ways in which fishing can alter the sensitivity of marine populations and ecosystems to climate. Among these, the alteration of demographic structure is most relevant to stock rebuilding because depleted stocks are likely to have truncated age structure and reduced genetic population structure (Olsen *et al.*, 2004). Populations with truncated age structure are more variable because they are measurably more nonlinear (express greater dependence on ecosystem state) and are more dependent on recruiting age classes (Anderson *et al.*, 2008).

Truncating the age structure may reduce the ability of populations to cope with sequences of poor conditions. A possible consequence of fishing-induced change in age structure may be an increased coupling between recruitment and environmental conditions. Support for this hypothesis comes from the Northeast Arctic cod, for which the correlation between recruitment and ocean temperature strengthened as the population declined and the modal age of the spawning biomass declined from 13+ to age 7 (Otterson *et al.*, 2006). An important implication of this hypothesis for overfished stocks is that, while recruitment may seem to depend only on the chance occurrence of favorable environmental conditions, rebuilding the demographic structure of the population will increase the probability of strong recruitment. More generally, Planque *et al.* (2010) concluded that:

> "*If, as it is agued here, exploitation can affect the way populations respond to climatic forcing, it is likely that recovery to a given population abundance or biomass will not be sufficient to also restore the patterns of population responses to climate. This will require that population characteristics other than biomass (e.g. demographic and spatial structures) also be restored. Another consequence is that statistical climate-population relationships may display recurrent appearance/disappearance sequences, as has also been observed for a number of populations*." (see Box 5.2).

Thus, stocks needing rebuilding may become more sensitive to environmental variation, exhibiting more nonlinear state dependence and variability (Anderson *et al.*, 2008). This sensitivity could affect the rate at which a population can recover, and increased variability may obscure the success of rebuilding. Attaining a biomass target may depend on first restoring the age structure of the stock. For example, the age structure of Georges Bank haddock became truncated following the stock collapse between 1970 and 1995 (Figure 5.3). The expansion of the age structure in the late 1990s preceded very strong year classes in 1999 and 2004.

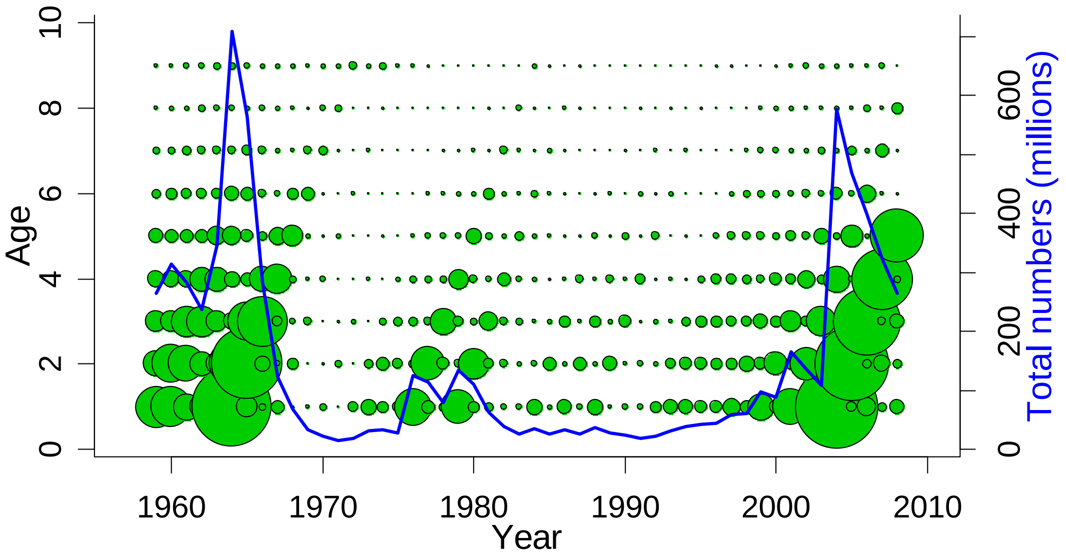

FIGURE 5.3: Age structure of the Georges Bank haddock stock shown in relation to total population size (solid line). The area of each bubble is proportional to the numbers at each age in each year from 1959 to 2008. The numbers were estimated with an age-structured stock assessment (NEFSC, 2008).

Habitat loss and carrying capacity

The importance of fish habitat was recognized in the 1996 Sustainable Fisheries Act (see Chapter 2). This reauthorization of the MSFMCA defined "essential fish habitat" (EFH) as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity."[6] The regional fishery management councils were tasked by October 1998 to: describe and identify EFH for each managed species; and to minimize to the extent practicable adverse effects on EFH. In implementing this act, the guidance for National Standard 1 specified that "if manmade environmental changes are partially responsible for a stock or stock complex being in an overfished condition, in addition to controlling fishing mortality, Councils should recommend restoration of habitat and other ameliorative programs to the extent possible."[7]

The habitat requirements of purely marine fish species have been catalogued through EFH amendments to fisheries management plans. For many of the most important commercially harvested species, EFH is defined by generalized habitat types, benthic substrates, and depth ranges of the different life stages, but rarely indicates specific geographic locations. At a fundamental level, carrying capacity may be limited by available habitat, but it is difficult to tell which habitat attributes are limiting for most species. For example, red snapper in the Gulf of Mexico has a strong association with vertical structures, such as oil and gas rigs; yet, the degree to which the addition of such

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[6] Sustainable Fisheries Act, Pub. L. No. 104-297 § 102(3), 110 Stat. 3559, 3561 (1996).

[7] 50 C.F.R. 600.310 (e)(2)(iii)(C) (2009).

habitat increases stock productivity or simply spatially redistributes stock biomass remains controversial (Shipp and Bortone, 2009).

Habitat considerations for certain life stages have been well defined because of known reliance of the species on specific habitats and the documented degradation of these habitats. Impacts on freshwater habitat are known to limit the recovery of anadromous species such as Pacific salmon. In the western U.S., for example, dams limit upstream migration, eliminating entire watersheds from salmon production. Altered flow regimes affect the thermal habitat, while logging and channelization of streams affect spawning and rearing habitat (National Research Council, 2004a). In the northeast U.S., dam removal on the Kennebec River and the installation of fishways on the Acushnet River have restored habitat to anadromous fish (RAE, 2013). Counts of river herring (*Alosa* spp.) have increased in these rivers since these fish passage improvements.

Habitat changes can also occur on broader spatial scales, and simultaneously affect multiple species. For example, the coastal salt marsh habitat of Louisiana is well known to support the early life stages of many of the Gulf of Mexico's commercially important stocks (Boesch and Turner, 1984). Coastal wetlands have been lost at a rate between 60 and 100 km^2/year, reducing juvenile habitat and likely causing reductions in concomitant productivity of many species (Boesch *et al.,* 1994; Day *et al.,* 2005). There are plans to restore the wetlands of coastal Louisiana (Boesch and Turner, 1984). Restoration would occur over similar multi-decadal time frames as the stock rebuilding plans for long-lived species, and over a broad spatial area. Successful restoration could affect the available habitat for multiple species and thus their productivity. In a study of salt marsh restoration in Delaware Bay, the intertidal fish community composition converged with that in reference marshes within the nine-year study period (Able *et al.,* 2008).

Similarly, various benthic habitats on the continental shelves are essential for feeding and reproduction of exploited species. Changes in the structure, function, or aerial extent of these benthic habitats may affect rebuilding rates. Mobile bottom-fishing gear exerts pervasive effects on benthic habitats on the continental shelf (Collie *et al.,* 2000). The disturbance to benthic communities caused by trawling and dredging has been quantified in numerous studies (National Research Council, 2002a), and is known to vary with depth, sediment type, and taxa (Kaiser *et al.,* 2006). Bottom fishing can reduce the production of benthic communities (Jennings *et al.,* 2001; Hermsen *et al.,* 2003). Differences in diet composition have been observed in demersal fish species in areas with contrasting levels of bottom fishing disturbance (Smith *et al.,* 2013). Finally, differences in growth rate of plaice, *Pleuronectes platessa,* were found across a gradient of fishing disturbance in the Celtic Sea (Shepherd *et al.,* 2010). These studies suggest that benthic habitat may limit fish feeding and growth, but results have been mixed, making them difficult to scale to the population level.

Some sensitive habitats have been designated Habitat Areas of Particular Concern (HAPCs) and afforded protection from bottom fishing disturbance, for example in the case of the juveniles of demersal species such as Atlantic cod. As management tools, closed areas protect fish habitat as well as fish stocks (see Chapter 4). For example, increases in the abundance, biomass, and production of benthic epifauna occurred following large area closures on Georges Bank (Collie *et al.,* 2005). The time scales of recovery for habitats range from years in soft sediments to decades and even centuries for hard substrates (National Research Council, 2002a).

Some environmental and climate effects on fish populations are likely to be mediated through habitat. Climate variability alters fish reproductive habitat (see discussion on climate changes and shifting baselines above). For example, the cumulative spawning habitat available for sardine and anchovy was evaluated across the California Current by relating shipboard collection with remote-data series of sea surface temperature (Reiss *et al.*, 2008). On the northeast shelf, intensity of the fall phytoplankton bloom has been hypothesized to stimulate benthic productivity and thereby the reproductive contribution and recruitment success of Georges Bank haddock (Friedland *et al.*, 2008). This mechanism may have played a role in rebuilding this haddock stock. The ranges and reproductive habitats of many species are likely to shift with climate change, and such shifts may be amplified by fishing (Hsieh *et al.*, 2008). These shifts include range expansions at the northern edge of species distributions (e.g., summer flounder juveniles rearing in northern estuaries) and contractions at the southern edge of species distributions (e.g., winter flounder declining in Delaware Bay).

In summary, many species depend on particular habitats to support the growth and survival of specific life stages, suggesting that habitat loss could limit rates of rebuilding. However, in most cases, the relationships between habitat and productivity have not been quantified. If habitat recovers quickly, it might accelerate stock recovery and increase the success of short-term rebuilding plans. If recovery is long term, habitat recovery may be more important for gradually increasing the carrying capacity of the stock.

Ecological interactions

In what ways do ecological interactions need to be considered when formulating Rebuilding Plans? The total productivity of a fish community is ultimately limited by production at lower trophic levels. Food-web models implicitly assume that consumers compete for limited food resources (Collie, 2001). When fish stocks are depleted, their prey species are consumed by other predators that may increase in abundance, thereby limiting availability of the common prey. It may then be difficult to simultaneously rebuild all overfished species to their single-species B_{MSY} levels without reductions in other consumer species. Food-web models of the Georges Bank fish community suggest that rebuilding the principal groundfish species (cod, haddock, yellowtail flounder) to their B_{MSY} levels would require restructuring of the fish community, and repartitioning of energy within the food web (Collie *et al.*, 2009; Link *et al.*, 2011).

Population growth can be limited by prey abundance at critical life stages. Recent work has related population growth of Atlantic cod to trends in zooplankton abundance. In particular, two copepod taxa, *Pseudocalanus* spp. and *Centropages typicus*, which are nutritionally important for larval cod, have declined in spatially discrete areas where cod populations have not responded to stock rebuilding measures (Friedland *et al*., 2013). The reproductive condition of medium sized cod (30-69 cm) off Newfoundland and Labrador was related to diet by Sherwood *et al*. (2007). Cod off southern Newfoundland with more pelagic diets had higher somatic condition, lipid stores, and spawning potential than more northerly cod, which preyed almost entirely on shrimp (*Pandalus borealis*). The authors hypothesized that rebuilding Newfoundland and Labrador cod stocks will require a return to a food web in which cod feed mainly on pelagic species such as capelin (*Mallotus villosus*).

In a multispecies context, biological reference points should vary with changes in growth, maturity, and especially natural mortality (Collie and Gislason, 2001). The estimation of reference points, including F_{MSY} and B_{MSY}, depends on the demographic parameters of the rebuilding species, which in turn depend on the dynamics of other species in the food web. In particular, the reference points of harvested prey species are conditional on the abundance of their predator species. The level of predation could be a factor in the rebuilding of prey species such as butterfish in the mid Atlantic and crab stocks in Alaska. Predation mortality can be incorporated in stock assessments implicitly with a time-varying natural mortality rate, or explicitly with a dynamic multi-species model (Hollowed *et al.,* 2000). To date rebuilding analyses have not incorporated time-varying natural mortality nor the unstable dynamics that this could produce.

Species that are predators as adults experience predation on their early life stages. Predation by clupeid species (such as herring) is hypothesized to be a substantial source of mortality on the eggs and larvae of gadid (cod) species (Daan *et al.,* 1985). This predator-prey feedback can lead to alternate clupeid- or gadid-dominated states. A meta-analysis of cod-herring interactions indicated negative effects of herring on cod recruitment for several stocks in the North Atlantic (Minto and Worm, 2012). When embedded in a length-based model of the Georges Bank fish community, this predation effect delayed, but did not prevent, cod rebuilding (Collie *et al.,* 2013). These, and similar results for the North Sea (Speirs *et al.,* 2010), suggest that, although such predation triangles may exist, they do not necessarily result in depensation that would prevent cod stocks from rebuilding. A detailed analysis of Atlantic cod on the eastern Scotian Shelf found no evidence that rebuilding was delayed by a high biomass of forage species that could prey on cod eggs and larvae; rather the lack of rebuilding was attributed to high natural mortality at the adult stages (Swain and Mohn, 2012).

The community aspects of rebuilding have been investigated with size-based models. Rebuilding a target species can have indirect predation effects on smaller species and on the juveniles of larger species (Andersen and Rice, 2010). Overfished species rebuild at different rates, altering the predator-prey dynamics compared with the unexploited fish community (Collie *et al.,* 2013). In simulations, prey species released from predation rapidly increased and overshot their unexploited equilibrium level. In contrast, large predator species increased slowly, failing to recover after 25 years of simulation. Again, the delayed rebuilding of predator species was not due to depensation, but suggests hysteresis in community rebuilding.

Although some species subject to rebuilding are currently managed as stock complexes (e.g., Caribbean grouper, Hancock Seamount groundfish complex), most Rebuilding Plans are for single species and fishery management plans require species-specific annual catch limits. An ecosystem perspective might consider aggregate species groups instead of rebuilding on a stock-by-stock basis. Aggregate production models have been used to estimate sustainable yield at the community level (Brown *et al.,* 1976;

Mueter and Megrey, 2006), and can be used to set a cap on the total allowable catch (TAC) across multiple species. On one hand, there is evidence of compensation within functional groups (Auster and Link, 2009), so functional groups based on diet similarity would be the units of management; depletion of one species could be offset by increases in another species in the same functional group. On the other hand, there is considerable overlap among functional groups, however they are defined. Most functional groups are dominated by a few species, such that the functional-group dynamics simply reflect the dynamics of the dominant species in the functional group.

Incorporating ecosystem considerations into Rebuilding Plans

A general conclusion is that Rebuilding Plans should consider the structure and functioning of populations and ecosystems in a wider sense to maximize the ability of fish populations to rebuild. Rebuilding Plans should ideally entertain a broader spectrum of ecosystem dynamics and possible outcomes than is typically considered in single-species rebuilding projections, particularly in view with what is currently known about the prevalence of such dynamics in nature. Reductionist approaches that try to separate the effects of fishing and the environment may overlook important interactions

Biological reference points based on MSY (and its proxies) are moving targets because of the complex and dynamic nature of ecosystems. Reference points, or possibly other performance criteria, should be sought that are appropriate for the observed dynamics, can accommodate ecosystem changes, and not have unintended consequences for rebuilding. If a reference point or formula is adjusted for prevailing ecological conditions, it should aim to reduce fishing mortality when productivity declines. As a result of ecosystem dynamics, including biological competition and predation, fisheries management involves tradeoffs among harvested species, even if conscious decisions are not made about the tradeoffs or they cannot be predicted with confidence.

What is possible given the present level of scientific understanding? We know many of the mechanisms that make ecosystems dynamic and our understanding is advancing rapidly. Environmental variables can be included in the estimation of reference levels where they are known to affect demographic parameters (e.g., weight-at-age, maturation, fecundity, recruitment) of the species. Some of the better-understood ecosystem considerations can be incorporated into management strategy evaluations (MSE) of rebuilding (Punt, 2011). Multispecies models of intermediate complexity can be statistically fit to time-series data of interacting species (Plagányi *et al.*, 2012). A likely consequence of applying these approaches is to increase the uncertainty in stock rebuilding projections.

In most cases, we do not yet now know enough to predict the future state or to manipulate ecosystems to achieve desired tradeoffs among species (even if there were agreement on which tradeoffs are desirable). Therefore, there need to be practical, operational, and robust management strategies for fisheries rebuilding (see Chapter 7 for additional discussion). Meta analyses and MSE are probably key tools to advance these methods. Analysts can embrace multiple working hypotheses and integrate performance outcomes across hypotheses, weighted by the probability of each hypothesis being true.

Other poorly understood ecosystem considerations can only be accounted for in a qualitative sense.

Stock rebuilding can proceed without full understanding of ecosystem dynamics. A previous NRC committee (National Research Council, 1999a, p. 5) concluded that "... a significant overall reduction in fishing mortality (*F*) is the most comprehensive and immediate ecosystem-based approach to rebuilding and sustaining fisheries and marine ecosystems." Since that report, there has been a significant overall reduction in *F* as indicated by a reduction in proportion of stocks suffering overfishing (Chapter 3). A better understanding of the dynamics of depleted fish stocks depends on the continuation of existing data collection programs, as many analytic methods are constrained by short time series. Process-oriented studies are needed to elucidate the interactions between fish and their environment (e.g., the dependence of fish production on habitat).

Some ecosystem considerations may imply longer rebuilding times or require lower fishing mortality rates for rebuilding. Conversely, favorable environmental conditions can reduce rebuilding times. Either way, ecosystem considerations should not be used as excuses for inaction. They do not contradict a tenet of fisheries science, that harvested stocks have finite capacity to compensate for increased mortality, but they do supplement and extend it. Most fish stocks can rebuild when fishing mortality is reduced (see Chapter 3). If population dynamics are non-linear, as recent studies suggest, fishing mortality may need to be reduced below a threshold level to initiate rebuilding (Collie *et al.,* 2004).

Stocks with episodic recruitment such as rockfish off the West Coast pose a special challenge because of the difficulty of distinguishing regime shifts from delayed rebuilding. However, the appropriate management action in both cases to rebuild a depleted stock is to reduce fishing mortality below F_{MSY}. The stock should eventually rebuild if there is episodic recruitment. If the stock has shifted to a lower productivity regime, it can still be sustainably harvested with the lower fishing mortality rate, but it may not rebuild to biomass targets defined under past conditions. In the case of a regime shift, the lack of recovery would not be considered a management failure. This example leads to a more general conclusion that ecosystem considerations can temper our expectations about the levels to which stocks can rebuild and the time it takes to get there. However they do not alter the general prescription of reducing fishing mortality on depleted stocks.

FINDINGS

5.1: Ecosystem considerations imply a broader spectrum of population dynamics and possible outcomes than is typically considered in single-species rebuilding projections. Stock biomass estimates and projections can vary greatly in response to alternative plausible assumptions (models) and parameter values used in simulations, because the underlying population dynamics are nonlinear.

5.2: With climate change, and because of the complex and dynamic nature of ecosystems, biological reference points (such as B_{MSY}) based on single-species production functions are likely to change over time.

5.3: Fishing and the environment interact in ways that are non-additive. Fishing-induced age truncation amplifies the effect of recruitment variability on population dynamics, and it may increase coupling between recruitment and environmental conditions leading to more variable recruitment. An important implication of this hypothesis for overfished stocks is that rebuilding the demographic structure of the population will increase the probability of strong recruitment.

5.4: Habitat loss and degradation may limit reproduction, feeding and growth, but studies have been inconclusive and it is difficult to scale results to the population level. For most species, the relationship between habitat and productivity remains unquantified. Process-oriented studies can elucidate the interactions between fish and their environment (e.g., the dependence of fish production on habitat).

5.5: In a multispecies context, growth, maturity, and natural mortality are influenced by the abundance of interacting species, although these effects are difficult to predict. Biological reference points for forage species may need to allow a larger proportion of the production of these species to be available to predators than for higher trophic-level species.

5.6: To address species interactions (5.5), stock assessments have been linked in a number of ways. Multispecies models can inform the natural mortality (predation) rates used in single-species assessments of prey species (e.g., herring, menhaden, walleye pollock). For species that are both predator and prey, incorporation of species interactions requires a dynamic multispecies model (e.g. age-structured multispecies models that have been developed for several ecosystems).

5.7: As a result of ecosystem dynamics, such as biological competition and predator-prey interactions, fisheries management involves tradeoffs between harvested species, even if tradeoffs are not deliberate decisions and outcomes are often unpredictable.

5.8: Scientific understanding of ecosystem dynamics is advancing rapidly. In some cases, understanding has advanced enough to model dynamics, which may be used to inform fisheries management decisions. However, in most cases, scientific understanding of ecosystem dynamics is insufficient to confidently predict the future state or to achieve desired tradeoffs among species (even if there were agreement on which tradeoffs are desirable). These cases depend on having pragmatic, operational management strategies that acknowledge this kind of uncertainty.

5.9: Ecosystem considerations, among other reasons, argue for more emphasis on rebuilding plans that maintain reduced fishing mortality for an extended period (e.g., longer than the mean generation time). This strategy rebuilds age structure and is more *robust than a focus on biomass targets, which may be more or less attainable depending on environmental conditions.*

6

HUMAN DIMENSIONS OF REBUILDING

Introduction

Rebuilding fish stocks is guided by §304(e) of the Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA),[1] the National Marine Fisheries Service (NMFS) guidance on Magnuson implementation, and judicial review. The declaration of a fish stock as overfished triggers immediate, prescribed remedial actions. The primacy of conservation and the secondary role of socioeconomic factors in this process reflect purposeful tradeoffs expressed by collective legislative, executive, and judicial input in U.S. fishery governance. At the same time, there are many experiences around the globe highlighting the social and economic impacts that can accompany declarations of overfished fish stocks.[2] What has garnered less attention is how social and economic factors can be utilized in the design of management actions and contribute to their efficacy, in some cases enabling rebuilding to be achieved with greater net social benefits.

There are multiple ways to assess the performance of a policy and its implementation, including the rebuilding of fish stocks. Stakeholders may have different perspectives on what performance and outcomes are considered successful. Among the primary motivations for rebuilding is an expectation that rebuilt fisheries will lead to healthier ecosystems and greater sustainable social and economic benefits (OECD 2010). Yet while the natural and human outcomes of fish stock rebuilding are often closely aligned, they are not necessarily so; rebuilding of a fished stock does not imply parallel effects on fisheries or social benefits. For example, rebuilding of a fish stock to a given biological benchmark (say, to B_{MSY}) can be associated with both long-run positive gains and short-run negative social and economic consequences. Whether these long-run gains offset the short-run costs depends on numerous factors including how the rebuilding actions are instituted, the characteristics of the fishery, and the assumed discount rate. Moreover, stock sizes that maximize expected economic (net) benefits are almost always different from B_{MSY} (Clark, 1990; Hilborn *et al.*, 2012). Further, the long-term gains may not be realized in the same segments of the industry that bore the short-run costs, and the socioeconomic transition that occurs during rebuilding (e.g., restructuring in the fleet and industry) may not be fully reversible, although the social science research on the nature of the socioeconomic transition is incomplete.

In general, the success of fisheries management and policy implementation in rebuilding fish stocks depends on how individuals and institutions respond (e.g., changes in fishing practices, compliance with rules, coping with social and economic transitions, etc.). Fishery

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[1] Magnuson-Stevens Fishery Conservation and Management Act § 304(e), 16 U.S.C. § 1854(e) (2012).

[2] For example, in the U.S., some of the more significant overfished declarations have been in the New England groundfish fisheries, Gulf of Mexico reef-fish complex, and in the West Coast groundfish fishery. Globally, examples include the catch of groundfish in the European Union, sea cucumber and rock lobster in the Galapagos, and nearshore fisheries in Chile.

managers manage people not fish. Because of the complexity and imperfect knowledge of the coupled human-natural systems (Liu *et al.,* 2007) that constitute fishery complexes, the ex post social and economic outcomes from a rebuilding plan can diverge from expectations.

Understanding the drivers of human behavior, the role of institutions, the past impacts of management actions, and the potential future impacts from a suite of management actions on social and economic systems is the domain of the social sciences (NOAA Science Advisory Board, 2009). It is across this broad domain that this chapter considers the social and economic dimensions of fishery Rebuilding Plans.

Many of the findings identified and discussed throughout the chapter are in part a consequence of the well-documented limitations associated with social science funding, staffing, and data collection under which NMFS (and all of NOAA) operates (e.g., NOAA Science Advisory Board, 2009). These resource constraints lead to differing approaches across the Regional Fishery Management Councils (henceforth, Councils) in preparing Fishery Management Plans (FMPs), amendments, and the National Environmental Policy Act (NEPA) documentation (EPA, 2005). Some Councils have in-house capacity to draft FMPs and supporting documents, while others use FMP development teams composed of Council, NMFS Regional Offices, and Science Center staff and university scientists. With increasing calls for reductions in the size of government programs, the social science demands and expectations on fisheries management will likely continue to outpace available funding and staffing.

The chapter begins with an overview of the broader social and economic considerations in fish stock rebuilding, including biological versus social objectives, short-term versus long-term economic costs and benefits, and direct and indirect community impacts. This includes discussions of challenging issues such as mixed stocks, data-poor situations, scientific uncertainty and incomplete information. The chapter continues with two sequential sections addressing the methods and NMFS guidance for economic and social impact assessments. These sections assess whether the NMFS guidance is consistent with established approaches for analyzing economic and other social outcomes and tradeoffs, and by reviewing a sample of rebuilding FMPs, whether the economic and social impact reviews conducted in practice incorporate analysis of these outcomes and tradeoffs. The chapter concludes with a discussion of impacts of fisheries management tools on rebuilding effectiveness, followed by the findings of the committee's analysis.

The committee carefully considered the analytical approaches necessary to conduct a retrospective or post hoc analysis of economic and social consequences of implementing specific rebuilding plans. However, the resources needed to adequately do such an assessment were beyond the scope of the committee in large part because the necessary socioeconomic data do not exist. More systematic collection of socioeconomic data by NMFS would have permitted more in-depth analysis of the actual socioeconomic impact of specific rebuilding plans (see discussion in NOAA Science Advisory Board's 2009 report). The committee did not have the resources to do the original data collection and analysis for these fisheries, and thus the chapter focuses on direct and indirect community impacts reported in the literature.

Socioeconomic implications of rebuilding targets

Fish stock rebuilding as mandated by the MSFCMA is based on "a prescriptive approach with tight timelines and limited flexibility" (Khwaja and Cox, 2010), "designed to achieve rapid rebuilding of biomass and spawning stocks consistent with the biological characteristics of the resource" (Larkin *et al.,* 2007). Specific rebuilding parameters mandated by MSFCMA are determined based on the stock specific potential rate of building at the time a plan is developed and the allowable time period for rebuilding specified in the Act and Guidelines (see Chapters 2 and 3). Exceptions to these mandates are limited (e.g., cases of conflicting international agreements, incompatible biology of the fish stock, or other environmental conditions).[3]

Strict adherence to mandated biological rebuilding—despite possible socioeconomic tradeoffs and short-term costs (see Box 6.1)—are deemed necessary to "[end] overfishing immediately" and to prevent "protracted political debate, while the resource continues to decline" (Rosenberg *et al.*, 2006). The review of empirical and biological outcomes of mandated rebuilding plans in the U.S. presented in Chapter 3, as well as experience from other regions, support the view that biological mandates such as these may be linked to success in rebuilding depleted stocks (Caddy and Agnew, 2002; Khwaja and Cox, 2010). In addition, available evidence suggests that projected net economic benefits, or net present value of successful rebuilding are often positive in the long run (Gates, 2009; Hanna, 2010; Khwaja and Cox, 2010; Sumaila and Suatoni, 2006; Sumaila *et al.,* 2012; World Bank, 2009).

BOX 6.1
Socioeconomic Tradeoffs and Costs

An example of tradeoffs between projected short-term costs and long-term benefits of rebuilding is seen in the original analysis of alternatives conducted for Amendment 13 to the Northeast Multispecies Fishery Management Plan (NEFMC, 2003). The economic analysis quantifies net economic value realized under rebuilding alternatives for all groundfish stocks covered under Amendment 13, including Atlantic cod, haddock, and yellowtail flounder, from 2003 through 2026. Rebuilding alternatives are considered that were anticipated to achieve rebuilding by either 2009 or 2014. As shown in section 5.4.2.5 of NEFMC (2003), Comparison of Rebuilding Strategies for 2009 Rebuilding Time Frame for Most Stocks, the projected difference in discounted harvest revenue compared to the no-action alternative is negative for all rebuilding alternatives through 2009. The effect is then positive from 2010 through 2026. Cumulative net present values do not become positive until after 2021. Similar patterns are seen for rebuilding alternatives which aim to rebuild stocks by 2014. Figure 6.1 reproduces figure 207 from NEFMC (2003), illustrating the trajectory of projected cumulative economic benefits (the sum of discounted consumer benefit, income payments and owner profits) over time. As shown by these projections, the net present value of cumulative benefits is negative for all management alternatives that would achieve rebuilding for roughly the first 15 years, after which positive cumulative benefits are projected. Similar discussions of short versus long-term net benefits of rebuilding are found in other analyses of rebuilding alternatives (e.g., GMFMC, 2004). Hence, even when positive net benefits are projected over a long run rebuilding trajectory, these are typically preceded by negative net benefits in the short run.

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[3] Magnuson-Stevens Fishery Conservation and Management Act §§ 304(e)(4)(A)(i)- (ii), 16 U.S.C. §§ 1854(e)(4)(A)(i)-(2) (2012).

The NEFMC (2003) analysis also demonstrates that longer rebuilding periods can increase projected benefits. The analysts concluded that although the "2009 rebuilding time would result in lower landings than [the 2014 rebuilding alternatives] until 2009" there would be "higher landings from 2010 to 2014, and roughly equivalent landings from 2015 onward" (NEFMC, 2003). In all cases, cumulative net economic benefits are greater under the rebuilding alternatives which aim for rebuilding to occur in 2014.

The focus on biological mandates can preclude the discussion, analysis, and implementation of fishery management alternatives that could provide greater potential economic benefits across commercial and recreational sectors (Agar and Sutinen, 2004; Holland, 2010a; Larkin *et al.*, 2007; 2011) and reduce adverse community impacts. Some of the community impacts associated with fishery management, in general, include changes in health and safety (e.g., Georgianna and Shrader, 2008), well-being of fishery dependent communities (e.g., Hall-Arber *et al.*, 2001; Clay *et al.*, 2010), and infrastructure and waterfront land use (e.g., Portman, *et al.*, 2009). The commercial and recreational fishing industry and those representing fishing communities often contest Rebuilding Plans due to the perceived inflexibility of these Plans with regard to such impacts, as well as the potential short-term economic costs (Hanna, 2010; Terciero, 2011).

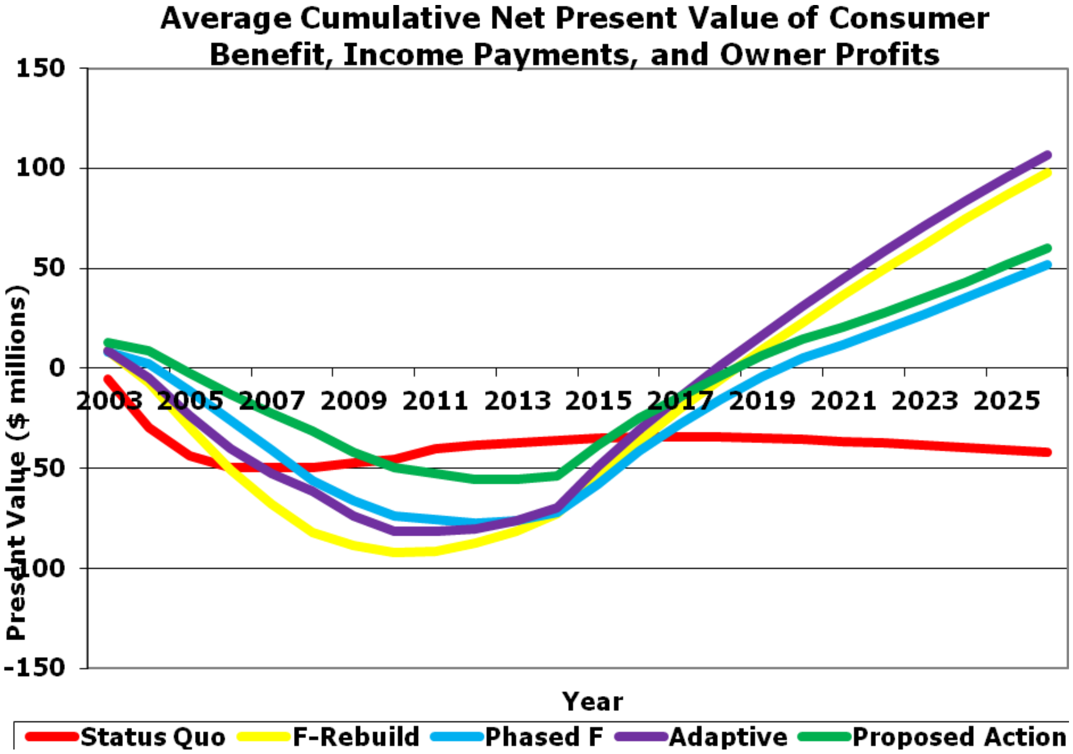

Figure 6.1. Economic net benefits for different rebuilding strategies (NEFMC, 2003).[4]

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[4] Methods for the economic analysis underlying Figure 6.1 are outlined by NEFMC (2003). Each rebuilding projection is based on a set of mortality rates designed to achieve biological rebuilding, with stock adjustments made to account for recreational and Canadian landings. Each net benefit trajectory represents the cumulative sum over time of consumer surplus, owner profits and returns to labor (or income payments), discounted at an annual rate of

In general, a fishery management strategy designed to maximize the economic benefits or minimize adverse community impacts (e.g., maintain cultural heritage, working waterfront industries, etc.) will diverge from those chosen according to biological criteria alone (Holland, 2010a; Larkin *et al.*, 2007, 2011; Da Rocha *et al.*, 2012; Hilborn *et al.*, 2012; see also Grafton *et al.*, 2007). As illustrated by Kompas *et al.* (2009), for example, pursuing MSY as a harvest target can "result in zero or even negative profits," whereas positive profits are possible if stocks are harvested at maximum economic yield (MEY). An analysis for the southeastern trawl fishery in Australia indicated that $B_{\mathrm{MEY}}/B_{\mathrm{MSY}}$ ranged from 1.10 (Spotted warehou) to 1.53 (Orange roughy in the Cascade), reflecting economic factors such as the influence of biomass on harvest costs that are not incorporated in biological models alone. Depending on fishery characteristics, the optimal harvest strategies from these socioeconomic analyses can either be more or less conservative (e.g., targeting higher or lower biomass) than those determined solely by maximizing sustainable yield (e.g., Clark, 1990; Grafton *et al.*, 2007; Johnston and Sutinen, 1996), although in general "long-term profitability is maximized at harvest rates lower than would produce MSY" (Hilborn *et al*. 2012).

While a fully-optimized strategy to maximize the socioeconomic benefits might not be feasible (see discussion later in the chapter), there are often potential socioeconomic gains from increasing the degree of flexibility to achieve a given target. For example, Larkin *et al*. (2007) found with respect to the 10-year rebuilding rule that extending the timeframe (as allowed in New Zealand for example) could result in significant economic gains depending on the economic and ecological characteristics of the fishery, and could better meet the needs of fishing communities. Larkin *et al*. (2007), contrasted alternative rebuilding scenarios for an illustrative moderate-lived fish stock and found that depending on the assumed discount rate,[5] expected net economic benefits increased between 3.5% and 19.4% when rebuilding timelines were extended from 10 to 20 or 30 years, and average TACs during the rebuilding period also increased between 46% and 97%.

An economic analysis as part of Amendment 13 in New England (see Box 6.1) also found that longer time horizons could increase projected benefits. However, extending the timeframe is not always the optimal economic plan (see, e.g., Sanchirico *et al.,* 2010) and is only

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7%. All projections are given in 2001 dollars. Illustrated net benefits are mean values over projected probability distributions calculated using Monte Carlo simulation, based on fitted theoretical probability distributions of age-based landings. Projections assume that that every fishing mortality target is achieved. Consumer surplus, owner profits and returns to labor are estimated based on results from a dockside demand model, specified as a system of price equations. These models capture both trends in seafood demand and supply/demand interactions. Consumer surplus estimates are calculated as the area under the demand function from zero to the quantity supplied (i.e., landings), after subtracting total vessel revenues. Return to labor and owner profit are calculated as the difference between gross revenues and fishing costs, with costs (fixed and operating) estimated from survey data.

[5] Because benefits and costs of regulations occur over multiple years, an analysis of the trade-offs between undertaking an action today and potential outcomes in the future needs to consider the time value of money (Goulder and Stavins 2002). Discounting, which is the method employed to make such a comparison, is analogous to a bank recognizing the time value of money by charging borrowers interest rates. A higher (lower) discount rate will place more (less) weight on benefits and costs in the present relative to the future (see Holland *et al*. 2010d and USEPA 2000 for a more detailed discussion). OMB Circular No. A-94 provides guidelines for discount rates to be used within cost benefit analysis of federal programs.

one dimension over which flexibility could be introduced into Rebuilding Plans (see discussion on fishery management later in this chapter). Even in cases where there are potential gains to increasing the timeframe, whether society deems these gains worthy of the increased risk requires consideration of tradeoffs between the potential socioeconomic gains afforded by this additional time and potential negative effects on the health of the fish, condition of the ecosystem, and likelihood that rebuilding will be achieved.

Another frequently discussed concern about current rebuilding approaches is the difficulty of rebuilding in the presence of ***mixed stocks***. As noted by Davis (2010), "[i]t may not be possible to rebuild very weak minor stock components of a mixed stock fishery without shutting down the fisheries on healthy stocks, hence there is an important socioeconomic issue involved and some possibly difficult tradeoffs." That is, mandated rebuilding of a stock with little or no commercial value might reduce feasible harvest levels for a highly valued, more abundant species. For example, as described by Rosenberg (2010), "[due to rebuilding measures in place for flounder and cod within the New England multispecies fishery] the higher abundance of haddock means lost opportunity for fishermen […] If effort could target haddock without bycatch then easing of restrictions might be possible."

The management complexities of rebuilding single stocks in multispecies fisheries are not unique to rebuilding or to U.S. fisheries management. For example, Pascoe (2000) estimated the opportunity costs associated with protecting and rebuilding the Australian south east gemfish fishery (a bycatch species) by curtailing catches of other target species in the complex. He found that the costs of protecting the gemfish in terms of the lost economic values associated with not being able to fish the other target species in the complex could be larger than the financial returns from harvesting gemfish even after the stock was rebuilt. This example demonstrates a case in which rebuilding of a species can lead to net economic losses due to the presence of mixed stocks. While this was the case for the gemfish fishery, the generality of the conclusion depends on many factors (e.g., price differences between species, discount rates, nature of the technical interactions, etc.) and it is not clear ex ante that the costs will always be as significant (see, e.g., Armsworth *et al* 2011). Hilborn *et al* (2012) provide a discussion and quantification of similar tradeoffs in the California current bottom-trawl fishery, concluding that rebuilding has come at "considerable short-term cost in yield from stocks that are not overfished." The types of analysis required to understand these trade-offs are discussed in more detail below.

In principle, a mixed stock exception allows flexibility to accommodate cases in which individual species are caught in conjunction with others, for example because of the difficulty and/or prohibitive cost of avoiding incidental bycatch (Holland, 2010).[6] In practice, however, the mixed stock exception does not generally apply for overfished stocks in need of rebuilding. Specifically, the mixed stock exception in MSFCMA applies only when a stock is not currently overfished, mitigating measures have been considered, and increased harvest will (i) not cause the stock to fall below its Minimum Stock Size Threshold (MSST) more than 50% of the time and (ii) generate long term positive net benefits to the Nation (see Chapter 2).[7]

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[6] One important factor that is often overlooked in discussions regarding managing mixed fisheries is the role of fishers' behavior (choice of where, when and how to fish) which is itself a function of the management institution, and how this effects the level of bycatch. See, for example, Abbott and Wilen (2009 and 2011), Holland *et al.* (2010c), Holland and Jannot (2012) and Wilen (2006) for further discussions on these interdependencies.

[7] 74 Fed. Reg. 3178, 3213 (Jan. 16, 2009).

An additional consideration is that many stocks are ***data poor***, especially relative to the data necessary to populate coupled natural-human system models required to understand the impacts of various management options. For example, Beddington *et al.* (2007) estimate that between 30 and 70% of fished stocks in Australia, New Zealand, Europe, and the U.S. have insufficient data for stock assessments (see additional discussion in Chapters 3 and 4). Quantitative stock assessments are available for about 85% of the stocks declared overfished in the U.S. (Chapter 3), but some of the stocks for which no quantitative assessment is available correspond to species complexes. Conceptually, Honey *et al.* (2010) defined data-poor methods as those that could be used to develop qualitative or quantitative control rules, without the guidance of a full stock assessment. From a socioeconomic perspective, most stocks are data poor since baseline data and understanding of socioeconomic trends and causalities do not exist (Abbott-Jamieson and Clay, 2010; Clay *et al.*, 2010). Fulton *et al*. (2011) suggest that human behavior is perhaps the greatest source of uncertainty in fisheries management, but the least adequately accounted for (see also Wilen, 2006).

As discussed in previous chapters, the ability to provide scientific advice on Rebuilding Plans, including stock status determinations and stock projections used to develop the Plans, is subject to several sources of uncertainty. Rebuilding may occur more slowly or rapidly than initially projected. For example, the projected rebuild date for Acadian Redfish was initially set at 2051, yet by 2010 stock assessments showed the stock to be successfully rebuilt, such that rebuilding was considered complete approximately forty years ahead of schedule (Nies, 2012). As a result of the uncertainty inherent in projecting future conditions, rebuilding plans are often adjusted (e.g., timelines, B_{MSY} and F_{MSY}) as new estimates of stock biomass and status (e.g., overfished; subject to overfishing) become available. These adjustments can cause unanticipated and significant economic and social shocks that are positive (e.g., due to stocks reaching a rebuilt status more rapidly than predicted, shorter rebuilding schedules, and more rapid increase in fishing than anticipated) or negative (e.g., due to further curtailing of catches). Recent events in the New England cod fishery (U.S. Department of Commerce, 2012) illustrate the potential harvest reductions that can occur and the potential for attendant social and economic impacts.[8] While regular stock assessment updates are necessary to incorporate new information on stock status, the constraining nature of the law once the overfished status is declared limits potential actions along the path to recovery that could be utilized to reduce the social and economic impacts on communities of the ensuing management actions.

Socioeconomic analysis of Rebuilding Plans

After the biological parameters of the rebuilding program, in particular the rebuilding biomass target and maximum time to rebuild, have been determined as mandated by MSFCMA, the Councils in conjunction with NMFS staff then examine formally and informally a range of management alternatives consistent with these parameters. The formal analyses of the socioeconomic impacts are found, for example, in Environmental Impact Statements (EIS) and Regulatory Impact Review (RIR) documents, and the informal analysis is integrated through

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[8] The Gulf of Maine Cod Working Group (2012) estimated that harvest reductions resulting from unexpected declines in estimated biomass would cause “New Hampshire groundfish revenues [to] be reduced by 91 percent, Maine groundfish revenues [to] be reduced by 54 percent, and Massachusetts groundfish revenues [to] be reduced by 21 percent.”

stakeholder participation in the Council process.

A number of guidance documents are of particular importance in defining the scope and nature of the economic and social impact analysis to be conducted when developing a Rebuilding Plan. These include NMFS' *Operational Guidelines: Fishery Management Plan Process* (NMFS, 1997), *Guidelines For Economic Reviews Of National Marine Fisheries Service Regulatory Actions* (NMFS, 2007a), and *Guidelines for the Assessment of the Social Impact of Fishery Management Actions* (NMFS, 2007b). The Operational Guidelines state that the FMP should include an analysis of the beneficial and adverse ecological, economic and social impacts of potential management options on the fishery as a whole, "in monetary or qualitative terms…" (NMFS, 1997). These Guidelines address the general nature and objectives of the economic and social impact analysis, including that changes should be considered "relative to the status quo." They also identify the scope of communities to consider and the nature of change (e.g., in fishing methods, likelihood of acceptance among fishermen, enforceability, and the effects on health and community viability).

Within the required RIR documents accompanying a Rebuilding Plan (NMFS, 2007a), the Analysis of Alternatives (AOA) presents the data, models, and analysis of the socioeconomic tradeoffs associated with the required reductions in fishing mortality. Alternatives considered within an AOA may alter numerous aspects of a Rebuilding Plan, including timeline (within the biological mandates), associated annual catch limits/ target fishing mortality rate, catch allocations (e.g., among fishery sectors), and the particular combination of input or output controls required to implement a particular rebuilding alternative.

NMFS published guidance on the economic and social analysis within AOAs follows broader guidance in OMB Circular No. A-4 (U.S. Office of Management and Budget, 2003) and Executive Order 12866, "Regulatory Planning and Review."[9] While RMFCs are free to incorporate a wide range of socioeconomic analyses within AOAs, primary emphasis is given to analysis of economic effects.[10] The Councils in conjunction with NMFS staff examine the social impact of a range of management alternatives predominantly in Social Impact Assessments (SIAs) as a component of the Environmental Impact Statements (EIS) under NEPA.

The committee evaluated the breadth, depth, and validity of socioeconomic analyses employed in assessing rebuilding, documented in the AOAs and EIS.[11] While a formal review of all rebuilding fisheries within U.S. jurisdiction was not feasible, the committee reviewed documentation for the following fisheries: Gulf of Mexico red snapper; West Coast canary

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[9] 58 Fed. Reg. 51735-51746 (1993).

[10] Some guidance is provided on non-economic social outcomes in the AOA, however. Among those sections of the guidance document that discuss non-economic social analysis is IV.3.e., "Changes in Other Social Concerns." As stated within this section, "the changes with respect to social concerns that are not captured in the preceding categories of [economic] effects should be addressed. Such concerns may be explicitly or implicitly identified in the problem statement, or they may arise during the development and review of alternative management actions." Required Fishery Impact Statements (FIS) under MSFCMA also require analysis of "social impacts of the proposed action on various components of the fishery being managed, over the entire range of the regulated species, on participants in the fishery and in other fisheries, and on fishing communities." These and other statements in the NMFS economic assessment guidance, however, provide little guidance as to the specific constructs, data and methods to be used when evaluating non-economic social effects, although such constructs, data and methods are present in the separate Social Impact Assessment (SIA) guidance.

[11] While the discussion and findings are cast within the RIR framework, the discussion also applies to the Preliminary Regulatory Economic Evaluation (PREE) that is completed prior to a preferred alternative being chosen.

rockfish; New England cod and haddock; Southeast gag grouper; and mid-Atlantic summer flounder. These fisheries were chosen because they span a number of the geographies and dimensions that are important in determining the socioeconomic outcomes, including recreational and commercial fishing (red snapper and summer flounder), mixed stock fishery (canary rockfish), and ecosystem considerations (cod and haddock, gag grouper). The goal of this review is to evaluate the ways in which socioeconomic information was used to inform the selection of preferred rebuilding alternatives from a broader candidate set that meets required biological parameters.

In this section, the nature of the economic analysis of alternatives in Rebuilding Plans is discussed first, followed by a discussion of the social impact analyses that accompany the plans. These two sets of analyses are separated because they are often completed in parallel within the fishery management process, and are part of different reporting requirements (and documents produced by the Council). In fact, there appears to be a potentially disjointed policy and guidance landscape between economic analysis mainly in AOAs within the context of an RIR and the social impacts primarily in SIAs within a NEPA EIS. This separation would seem to discourage the integration of economic and sociocultural analyses.

ECONOMIC ANALYSIS OF ALTERNATIVES IN REBUILDING PLANS

This section first discusses the economic guidance on AOAs provided by NMFS and then presents findings with respect to the reviewed AOAs developed for Rebuilding Plans. Two central questions are addressed: First, is NMFS guidance for rebuilding AOAs consistent with established approaches for the analysis of economic outcomes and tradeoffs? Second, do rebuilding AOAs in practice incorporate sufficient analysis of these outcomes and tradeoffs?

NMFS guidance on the Analysis of Alternatives

NOAA provides guidance on the types of economic effects that should be considered, the appropriate ways to measure these effects, a summary of underlying economic models, and the types of data and indicators that can or should be used to estimate different economic effects (NMFS, 2007a). The guidance document, however, is not intended to prescribe a particular method but rather provides general assistance in preparing an economic analysis (see Appendix I for Section IV of the document). For example, in terms of the scope of the analysis, the guidance document states that, “economic analysis related to the performance of the relevant commercial and recreational users, non-consumptive users, processing sector, and retail or other market sectors is needed ….” The decision on which sectors to include depends on the specific context. Moreover, while suggesting specific types of quantitative analysis and data, the guidance allows significant flexibility:

> “At a minimum, a qualitative analysis should discuss the relative magnitude of changes in performance. The qualitative components of the analysis should be replaced with quantitative components *when this is the appropriate option*. Information should be tailored to the sector(s) being analyzed, including commercial fishing and processing, recreational and subsistence fishing, and non-consumptive uses of fishery or other living marine resources. Examples of the

> information that should be provided in an RIR, if relevant to the analysis, *may include* the following…" (emphasis added; NMFS, 2007a).

This flexibility aside, the guidance for AOAs is consistent with widely accepted norms for economic analysis. For example, the guidance document recommends a benefit cost framework that compares (either quantitatively or qualitatively) the aggregate benefits and costs for any alternative, along with an analysis of the distribution of the impacts. In cases "where a specific action is mandated by statute or some other binding ruling, a cost-effectiveness analysis" is recommended as an alternative (NMFS, 2007a).[12] The context of the decisions under consideration typically dictates the appropriate framework. These methods are widely accepted and have well-established properties (see, e.g., Holland *et al.*, 2010; Boardman *et al.*, 2006; OECD, 2010; Just *et al.*, 2004).[13] Within these frameworks Councils can consider tradeoffs across time, communities, and types of users. Also suggested is an evaluation of changes in jobs and income, for example as forecast using regional economic models. In addition, the guidance document briefly discusses analytic details such as (i) the need to justify in any forecasting exercise, (ii) assumptions on exogenous factors (e.g., demand for seafood), (iii) the choice of discount rate, (iv) the time period of analysis, and (v) the role of risk and uncertainty.[14] These instructions, while concise, are also consistent with widely-accepted norms for economic analysis, as discussed in, e.g., Boardman *et al.* (2006), Holland *et al.* (2010) and OECD (2010).

The NMFS guidance is less clear regarding the treatment of different types of economic information within an AOA. An advantage of structured frameworks such as benefit cost analysis and cost-effectiveness analysis is the existence of clear guidelines—consistent with economic theory—regarding the use (e.g., aggregation and comparison) of different types of data (cf. Boardman *et al.*, 2006; Just *et al.*, 2004). As discussed in the next section, the NMFS guidance requires quantitative or qualitative presentation of many types of socioeconomic data including varied measures of economic benefits and costs, along with other indicators that do not reflect well-defined benefit or cost measures. For example, as noted by Holland *et al.* (2010), "while the creation of jobs may be desirable from a variety of perspectives—and may represent an informative *economic indicator*—it does not usually represent an economic benefit that is counted in [benefit cost analysis]."

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[12] For example, a cost-effectiveness analysis could be used to determine the minimum cost of achieving a reduction in fishing mortality over time, where the fishing mortality rate is mandated in a Rebuilding Plan. On the other hand, a benefit-cost analysis would be required to fully evaluate the net economic effects of a proposed mixed species stock exemption; this would seek to quantify the net benefits resulting from increased allowable harvest of one or more species in the mixed stock complex compared to the net benefits (or costs) associated with a longer rebuilding time of the species under mandated rebuilding.

[13] Holland (2010b) suggests management strategy evaluation (MSE) as a potential complement to a benefit cost framework for rebuilding analysis.

[14] For example, with respect to risk and uncertainty, the document outlines a tiered approach that increases in complexity and possibly the quality of information: qualitative discussion, sensitivity analysis, and Monte Carlo analysis. Sensitivity analysis considers running various scenarios of the forecast model under different assumptions regarding a parameter, such as ex-vessel price of fish, cost of fuel, discount rate, and comparing the differences in net present value. Monte Carlo methods are more sophisticated tools that can provide a distribution of outcomes under a wider range of uncertainty than a sensitivity analysis (Judd, 1999).

Indicators of economic effects

The indicators of the economic impacts presented in an AOA differ depending on the sector or user group. Table 6.1 illustrates the set of information requirements included in the NMFS guidelines, along with an indication of whether these requirements reflect measures of well-defined economic benefits/costs or ecosystem service values.[15] For example, according to NMFS guidelines, AOA's should include the impact of rebuilding on participation in the fishery (e.g., number of vessels, anglers), the reduction in catches, and changes to the economics of fishing (fish prices, costs of fishing) across all of the alternatives including no action. To address National Standard 8 in MSFCMA, the scale of these indicators must capture the geographic distribution of the impacts (e.g., communities and ports) and the different types of users within the broad categories.

As shown by Table 6.1, the socioeconomic information requirements[16] for an AOA vary widely, and include theoretically appropriate measures of economic benefits and costs, along with numerous other measures that are not necessarily correlated with economic benefits. For example, many required indicators reflect measures of economic impact, activity or gross production. The set of information requirements and indicators fall along a continuum in terms of data needs and complexity. For example, an indicator such as the actual (or predicted) change in days at sea is easier to calculate and less uncertain (i.e., due to readily available monitoring data and the relative simplicity of the indicator) than a measure of the change in commercial fishing profits. However, a change in days at sea is difficult to interpret in terms of the overall impact on the economics of the fishing operations (fewer days could be accompanied by higher prices of fish and therefore correspond to higher fishing revenues and vice versa). An increase in profits, on the other hand, represents an economic benefit of the particular action for the commercial fishery. Estimating changes in profits requires estimates of operating costs and sophisticated econometric techniques for analysis. These may not always be available or feasible within the context of a rebuilding AOA (e.g., the time, expertise or data may be unavailable). There are hence tradeoffs in the type of economic information used to evaluate rebuilding alternatives.

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[15] Ecosystem service values are defined as "the flows from an ecosystem that are of relatively immediate benefit to humans and occur naturally" (Brown *et al.*, 2007). Additional discussion of these values is provided later in this chapter.

[16] Requirements are for either qualitative or quantitative consideration as appropriate within a given context.

TABLE 6.1: Information Requirements Listed in Section IV.6 of Guidelines for Economic Review of NMFS Regulatory Actions—Description and Economic Interpretation

Information Requirement	Interpretation (type of indicator or estimate)	Well-Defined Measure of Economic Welfare (benefit or cost)?	Focused Solely or Primarily on Commercial and Recreational Fisheries	Quantifies Ecosystem Service Values Beyond Those Realized by Recreational and Commercial Fisheries
Expected levels or changes in participation (number of fishing vessels and/or anglers, etc.) and activity (number of fishing trips, days at sea, etc.).	Economic impact, activity or gross production	No	Yes	No
Expected levels or changes in harvests (commercial, recreational, and subsistence) and their distribution by sector.	Economic impact, activity or gross production	No	Yes	No
Expected levels or changes in non-consumptive use of the resource.	Economic impact, activity or gross production	No	No	No
Expected changes in prices (commercial ex-vessel prices and recreational access prices).	Market prices	No	Yes	No
Expected changes in harvesting costs (fixed and variable costs, including capital and landing costs), as well as equivalent costs for non-consumptive use activities.	Benefits and costs	Yes	Yes	No
Expected levels and costs of processing.	Economic impact, activity or gross production; benefits and costs	Yes	Yes	No
Expected changes in benefits and costs incurred by specific user groups, including effects on small entities.	Benefits and costs	Yes	Yes	Possibly (to the extent that these capture benefits and costs outside of recreational and commercial fisheries)
Expected effects on employment.	Economic impact, activity or gross	No	Yes (unless significant	No

	production		employment effects are expected in other sectors)	
Expected effects on profits, competitive position, productivity or efficiency of individual fishermen, user groups, or fishing communities.	Multiple, including measures of benefits and costs	Yes (profits can approximate producer welfare); No (all others)	Yes	No
Expected effects on the reporting burden.	Compliance requirements	No	Yes	No
Expected impacts on recreational and subsistence use, including changes in participation and catch rates and, to the extent practicable, their consumer surplus; for subsistence fishing, food and cultural availability.	Multiple, including (i) economic impact, activity or gross production, and (ii) benefits and costs	Yes (consumer surplus); No (all others)	Yes	No
Expected management and implementation costs attributable to the action, including enforcement costs.	Benefits and costs	Yes	Yes	No
Expected effects on non-use values.	Benefit and costs	Yes	No	Yes
Expected effects on fishing capacity.	Industry size	No	Yes	No

Analysis of alternatives in practice

While all of the AOAs must be "a reasoned assessment of the expected direction of change in net benefits to the Nation, as well as the specific effects of individual entities of a proposed regulatory action," the guidelines are not prescriptive (NMFS, 2007a). This reflects the need to adapt analyses to the characteristics of affected fisheries and stakeholders, and to variations in data and model availability. As a result, there is variation in the economic evaluations contained in rebuilding AOAs implemented by the RMFCs.

The variation of the AOAs is in part due to the idiosyncratic nature of the economic science available across the regions. That is, in some regions, NMFS and academic economists (many times in partnership) have models and analysis on a specific fishery readily available when a rebuilding AOA is carried out. For example, researchers might have access to multiple years of industry survey data to develop measures of fishing costs and/or data to estimate demand curves for the fish from which to calculate consumer surplus. In some cases, the net benefits of recreational anglers in affected fisheries may be estimated because associated research has been done previously in the region. In other cases, no economic research has been conducted on the particular overfished fishery and the effort necessary to conduct the research does not fit the regulatory time frame. For example, without cost data, the analyst will likely focus (rightly so) on gross fishing revenues or discuss the impacts qualitatively. In economic science, quantitative estimates that allow direct comparisons across sectors of the different alternatives are preferred, but qualitative descriptions are illustrative and also valuable for decision-makers.

The variability in data availability and research stem in large part from the lack of economic data collection mandates for the NMFS (unlike in the stock assessment realm). In many instances, the commercial fishing industry opposes the collection of economic and fishing (e.g., location of where fish are caught) data due to confidentiality concerns (National Research Council, 2000). Rules on the collection of economic data were also only recently relaxed during the reauthorization of MSFCMA in 2006 (see sections 303(b)(7) and 402(a) of the amended MSFCMA, and discussion within National Research Council, 2000). Another limiting factor to comprehensive economic analysis is the predominant focus on commercial and recreational fishing in the assessments of the economic value of fish stocks to the Nation found in the Stock Assessment and Fishery Evaluation (SAFE) Reports. These estimates often do not capture the total economic value to the Nation of the fish stock, as that would include the non-fishing recreational and potential non-market values (see discussion below).

While the Committee's charge does not include peer review of specific rebuilding AOAs, a number of areas were identified that, if addressed, could improve the analysis of social and economic effects within rebuilding AOAs. We discuss the primary findings of this review below.

Forecasting rebuilding effects over time and space

Rebuilding AOAs in general follow guidance provided by NMFS on analysis of economic effects. However, while the recommended types of tools and analysis are applicable to all RIRs, rebuilding AOAs are complicated by a number of factors. These include a need to forecast effects during a transition that may extend over long periods of time. These forecasts are complex because the associated economic and social dynamics are both impacted by and impact the transition. The Committee's review suggests that rebuilding AOAs differ substantially in their treatment of these dynamics, and particularly in their treatment of endogenous versus exogenous factors in the coupled natural-human system of the rebuilding fishery.

Endogenous factors are factors that are impacted by the alternative under consideration; these are effects that are determined *within* a fishery's socio-ecological system. These include fishers' decisions on where, when, for what species, and how to allocate fishing effort. These decisions often affect the dynamics of rebuilding; reallocation of fishing effort can either slow or speed recovery.[17] Other potentially important feedbacks include potential changes in net fishing revenue, above and beyond that related solely to an assumed change in harvest. Such net revenue changes may be caused by a variety of endogenous factors including price responses to reduced landings (e.g., NEFMC, 2003), more abundant larger fish (especially in fisheries where there is a significant price gradient over size), or reduction in the search cost for fish, as the fish populations rebuild. These changes are likely to influence entry/exit decisions and profitability of the fishing fleet during the rebuilding period. The omission of these factors from the forecasts of

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[17] An example would be reallocation of fishing effort to another species in the complex; this might reduce the impact of directed fishing on a rebuilding stock but increase the impact of bycatch, with concomitant impacts on rebuilding dynamics.

economic effects can lead to an overestimation of costs to the fishing industry from rebuilding. To fully capture the effects of these and other endogenous factors, the analyst must couple an economic model of the commercial and recreational fishing enterprise with the fish population dynamics.

Another important yet often overlooked endogenous factor is the role of fishery management. The link between the type of regulatory structure (regulated open-access, limited-entry, catch share) and the economics of fishing is well-known (see, e.g., Sanchirico and Wilen, 2007 and citations therein). The implication of this link is twofold. First, using data on fishing operations and socioeconomic impacts from one regulatory regime to forecast the impacts in another regime may lead to generalization errors (see, e.g., Wilen, 2007). Second, any assumption regarding fishery management in a distant time period is speculative at best. Most analyses proceed under the assumption that the relevant regulatory structure will remain fixed over the rebuilding horizon, unless changes in regulatory structure are under consideration as part of the AOA. While changes in regulatory structure are difficult to predict—perhaps justifying these common assumptions—they can lead to misleading forecasts of socioeconomic effects when regulations change over time.

Exogenous factors are those that are not impacted by the specific rebuilding alternative, but may change over time (and potentially over space). Changes in these factors also influence socioeconomic impacts. Given the length of time covered by many rebuilding analyses, the potential impact of these exogenous factors can be substantial. An example would be the price of fish when there are many substitute fish available to the consumer. In this case, there would be little change in the price of the particular fish due to the reduction in landings, yet fish prices might change substantially over time due to external events. Similarly, the price of fuel is not likely to change as a result of rebuilding, yet will likely change over a rebuilding timeline that may extend over decades. Other exogenous dynamic variables, such as changes in coastal population, alternative fishing opportunities, and demand for recreational fishing also could also be incorporated to provide the Councils with more robust estimates of future impacts.

The analyst has a number of options for addressing relevant exogenous factors within a rebuilding AOA. First, an analyst might assume that these factors are fixed over time. For example, in the canary rockfish rebuilding AOA, which covers a 50-yr time span, the analyst assumed that fish prices and fuel costs would remain constant. Another option is to assume that these factors change over time based on historical rates and patterns (e.g. fuel or fish prices). Third and perhaps most relevant, one can conduct sensitivity analyses to evaluate the potential sensitivity of socioeconomic effects to a range of possible outcomes for exogenous factors that may change over time.[18]

When addressing the potential role of endogenous and exogenous factors within a rebuilding analysis, the analyst must balance the additional information provided by an approach that accommodates change in these factors against the time and data required to develop more complex models. As a generalization, many AOAs err on the side of oversimplified economic

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[18] Some factors might simply scale up or down the impacts but others could impact the relative ranking of alternatives. For example, assuming a constant price of fuel out into the future could lead to conclusions that while closing off areas further from shore are less likely to lead to result in the same economic impacts as closing off inshore areas both are economically viable options. On the other hand, if fuel prices are likely to rise in the future, the conclusion could be closing the near shore areas leads to unprofitable fishing while closing off the offshore areas is still economically viable.

analysis, which stands in contrast to the relatively complex fish population dynamics models used to forecast biological components of the fishery. Many of the stylized assumptions used in forecasting limit the applicability of AOAs to meaningfully quantify future social and economic impacts. The result is that rebuilding AOAs primarily address short-run economic impacts. When longer-run analyses are conducted, they typically rely on simplifying assumptions that limit their relevance to longer-term forecasting (e.g., assuming fixed prices over time). Therefore, many of the analyses are more appropriately considered a short to medium-run analysis even if they are being simulated over a longer time span. As a result of these simplifications and underlying uncertainty in natural and human factors, the ability of these models to accurately project conditions that will occur in the far future is limited.

The development of models that couple the dynamics of the natural and human systems (e.g., bioeconomic models) could improve forecasting of rebuilding effects, as they would incorporate the behavioral responses of the industry, the changes in the fish stocks, and other endogenous changes over time within a single modeling framework. Such models would also provide improved forecasting of the changes in fisheries (including, e.g., the number and type of vessels) that would accompany rebuilding of fish stocks, as these two types of outcomes do not necessarily move in tandem. Even in the absence of fully-developed models of this type, greater attention to potential changes in both endogenous and exogenous factors (e.g., ecosystem considerations) over time, and the feedback among them, would provide a deeper and broader understanding of the socioeconomic impacts to fishery managers.

Data and model availability

As previously mentioned, the ability to carry out quantitative assessments is complicated by a lack of necessary data and models. For example, in the AOA surrounding Amendment 13 in the New England Multispecies Fishery, the analyst measured changes in producer and consumer surplus and carried out a Monte Carlo analysis of these economic changes under different assumptions regarding the level of uncertainty (NEFMC, 2003). In contrast, the analysis surrounding Amendment 16-2 in the West Coast Groundfish fishery used fishing revenue and landings and did not account for uncertainty in these estimates (PFMC, 2003). Addressing data and modeling gaps will require resources beyond those typically available to the Councils. These efforts would require a collective and collaborative enterprise across the regions, in which analysts collaborate to create standardized assumptions and analysis reflecting best practices.

Because of the lack of data or appropriate models, the AOAs often use proxies to measure economic effects.[19] For example, fishing profit is often approximated using accounting techniques, whereas the true measure of economic profits requires an estimate of economic costs (opportunity costs) and incorporates the effects of a rebuilding stock on the revenue and costs of fishing (as discussed above). In other cases, fishing revenue is used as a proxy for fishing profit and as such does not account for the costs of fishing (that could be going down over time as the fish stock rebuilds). In the AOAs reviewed, analysts provided explanations of the pros and cons of the different proxies. However, these explanations often lacked (i) a discussion of the

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[19] In other cases, benefit transfer, or a parallel transfer of biological information, is used to approximate economic or biological outcomes based on research conducted elsewhere or for other purposes (Johnston and Rosenberger, 2010).

quality of the data used to measure the proxy and (ii) guidance for the Councils on how to interpret the proxy, considering both the quality of the data and theoretical differences between the indicator and proxy. Inconsistencies in how proxies are measured (e.g., what was considered a fixed or variable cost, whether gross revenues included different prices for different sizes of fish, etc.) across the different AOAs also reduces the comparability of socioeconomic impacts across fisheries.

A tiered rating system to evaluate the proxies in terms of data and theoretical differences is one possible method of conveying the uncertainty around estimates. For example, results classified under Tier 1 might be derived from a peer-reviewed methodology and up-to-date socioeconomic data to measure economic benefit or cost. Those classified under Tier 2, in contrast, could utilize older or more limited data in an otherwise rigorous analysis. For instance, the Amendment 27 AOA for the Gulf of Mexico red snapper fishery uses cost data that were over 10 years old in the measurement of fishing profits. The analyst had no other cost data available, and the associated assumptions in the analysis were clear. Nevertheless, the use of older cost data (from a different management regime) introduces a source of potential error. Tier 3 could identify indicators measured by imprecise proxies or otherwise flawed data. A ranking system such as this could enable a Council to place more weight on those indicators that are considered more reliable and precise. Such a rating system, however, would require careful development and scrutiny to ensure scientific validity and salience to the analyses being conducted.

Comprehensive measures of economic effects

Within the reviewed rebuilding AOAs, the quantified economic effects emphasized outcomes pertaining to commercial and recreational fisheries. This reflects a similar, if implicit emphasis in NMFS guidance. For example, of the fourteen illustrated “examples of the information that should be provided in an RIR” (NMFS, 2007a), twelve address economic outcomes in these two sectors alone. Within the reviewed AOAs, nearly all quantified socioeconomic effects relate directly or indirectly to participation (e.g., number of vessels fishing), net economic benefits, or economic impacts (e.g., jobs, income) in the commercial or recreational fishery. While the guidance document discusses the need to quantify the non-market ecosystem services that are potentially generated from rebuilding the fish stock and other measures of socioeconomic effects, such quantitative measures are rarely found in rebuilding AOAs. Rather, a lack of readily available information typically leads the analyst to include a qualitative discussion of these effects, if they are mentioned at all. For example, while NMFS guidance explicitly lists “expected effects on non-use values” as an example of “information that should be provided in an RIR, if relevant to the analysis,” (NMFS, 2007a) none of the rebuilding AOAs reviewed by the committee contained a quantitative analysis of these values. The omission of quantitative information is particularly relevant for values of affected ecosystem services and other non-market benefits. The NMFS guidance document, reflecting established norms for benefit cost analysis, identifies non-market values as one of the relevant components of analysis: “Not all goods and services important to people are exchanged through markets, nor receive market prices. Including non-market values may be particularly important when considering amenities, such as habitat, ecosystem, recreational experiences, and protected resources, or issues affecting cultural heritage, historical and/or archeological assets, or other unique community resources." (NMFS, 2007a) Established methods exist to quantify such non-market benefits (Freeman, 2003; Holland *et al.,* 2010). Yet, unlike regulatory benefit cost analyses at U.S. EPA and elsewhere in which non-market benefits are routinely considered (Griffiths and Wheeler, 2005), rebuilding AOAs typically either do not include these benefits or provide a brief qualitative discussion.

The omission of non-market values may or may not influence the selection of a rebuilding alternative. For example, if market and unquantified non-market benefits are correlated and/or if unquantified non-market benefits are small relative to market benefits, the inclusion of quantified non-market benefit estimates might not change the qualitative conclusion regarding different alternatives. In other cases, however, an alternative that yields lower market returns but larger non-market benefits could be discounted, or not considered at all by a Council due to the lack of quantitative measures of these services. In such cases, the omission of non-market benefit or cost estimates from an AOA could result in an error in calculation of economic net benefits and a selection of regulatory alternatives based on partial and potentially incomplete information.

In many cases, quantification of non-market benefits and costs may not be feasible due to data limitations. Yet even in these cases, benefit transfer techniques are increasingly available to enable approximations of these benefits (Johnston and Rosenberger, 2010).

Treatment of Risk and Uncertainty

The role of risk and uncertainty on the socioeconomic effects of rebuilding may be evaluated by considering at least three broad components (Holland *et al.,* 2010): (1) what are the sources of risk and uncertainty in designing Rebuilding Plans; (2) whether to use (and model) a consistent decision framework that includes risk and uncertainty explicitly into the decision-making process (e.g., maximizing the expected value of the fishery subject to different types of stochastic shocks, see, e.g., Sethi *et al.,* 2005) ; and (3) whether to estimate a distribution of outcomes with respect to any alternative and present a range of possible outcomes rather than point estimates (e.g., Monte Carlo analysis).

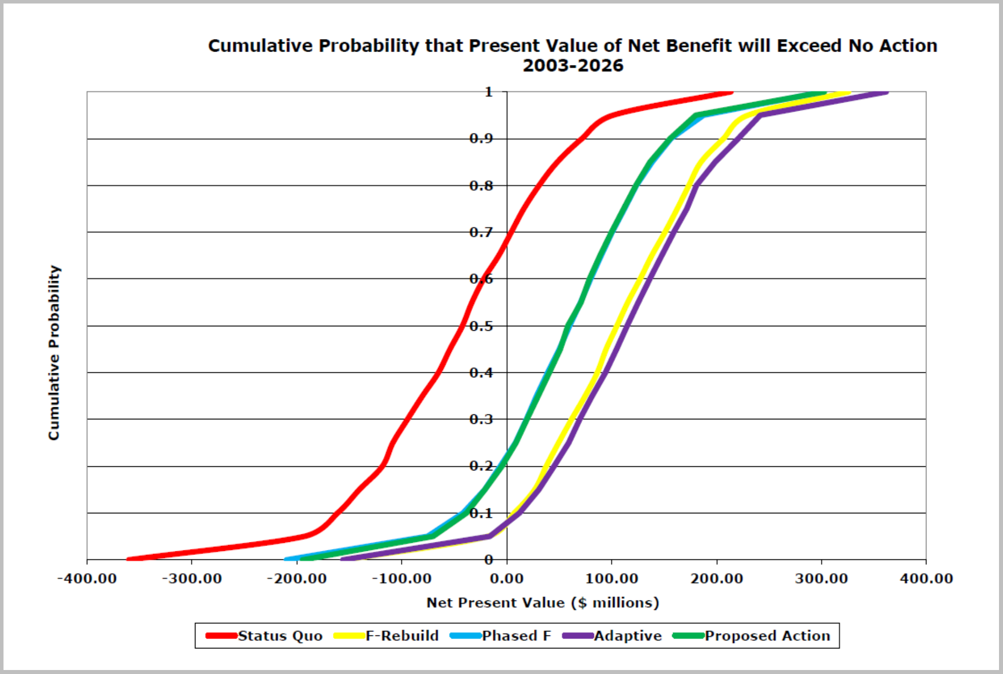
The reviewed Rebuilding Plans and the alternatives considered addressed biological and implementation uncertainty in the evaluation of the different rebuilding times and employment of buffers in the settings of ABCs, ACLs, etc., as discussed in Chapters 3 and 4. There are, however, other sources of uncertainty that could inform the setting of the rebuilding targets. For example, fish, labor, and fuel prices are uncertain over time. Currently, these other sources of uncertainty and risks are considered, if captured at all, at the time of generation of the AOAs rather than during the determination of the rebuilding targets. Uncertainty and risk are therefore treated in a sequential rather simultaneous fashion, which only considers a subset of the risks faced by managers and fishers. Research in the decision-sciences has shown that considering multiple sources of uncertainty simultaneously can lead to different management outcomes than focusing on individual aspects in a sequential manner (see, e.g., Sethi *et al.,* 2005). Without further analysis, it is not clear if this partial treatment results in buffers that are overly cautious or too risky from society's perspective (see, e.g., Sethi *et al.,* 2005; Kapau and Quass, 2013).

The guidance document discusses risk and uncertainty but does not recommend using a decision-theoretic framework, such as expected value analysis (Holland *et al.,* 2010) that can consider and weigh multiple sources of risk simultaneously. Rather, the discussion focuses on using sensitivity analysis, which can investigate how a measure such as net present value changes with a change in a parameter—the range of values of the parameter could stem from uncertainty on its level in the future. Sensitivity analysis is informative but provides little guidance for the Councils on the relative importance of the uncertainty of one parameter over another or on potential synergistic or opposing effects of multiple types of uncertainty. Many sensitivity analyses present the impacts as point estimates rather than as a range of possible outcomes that would emerge from a decision-making under uncertainty framework.

An improvement over sensitivity analysis is Monte Carlo analysis, whereby an analyst can evaluate the expected net present value, considering multiple sources of uncertainty at one time, and assigning probabilities (or frequencies) to different outcomes. Monte Carlo analysis was used in the analysis of Amendment 13 in the New England cod fishery rebuilding AOA (see Figure 6.2 in Box 6.2).[20]

BOX 6.2
Cumulative Probability

Figure 6.2. Cumulative probability that the present value of the net benefit of five different alternatives considered by the New England Fishery Management Council will exceed a no action alternative over the period 2003-2026. For example, the figure illustrates that there is a 70% chance that present value of net benefit from the status quo, which represents maintaining the current rebuilding targets, will exceed the no action alternative. (Source: Figure 6.2 is Figure 210 found on page I-603 in Northeast Multispecies Amendment 13 SEIS December 18, 2003).



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[20] Note that Monte Carlo and similar analyses require that the range of possible outcomes is bounded, and that the analyst is able to specify or approximate probability distributions for these outcomes.

A more standardized approach to accounting for risk and uncertainty in rebuilding AOAs will provide the Councils with a greater understanding of the implications of risk and uncertainty for decision-making. The literature on decision-making under uncertainty is rapidly advancing both in the understanding on how people respond to risk and in the ability to model and analyze decisions under these conditions. For example, recent advances in computing capacity have allowed researchers to develop a richer understanding on how investing in learning can influence the optimal set of decisions over time in the presence of multiple forms of uncertainty (Walters, 1986; Bond and Loomis, 2009; Zhou *et al.,* 2010). Operationalizing learning, risk, and uncertainty into an AOA might be years away but these fundamental features are present in Council decisions and should be included and considered rigorously.

SOCIAL IMPACT ANALYSIS OF ALTERNATIVES IN REBUILDING PLANS

In this section, the guidance on social impacts provided by NMFS is discussed, followed by a discussion of the SIAs developed for a sample of Rebuilding Plans. As was the case for the review of the economic analyses in the previous section, the committee's evaluation focuses on two central questions. First, is NMFS guidance for SIAs consistent with established approaches for the analysis of social outcomes and tradeoffs? Second, do EISs in Rebuilding Plans in practice incorporate analysis of these outcomes and tradeoffs?

NMFS guidelines for measuring social impacts

The NMFS guidance for SIAs aim to "provide Councils and fishery managers with an understanding of the objectives and techniques of SIAs...[laying] out the general process, analytical content and form of SIAs" (NMFS, 2007b). Whereas economic assessments address the market and non-market values and systems, SIAs consider the social and cultural values and systems, i.e., describing the social characteristics of a fishery and community (i.e., social factor analysis) and describing the effects of social changes (i.e., social impact assessment). SIAs are used to predict potential adverse impacts from management changes, or to evaluate the likelihood that the current social and cultural context has been caused by past changes in fisheries management associated with stock availability.

While SIAs are required under NEPA, the amendments of the MSFCMA have expanded the scope of SIAs to consider cumulative social impacts and clarified social factors, including definitions of fishing community,[21] and the charter, commercial, and recreational fishing sectors. The SIA calls for the use of a social factor analysis framework that catalogues five major categories of social variables of interest in fisheries management: lifestyle (e.g., indigenous

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[21] Community has many definitions in social science, but the Magnuson Stevens Fisheries Conservation and Management Act defines fishing community as "…a community which is substantially dependent on or substantially engaged in the harvest or processing of fishery resources to meet social and economic needs, and includes fishing vessels owners, operators, and crew and United States fish processors that are based in such a community." (16 U.S.C. § 1802, Sec. 3, 104-297 (16)). It is clear that fishing communities engage in fishing in a complex, multi-species manner, shifting between species and activities and through geographic space both on land and at sea (e.g., St. Martin and Hall-Arber, 2008; Jacob *et al.,* 2013; Tuler *et al.,* 2012; Hall-Arber *et al.,* 2001).

peoples, subsistence fishing, ethnic fishing practices, etc.); attitudes, beliefs and values (e.g., fishery and community norms and values); social organization and structure (e.g., at the fishery, community and family levels of analysis); population demographics (e.g., education, ethnicity, etc.); and dependence on and participation in the fishery (e.g., historic and present participation data). The social factor landscape is charted graphically to consider a baseline (i.e., community profile under the fishery management status quo), baseline projections without management changes (i.e., social transitions underway independent of fishery management), baseline projections with the management changes under consideration, and an overall social impact assessment, across each of the five social factor categories. See NMFS, 2007b, pg 22 for the Framework for Social Factors Analysis table.

The prerequisite for a SIA is the development of the baseline case, or status quo in the fishery. While the baseline arises from community profiles conducted every three to five years (Abbott-Jamieson and Clay, 2010; NMFS, 2007b), the funding and staff resources have been insufficient to update community profiles—consequently more rapid assessment and streamlined methods are being developed for updating social baselines (Feeney, 2012; Tuler *et al.,* 2012).

Estimating the social changes from each alternative action should be grounded in the baseline information and assessed with the same variables used to estimate social change in the status quo. Occasionally the anticipated change in the status quo may be expressed in qualitative terms since some factors, e.g., lifestyle changes, are not currently or readily expressed in direct numerical terms. The guidance notes that the SIA may gather additional information through literature reviews, surveys, analytical deduction, focus groups, and Delphi methods (i.e., facilitated expert panels focusing on forecasting based upon their collective professional judgment), population samples, and statistical analyses, and should be integrated with economic and biological assessments.[22] Further, the SIA "must forecast for a period of time (several years) beyond the year in which the conservation goal is attained…long enough to allow a consideration of all expected social effects. Care should be taken to ensure that the assessment time-frames are the same for the ecological, economic, and sociological impact analyses." (NMFS, 2007b). The guidance also identifies the wide range of methods for projecting social impacts.

While the guidance for SIAs is consistent with widely accepted norms for SIAs, rapid advancements are being made and new methods have been developed since its 2007 publication. A few examples include: performance measures—distributional outcomes, stewardship, and governance measures (e.g., Clay *et al.,* 2010); well-being (e.g., Pollnac *et al.,* 2006); and community vulnerability, resiliency and dependency indicators (e.g., Jacobs *et al.,* 2013; Helies *et al.,* 2010). For example, the recently developed streamlined vulnerability assessment tools are applying theoretical and analytical frameworks from risk analysis and behavior research from the hazards and emergency management and environmental pollution control context (Tuler *et al.,* 2012).

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[22] The integration of social, economic and biological assessments is predominantly achieved through the decision-making process via early involvement and cooperation among social scientists, economists, fishery biologists, and fishery managers.

Analysis of social impacts in practice

In a sample of rebuilding plans from across the country, the scope and nature of the SIAs were reviewed and illustrated wide variability (Table 6.3).

Table 6.3. Elements of the Social Impact Assessments in Rebuilding Plans

Rebuilding FMP	**Scope of Social Impact Assessment**
Red Snapper, SAFMC	✓ ***Community profiles*** (3)—compiled from permit, processor, and census data. ✓ ***Recreational fishery demographic*** data review—Marine Recreational Fishery Statistics Survey (MRFSS) and Marine Recreational Information Program (MRIP) data. ✓ ***Community dependency***—composite of indicators: communities ranked based upon dealer reported landings of red snapper and shrimp (snapper bycatch); permit data (number of owners, active and inactive permits, percentage inactive, number of vessels); and processed pounds, value and employment in shrimp fishery.
Cod & Haddock, NEFMC	✓ ***Community profiles***—interviews; secondary data: by gear type, ethnicity, and education-level; . ✓ ***Recreational demographic*** data—MRFSS/MRIP telephone and intercept surveys. ✓ ***Community dependency***—composite of indicators: participation in leasing program (numbers and value) by port, region and time; processor data (number of employees, wages paid, by state); percentage of labor force involved in fishing; percentage of related occupations within relevant Bureau of Labor Statistic categories; summary measure of a series of dependence ratios that compare number of fishermen per hundred community residents to various alternative occupations fishermen could enter with their skill profiles; State of Maine regulatory impact survey. ✓ ***Vulnerability***—comparing communities based on five fishing related occupations; percentage of total employment; alternative occupation ratios; dependency ranking from a MARFIN study; summary from social impact public meetings. Compared the social impact on communities based upon: likely regulatory discarding; safety; disruption in daily living; changes in occupational opportunities and community infrastructure; and formation of attitudes. ✓ ***Sociocultural context***—for each alternative, changes were considered for the following indicators: size and demographic characteristics of fishery workforce; cultural issues (attitudes, beliefs, values of fishermen, their families, and communities); social structure and organization (capacity social support and services to families); non-economic social aspects (lifestyle, health, and safety issues); and historical dependency (structure of fishing practices and income distribution). ✓ ***Temporal analysis***—2009 Framework Adjustment 44 contained a basic

	comparative analysis of seven ports over time to assess cumulative and disparate impacts of management measures, using secondary fishery economic and demographic data.
Canary Rockfish, PAFMC	✓ ***Recreational demographic*** data—MRFSS/MRIP telephone and intercept surveys. ✓ ***Community dependency***—upon commercial fisheries and upon recreational fisheries, ranking communities based upon indicators: number of permits as percentage of each state's total number of permits; number of commercial fishing vessels; revenue from landings as share of coastwide revenues from landings; number of processors/buyers; number of charter vessels as percentage of each states total number of charter vessels; number of private/rental angler trips as a percentage of each state's total number of private/rental angler trips; number of private/rental groundfish angler trips as a percentage of each state's total number of private/rental groundfish angler trips; number of party/charter trips as a percentage of each state's total number of party/charter trips; number of party/charter groundfish trips as a percentage of each state's total number of party/charter groundfish trips. ✓ ***Resilience***—community rankings based upon indicators: industry diversity index; unemployment rate; percentage of the population living below poverty line; isolated cities; and population density. ✓ ***Vulnerability***—Social Vulnerability Index (SoVI) score rankings counties based upon communities that are both highly engaged in fishing and highly dependent upon fishing, thus having low resilience: SoVI project team has identified seven indicators that has explained 69% of the variability in vulnerability measures (i.e., race and class; extreme wealth; elderly residents; Hispanic ethnicity; care dependent females; Native American ethnicity; and service industry employment). Before employing the SoVI methods, earlier FMP amendments used indicators from existing U.S. Census and Bureau of Labor Statistics datasets.

Thus, the capacity of SIAs to provide comprehensive and valid perspectives on the total social effects of rebuilding varies substantially across fisheries and Councils. Further while the reviewed SIAs added innovative social science methods and indicators over time as those methodological approached became available, the result has been SIAs that are difficult to compare over time and they cannot fulfill the aim of NMFS' SIA guidance without that consistent baseline data to make projections. Further, economic data through benefit cost analyses and its use in regulatory decision-making is generally more established than social assessments methods; thus, while there is a deficiency in the nature and scope of all socioeconomic data, fewer social data are collected and available for fisheries management than economic data.

Forecasting rebuilding effects over time and space

With varying degrees of specificity, rebuilding FMPs acknowledge the social context and potential impacts, qualitatively, from management actions. For example, the red snapper SIA emphasized the social impacts on the shrimp fishing coastal community from lower shrimp prices, higher oil prices, and the fact that the coastal communities were still recovering from hurricane Katrina. Some communities were more likely to be impact through reduced shrimp fishing effort than others. By contrast, the SIA for cod and haddock within the New England multi-species groundfish complex fishery acknowledged a finer scale of social impact, including sociocultural forecasts, e.g. see following statements from FMP Amendment 5, starting on page 366 (NEFMC, 1993):

- "Fishing-dependent communities…will vary in their ability to adapt to the proposed actions." (citing examples, Stonington, ME and Gloucester, MA).
- "The sociocultural impacts will not be uniform across the region, across vessel sizes or even across gear types. Nor will the impacts be the same for each community, each generation of fishermen, each ethnic group, and each organization. It is partly this certainty—that the impacts will vary—that creates anxiety among all who are involved in the fishing industry."
- "The impacts of a restrictive management system, or of economic hardship brought about by declining stocks, will likely magnify…conditions, further polarizing groups within individual communities. The divisiveness could be exacerbated by members of one group only reporting violations by fishermen from ethnic groups other than their own."
- "For a variety of reasons, including scientists' earlier mistakes in predicting some stock sizes (e.g., herring) and past experience with regulatory change…many fishermen do not believe that the new regulations will have the positive benefits predicted…. Fishermen's fears about the impact of the proposed measures could lead to a greater degree of non-compliance with regulations and/or technological innovations…"

While the social impact forecasts on New England communities are relatively specific, they are not quantitative and do not present changes from baseline data to analyze long-term trends. Nonetheless, the on-going adaptive management activities undertaken by the NEFMC indicate substantial advancements in the scope and nature of the SIAs from Amendment 5 (1994) to Amendment 13 (2001), and expansion of stakeholder engagement opportunities in Amendment 13 and 16 (2009), as the social science methods have progressed. Nonetheless, it is not clear that these new applications of social analyses are part of a long-term baseline data collection effort.

Canary rockfish and the other case studies showed a similar pattern—Councils have incrementally increased the scope and nature of their SIA methods in subsequent FMP Amendments. While these advancements compound the challenge of establishing and systematically monitoring baseline social and economic data, they reflect critical development and evolution of the state of the knowledge. Further, given the potential for disproportionate social impacts in specific communities, states are occasionally investing in additional social analysis to contribute to the overall social and economic impact assessment (e.g., Maine's regulatory impact survey in Northeast groundfish/cod and Washington State's depressed communities analysis in Pacific groundfish/canary rockfish).

Overall however, baseline social impact data are rarely available, precluding projections of impact into the future and any qualitative or quantitative assessment of tradeoffs. For example, across the country 177 coastal community profiles were completed by 2005, with the intention of updating the profiles every three to five years, but staffing and funding limitations have prevented these updates (Feeney, 2012; Abbott-Jamieson and Clay, 2010).[23] There have been comprehensive community case studies to qualitatively characterize community vulnerability (e.g., McCay and Cieri, 2000 in mid-Atlantic; Hall-Arber *et al.*, 2001 in the Northeast), although the longitudinal monitoring does not exist.

Indicators of social impacts and the models of vulnerability

Since direct social data are rare and expensive, and time-consuming to gather, particularly the non-quantitative factors (e.g., social and community networks, cultural heritage values, subsistence fishing practices, etc.) that contribute to community dependence, resilience and vulnerability, indicators are one strategy to address this deficiency. Most commonly, indicators depend upon existing, secondary data, which emphasizes the quantitative economic activity and outcome measures. However, there are numerous indices for vulnerability emerging, often with financial support and research staff contributions from the NMFS regional science centers. Each employ slightly different definitions and methods (see Box 6.3).

BOX 6.3
Advances in Vulnerability and Resiliency Measures

Rapid Impact and Vulnerability Assessment (RIVA)—New England
Building from concepts of risk vulnerability in environmental pollution and risk analysis (i.e., exposure, sensitivity, adaptive response actions, and adaptive capacity), the rapid impact and vulnerability assessment (RIVA) model was developed and refined through support from the NMFS Northeast Fisheries Science Center. RIVA gathers field data (e.g., interviews, secondary data sources) and analyzes causal pathways linking stressors, consequences, and the factors contributing to vulnerability. Through an iterative qualitative and graphical analytical strategy, themes of potential causal links emerge and are ground-truthed with community informants. (see, http://seri-us.org/sites/default/files/RVA%20guidance.pdf)
Social Vulnerability Index (SoVI).
The index synthesizes thirty socioeconomic variables, primarily from U.S. Census Bureau, to measure a communities ability to prepare for, respond to, and recover from changes in regulations. SoVI was used in the Pacific canary rockfish FMP amendment process. The SoVI project team identified seven indicators that has explained 69% of the variability in vulnerability measures (i.e., race and class; extreme wealth; elderly residents; Hispanic ethnicity; care dependent females; Native American ethnicity; and service industry employment). It applied the measure to coastal counties and compared counties. (See http://webra.cas.sc.edu/hvri/products/sovi.aspx)

Vulnerability Index—Gulf of Mexico
Composite measure from indicators of social, economic, and ecological vulnerability and resiliency, and social disruption. Social vulnerability and resilience are measured with their own cluster of indicators, including population composition, poverty, and housing characteristics.

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[23] See http://www.st.nmfs.noaa.gov/humandimensions/community-profiles/index for comprehensive dataset of community profiles.

Economic structure underlies economic vulnerability and resiliency, whereas natural and technological disaster measures are indicators of ecological resiliency. Social disruption is measured by housing, economic and personal disruption measures. Another iteration of the Vulnerability Index applied in the Gulf consists of measures of employment opportunity and community well-being from U.S. Census and other data sources from the SIA (See Jacob *et al.,* 2013; Helies *et al.,* 2010; Jepson and Jacob, 2007)

Engagement, Dependence, Resiliency Metrics—Pacific Council
Annual engagement measure for commercial (total number of vessels with at least one landing by port; total commercial ex-vessel revenue by port; and total buyers that received at least one landing by port) and recreational fisheries (number of charter vessels per port, total rental charter trips by port). Dependence is a composite measure of vessels or revenues from particular fishery as proportion of total vessels and fishery, both commercial and recreational. Resiliency metrics are a suite of indices to collectively represent county-level resiliency and permit comparisons across communities. Indices include an industry diversity index modified from the ecosystem diversity Shannon-Weaver Index, population density, unemployment rate, percentage of population below the poverty line, and isolation of cities (see www.pcouncil.org/wp-content/uploads/1112GF_SpexFEIS_ApdxE_vulnerability_analysis_100806b.pdf)

Applications of these new vulnerability methods are improving the understanding of the scope and nature of social impacts, including enhancing opportunities for greater integration between social and economic impact analyses. For example, a 2011 vulnerability assessment of New Bedford, MA illustrated the comprehensive community-wide impact from groundfish regulations, including employment of dock-side crew, damage to public docks, and other extended social impacts (Tuler *et al.,* 2012). In addition to these social costs, there are considerable unmeasured economic costs associated with these social impacts (see the discussion of non-market and ecosystem service values in the economic sections above).

While increasingly sophisticated social impact science is being developed and documented in rebuilding FMP development, a recent Council staff review of sociocultural information collection and use concluded that "very little of the formal social impact assessment work done to date has been used in decision making" (Feeney, 2012). Others have identified slow progress toward inclusion of sociocultural analysis (Abbott-Jamieson and Clay, 2010), limited utility of qualitative descriptive social data in FMPs (Sharp and Lach, 2003), and consequently, likelihood that Councils will "see social impact assessments as more useful if those assessments were provided in a format analogous to fisheries economists and fisheries biologists' formats [i.e., quantitative]" (Pollnac *et al.,* 2006).

Further, the economic and social guidance are not well integrated, which exacerbates challenges to their integration and utilization in management, particularly as both fields of social science continue to experience methodological advancements. For example, emerging bio-economic tradeoff analyses account for economic and biological dimensions, but not other, potentially significant social implications (Daniel *et al.*, 2012).

Public participation and consideration of social impacts

As described on the Pacific Fishery Management Council web-site, "The Council process is a bottom-up process, emphasizing public participation and involvement in fisheries management. Public input is encouraged and appreciated."[24] Similar statements can be found from other Councils, since the fisheries management process is highly participatory. The mandated administrative procedures of fisheries management provide for considerable public hearings, testimony, comments, and other opportunities to hear from interested stakeholder groups. In fact, during the 2012/2013 Gulf of Maine cod quandary, the NEFMC and NMFS Northeast Regional Office increased opportunities for public comment, including a series of community meetings "to discuss commercial and recreational fishery management alternatives…[and] to provide opportunity for commercial and recreational fishermen and others to provide input to help inform what management measures we ultimately adopt…"[25] A February 1, 2012 joint statement from the NOAA Acting Assistant Administrator for Fisheries, Samuel Rauch, and NEFMC Chair Rip Cunningham on joint meetings with the fishing industry concluded, "…we know whatever measures are ultimately adopted will have economic impacts on fishermen and fishing communities. Together, we remain committed to identifying measures that will keep fishermen on the water and allow this iconic resource to continue to rebuild."[26] Thus, public participation is providing an avenue for social information to reach and potentially influence fisheries management in ways that formal, systematic and rigorous social science and impact assessments appear not to be.

Public participation infuses the social impacts and sociocultural information into the Councils deliberations, although measuring and characterizing the impact or influence of this information (i.e., operationalizing influence) is difficult and typically not done. The same Council report that found very little use of SIAs confirmed in a small survey that Council members learn of potential social impacts in a variety of ways: from informal conservations with stakeholders; stakeholder comments at public meetings; and personal perceptions, knowledge and experience. These informal sources comprised 60% of the total sources of information on social impacts, whereas FMP documents and presentations from Council staff or social scientists comprised only 20% of the Council members' sources (Feeney, 2012). Thus, socioeconomic impact information likely influences the management of an overfished stock, but may more likely do so through an informal, non-systematic, and less rigorous manner than the systematic, formal SIAs. Empirically assessing how information or input influences decision-making is an emerging field of study and has not been applied in fisheries management (see e.g., Betsill and Corell, 2008; Dür, 2008).

Nonetheless, there are considerable benefits to a participatory process that is open and transparent. These include a capacity to enhance the credibility and legitimacy of the process in the eyes of stakeholders, enhance mutual understanding, build trust, resolve or avoid disputes, increase stakeholder acceptance of management, and contribute to greater likelihood of compliance with the rules (e.g., Wilson, 2010; Pita *et al.*, 2010; Jentoft and McCay, 1995; Kapoor, 2001; Berkes, *et al.*, 2000; Berkes, 2004; 2007). At the same time, participatory processes have limitations—e.g., slower decision-making may favor the well-funded, connected and vocal stakeholders over the disadvantaged, (see, Suarez de Vivero *et al.*, 2008; Mikalsen and Jentoft, 2003).

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[24] http://www.pcouncil.org/

[25] www.nero.noaa.gov/nero/hotnews/gomcod/Gulf%20of%20Maine%20Cod%20Working%20Group%20Meeting.pdf

[26] www.nero.noaa.gov/nero/hotnews/gomcod/StatementRauchCunningham021012.pdf

Social impact of managing risk and uncertainty

The treatment of risk and uncertainty is a challenge for fisheries science, as discussed in earlier chapters, including the social sciences. However in addition, *how* risks and uncertainty are addressed, discussed and managed in the fisheries management also has an impact on stakeholders. For example, the retrospective bias in stock assessments discussed in Chapter 3 and examples of substantial and rapid fluctuations in the stock's status (referred to colloquially as whip-saw and yo-yo effects) can have a negative impact on stakeholders and to the overall climate for fisheries management. In New England, significant and rapid reductions in fishing effort from one year to the next have occurred for Georges Bank yellowtail, Gulf of Maine cod, witch flounder, pollock, Georges Bank cod, Georges Bank winter flounder, and plaice (Nies 2012). At a minimum, surprises such as these complicate management and can be a source of frustration among managers, industry, and other stakeholders. At the same time, however, they can undercut the perceived credibility and legitimacy of stock assessment science (and resulting Rebuilding Plans) among stakeholders. Here, credibility reflects whether stakeholders, such as fishermen or NGOs, perceive fisheries science and the methods of stock assessments as meeting a standard of plausibility and adequacy, whereas legitimacy refers to whether stakeholders perceive the output of the stock assessment process as unbiased and meeting the standards of fairness (Wilson, 2009).

There are emerging tools and strategies for discussing and addressing such uncertainty within a participatory process. For example, the International Council for the Exploration of the Seas' (ICES) Working Group on Fisheries Systems considered the social implications of underemphasizing and overemphasizing uncertainty. They recommended addressing uncertainty in a transparent manner, early and continuously in the fisheries decision-making process, and identified specific tools for doing so. The "pedigree analysis" is a multi-criteria, qualitative characterization of the origins and status of information and data (Dankel *et al.*, 2012); in other words, it is a systematic documented tracking of the pathways of information and data use—where information and data originate, how they is used, what assumptions are made about the information and data . An uncertainty matrix is a classification method where a panel of experts numerically rates the nature and scale of the uncertainty on several defined parameters (Walker *et al.*, 2003). Systematic, diagnostic methods such as these can be coupled with extended peer-review communities, involving multiple disciplines and stakeholder perspectives (Dankel *et al.*, 2012; and Wilson, 2010).

Uncertainty can also be pervasive in data poor situations in which managers must often use whatever data are available to construct reasonable FMPs. Managers in many data poor situations employ participatory approaches and incorporate traditional or local knowledge when considering alternative options. The Q-method has been used to identify and quantify fishers' ecological knowledge and bias (Carr and Heyman, 2012).[27] Further, data poor situations are

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[27] The Q-Method is based upon the conceptual framework of factor analysis, seeking correlations between variables. The Q-Method is concerned with individuals' viewpoints, seeking shared views or correlations across a sample of individuals and clarification on points of agreement and disagreement. Danielson, *et al.* (2010) evaluated the used of Q-Methods in evaluating public participation processes. They noted the advantages (i.e., relies on a minimal number of research participants, and is very efficient) and limitations (i.e., does not permit generalization to a population, requires considerable expertise to carry out, and results can be sensitive to sample selection)

often accompanied by limited resources for monitoring and enforcement. In these situations, participatory monitoring activities have been constructive in managing the risk and uncertainty (Bently and Stokes, 2009; Parma *et al.,* 2003).

One strategy that aims to increase the transparency of the stock assessment modeling has been participatory modeling—it has been more commonly applied in nutrient load and watershed management, although has been explored in a fisheries context. Röckmann *et al.,* (2012) illustrated the potential of participatory modeling in stock assessment—facilitating and structuring dialogue about uncertainty and the quality of the state of knowledge among scientists and stakeholders, enhancing scientific understanding, and increasing the perceived legitimacy of the process among stakeholders. At the same time, participatory modeling became the effective tool for openness and transparency in joint problem solving, but less effective as a sophisticated modeling output.

Well designed collaborative research methods have shown to directly enhance the credibility and legitimacy of the resulting science, along with the potential to increase acceptability of management actions; produce greater mutual understanding and trust among partners; and opportunities to integrate diverse sources of knowledge about the coastal and marine environment (Hartley and Robertson, 2006; 2008; and 2009; Conway and Pomeroy, 2006; Johnson and van Densen, 2007; St. Martin *et al.,* 2007; National Research Council, 2004; and Heyman, 2011)

Fisheries management and rebuilding

The development and success of Rebuilding Plans cannot be understood fully outside of the broader context of fisheries management within which they are implemented. For example, the management institutions and approaches used to control harvest under Rebuilding Plans affect the incentives facing those who fish. These in turn can affect fishing behavior and both the biophysical and socioeconomic outcomes of rebuilding. This section discusses potential interactions between the ways that fishery management occurs (and has occurred) in the US and the outcomes of Rebuilding Plans, with particular emphasis on incentives for specialization and attendant impacts on the short-term costs of rebuilding.

The historical paradigm for managing fisheries in the U.S. has been to allocate a portion of a species' total allowable catch to fishing sectors (usually defined by gear type or size of fishing vessel) and to accompany this allocation with additional controls on fishing locations, seasons, technology, and entry. In the west coast groundfish fishery, for example, the total allowable catch for sablefish is allocated to a trawl sector, fixed gear sector, and open-access sector (smaller fishing vessels) and there are also entry and gear restrictions and no fishing zones (PFCM, 2011a). Scholars denote these as regulated open-access or limited-entry fisheries (Hanna *et al.,* 2000; Wilen, 1985).[28] For fish stocks and regions that have avoided overfished and overfishing status, the current approach has received some measure of success at least from a biological perspective. However, the incentives created by regulated open-access or limited entry regulations also affect the economics of fishing and the resilience of coastal communities—the

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[28] Whether a fishery is regulated open-access or limited entry is based on the presence or lack thereof of controls on access, where the latter have, e.g., license programs. Regulated open-access fisheries are open for entry, but the fishing enterprise operate under a set of regulations (e.g., closed seasons, areas, gear restrictions, catch totals).

implications of which are becoming clearer over time (Tuler *et al.,* 2012; St. Martin and Hall-Arber, 2008). A declaration that a fish stock is overfished implies that past management approaches (reflecting the historical paradigm) have failed to maintain stocks and economic returns at desired levels (see, e.g., Sanchirico and Wilen 2007; World Bank 2009), and have thereby stressed the local economic and social fabric of fishing communities (Tuler *et al.,* 2012; Georgianna and Shrader, 2008; Hall-Arber *et al.,* 2001; Portman *et al.,* 2009).

The potential economic and social impacts from regulated open-access or limited-entry commercial fisheries are well-known as are the potential solutions (see, e.g., Wilen 1985; Homans and Wilen 1997; and World Bank 2009). For example, fishermen have been observed increasing investments in inputs (e.g., size of boat, engine horsepower, sonar, type of gear) as other inputs become more constrained due to regulation (see, e.g., Wilen, 1985). This type of behavior while economically justifiable for any given fishermen increases the costs of fishing, and reduces profit margins and their ability to mitigate shocks to revenue in any year. Rather than discuss all the well-known effects of these regulatory institutions, we focus on three here due to their particular relevance to the broader socioeconomic outcomes of rebuilding: (1) the inability of fishers to adjust their fishing practices throughout the year; (2) the lack of diversity of the fishing operations, and (3) the impacts on community resilience.[29] We note, however, that empirically disentangling the impact of one effect from the others is difficult.

In regulated open-access fisheries that experienced overfishing, the typical regulatory response was to reduce fishing mortality and hence catches of commercial and recreational fishers. In the past, these goals were often achieved at least in part using “input controls” that restrict how, where and when a fisher is able to fish; these controls constrain fishing operations, for example with shorter seasons, reduced fishing areas, or reduction in gear efficiency (Homans and Wilen, 1997). If estimates of the fish stock abundance continued to show downward trends, these constraints were typically increased to prevent the stock reaching an overfished status.

Regardless of whether additional input controls on the fishing operation effectively addresses overfishing, the constraints reduce the ability of the fishing operation to adapt (e.g., timing and spatial fishing location) in response to changes related to a Rebuilding Plan. The implication is that the potential for behavioral adaptations to mitigate the short-run economic costs associated with further reductions in fishing mortality due to rebuilding is lower, everything else being equal. Thus common input-control approaches to fisheries management can exacerbate the short run costs associated with rebuilding, because these controls restrict the adaptation possibilities available to fishers.

The second economic and social implication associated with regulated open-access or limited-entry fisheries is that it institutionalizes specialization in fishing operations. For example, restrictions on allowable gear types, combined with non-transferrable licenses associated with fish stocks can restrict fishers ability to switch between stocks. Specialization

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[29] We focus here on the regulatory institutions that existed at the time the stocks were classified as overfished. In the United States, no fish stocks were classified as overfished that were under an individual fishing quota management system at the time of classification. This does not mean, however, that more rights-based approaches (e.g., catch shares) are immune to creating similar specialization, e.g., an individual quota allocation might be restricted to a particular species in a particular location and sometimes with a particular gear type. On the other hand, rights-based approaches can and often do reduce other constraints on the fishing operation in regulated open-access fisheries, such as short fishing seasons and there is nothing inherent in their design to have such restrictions in place (see, e.g., Sanchirico *et al.,* 2010 for a discussion of programs in New Zealand, Australia, Iceland, Canada, and the U.S.).

has economic advantages, for example, it can reduce the costs of fishing or increase fishing revenue. In addition, there might be ecological advantages if specialization results in the use of more selective fishing gear and therefore reduces bycatch (see, e.g., Garcia *et al.*, 2012 for arguments against increasing selectivity in targeting).

Specialization also results in a lack of diversity in fishing portfolios (Kasperski and Holland, 2013) and is often accompanied by large capital investments in fishing technology suitable for a limited number of stocks. The lack of diversity and highly capitalized fleets are not necessarily an issue when a fishery is healthy and catches are controlled. However, if the fishery is driven to overfished status (either due to fishing or environmental factors), the economic costs from reductions in catch are likely to last longer and be greater than if the fishing operations were less specialized and capitalized (i.e., allowing fishers to adapt more successfully to additional constraints on the harvest of particular stocks). In fisheries, a more diverse fleet could mitigate some of the costs associated with rebuilding by focusing effort on other species in the same area or other fishing regions. This type of behavior is currently restricted by regulatory approaches that reduce the flexibility of fishing operations.

Third, the effects of institutionalizing specialization on the fishing sector can also ripple throughout the community and can potentially increase other community costs due to rebuilding. Specialized and highly capitalized fishing fleets require, for instance, specially trained processing and support industries, and the overdependence of these industries on a few fish stocks increases the risks for large economic and social downturns in coastal communities if the stocks become overfished and rebuilding plans are implemented. For example, a 2011 vulnerability assessment of New Bedford, Massachusetts illustrated the comprehensive community-wide impact from groundfish regulations (Tuler *et al.*, 2012). The fisheries management actions contributed to a reduction in the fleet size, with corresponding decline of support services—less fuel, ice, and repair services. The function and employment of lumpers (crew who unload fish at that dock) changed; with the regulatory constraints on when vessels could leave or return to dock and the fewer overall vessels working, lumpers were accepting work whenever it was available, and making themselves available 24 hours a day, seven days per week, including for back-to-back boat unloading. More vessels were remaining at dock in New Bedford, resulting in great dock crowding, which in turn made it more difficult for fishermen to conduct repairs and affected the condition of vessels and gear. Crowded docks also contributed to more damage to the dock, including increased spills of hazardous materials.

Expanding flexibility through management measures

There are a number of options that the Councils might pursue in conjunction with implementing a Rebuilding Plan to introduce more flexibility for fishermen and fishing communities. For example, the U.S. west coast groundfish fishery individual fishing quota system (catch share) allows risk pools, which are ways for fishermen to mitigate the costs associated with very low bycatch levels of canary rockfish (Holland, 2010c; Holland and Jannot, 2012). Introducing additional flexibility, however, will not mitigate all of the near term costs and will not alleviate the potential for necessary reductions in the size of the fishing fleet after rebuilding has occurred.

Other ways to improve flexibility, include allowing the conversion of sector-based individual fishing quota allocations from one gear to another (e.g., mobile gear quota for west coast sablefish converted to fixed gear), removing or lessening season length or area restrictions, allow conversion of either quota for one species to another or days at sea for one species for another (not necessarily at a 1:1 ratio). For example, Iceland permits the conversion of quota for one species to another (e.g., cod to Greenland halibut) within its individual fishing quota system (Sanchirico *et al.,* 2006). To avoid significant overages in any one species' total allowable catch, Iceland uses trading ratios and caps on the amount of species conversion that an owner can undertake within the season.

Unlike disaster relief or vessel buy-back programs where the fishermen receive direct compensation, these measures attempt to directly address the flexibility (or lack thereof) of fishing operations by providing opportunities for fishermen to mitigate some of the costs associated with rebuilding by changing their behavior (fishing for different species, times, and locations). In many respects, the added flexibility from these changes might improve the resilience of the communities and fishing industries to future shocks whether or not they come from a declaration of an overfished stock.

Government mitigation measures

By law, overfished stocks require a rapid management response, even if the socioeconomic impacts are difficult to measure, predict, or are believed to be severe. Fishing communities feel these impacts and respond to perceived and real harm (e.g., seek relief through the court system, appeal to state and federal elected officials). This social response is an observed human behavior in many contentious resource management processes and not unique to fisheries—e.g., Spotted Owls (Noon and Murphy, 1994), California water resource management (Hundley, 2001), and wolf management and restoration (Nie, 2001). Fishery managers have sought to consider mitigation opportunities to alleviate potential impacts while developing Rebuilding Plans.

Managers and elected officials have used disaster relief packages, Congressional earmarks, and other mitigation measures to address the social and economic displacement of fishermen and fishing dependent communities once a fish stock is declared overfished. These methods differ in a number of dimensions, including whether the measure operates within or outside the FMP and whether the measure is implementable within the Council or requires Congressional approval. For example, government responses that operate outside of Rebuilding Plans include declarations of fisheries disasters, vessel buyback programs, loans and direct funding opportunities, collaborative research, data reporting and monitoring systems, and other mechanisms (Hanna, 2010). Additionally, U.S. Congress can and has acted on its own to provide various forms of financial relief, mandate or direct specific NOAA action, or support other stakeholders directly.

Table 6.4 contains a small sample of mitigation measures for a number of rebuilding fisheries, including examples taken within and others outside the FMP itself. In the Gulf of Maine cod, for example, recent measures used to partially mitigate the socioeconomic impacts of an otherwise-mandated reduction in harvest included an invocation of section 304(e)(6) as justification for one-year interim action to reduce rather than end overfishing on the stock, a federal disaster declaration, and a transfer of 2012 carryover quota to 2013. A request for a second one-year interim action was declined by the NOAA Regional Administrator in 2013. Another example is the emergency rules adopted in the South Atlantic red snapper fishery that allow recreational three-day weekends and commercial mini-seasons, temporarily lifting the harvest moratorium in response to lower-than-expected discard mortality (South Atlantic Fisheries Management Council, 2012).

The effectiveness of many of these ad hoc measures has been questioned in the U.S. and internationally (Holland *et al.*, 1999; Minnegal and Dwyer, 2008). While rigorous socioeconomic research and findings in rebuilding has been predominantly utilized in the context of impact assessments, it has not been gathered to inform and guide the design and implementation of mitigation options (e.g., more precisely target the relief to the communities more impacted, with the greatest vulnerability and least resilience). This presents a substantial opportunity for the application of social and economic sciences in fishery rebuilding.

TABLE 6.4: Illustration of Measures Taken to Mitigate Socioeconomic Impacts of Rebuilding

Rebuilding FMPs:	***Mitigation measures:***
PFMC: Canary Rockfish	trawl vessel buyback program removed 34% of vessels with groundfish permits in 2004 (Hanna, 2010)
NPFMC: Bering Sea Snow Crab	Federal relief money for Alaska coastal communities (NPFMC, 2000) Federal loan program to buy out vessels (Department of Commerce, 2004)
NEFMC: Gulf of Maine cod	$30M to assist industries and communities to develop alternative fisheries, improve fishery infrastructure, provide job training. (Amendment 5, NEFMC, 1993) $22M voluntary vessel buyback program, (Wang and Rosenberg, 1997) Congressionally-mandated cooperative research (Hartley and Robertson, 2006; Hanna, 2010) $16M to assist industry transition to sector management and $10M to develop data reporting and fishery monitoring system (NOAA, 2009) Allowing 2012 year’s groundfish quota to carryover to 2013 in order to “help mitigate some of the economic impact on the fishing industry.” (Bullard, 2013)

SUMMARY

The primary focus of MSFCMA's rebuilding mandates on biological conservation contributes to tensions among managers, fishermen, elected officials, and other stakeholders, particularly when these mandates constrain flexibility to address socioeconomic consequences. Several factors can contribute to these tensions, including divergences between expectations and reality in rebuilding trajectories, lack of understanding of the social and economic context within which rebuilding plans are implemented (because of a lack of data and analysis), and the disconnect between the participatory process—where socioeconomic impacts are often discussed but not systematically assessed—and the FMP outcome. Unexpected (or concern for potential) social and economic outcomes are often addressed outside of an FMP through federal disaster declarations, Congressional initiatives, and other ad hoc efforts. These efforts are rarely informed by social science and thus may not achieve their full potential or intent (e.g., ineffective buyback programs, financial assistance not targeted at communities with largest impacts, or unintended consequence could be to make the funds to cover next disaster declaration even larger). Despite evidence of success in the biological rebuilding of many fish stocks, the social and economic dimensions of rebuilding (including both behavioral drivers and consequences) cannot be taken for granted as deterministic functions of fisheries stock size.

Current understanding of the socioeconomic consequences of rebuilding is limited by a lack of detailed analyses conducted after Rebuilding Plans have been implemented (i.e., ex post). While some studies provide rudimentary ex post assessments of the economic and social impacts of Rebuilding Plans (e.g., by measuring changes in fishing revenues that have occurred when catch increases due to rebuilding, such as NRDC, 2013), there is an overall dearth of rigorous ex post assessments of rebuilding plans across economic and social dimensions. The lack of retrospective socioeconomic analysis leads to uncertainty over the net economic and other social benefits of rebuilding that have been realized, in contrast to those that are predicted. As discussed above, economic and social analyses of Rebuilding Plans (e.g., as part of Environmental Impact Statements, Regulatory Impact Review, and the Social Impact Assessment) are only required prior to implementation. There is no requirement for NMFS or others to conduct follow-up, retrospective or ex post economic or other social analysis.

Challenges to measuring impacts ex post include the lack of data (as discussed above) and the difficulty of establishing what would have occurred in the absence of a Rebuilding Plan (i.e., counterfactual conditions). Measuring the impacts from rebuilding, for instance, requires disentangling changes in net benefits due to a single Rebuilding Plan (reduction in F and T_{REBUILD}) from changes that might have occurred due to other, exogenous factors (e.g., habitat change, economic conditions) and other endogenous factors (e.g., change in regulatory structure). For example, the recent shift to sector management in New England will confound any analysis of the economic gains and losses associated with the rebuilding plan, because the two occur simultaneously. The lack of ex post assessments of regulations is not unique to fish stock rebuilding – other agencies such as U.S. EPA have discussed the general lack of ex post economic analyses (e.g., U.S. EPA National Center for Environmental Economics, 2012). Without rigorous ex post analyses, however, it is impossible to quantify the net economic or other social benefits that have been realized due to U.S. Rebuilding Plans.

FINDINGS

6.1: Compliance with MSFCMA requires that economic and social considerations for rebuilding plans are contingent on biological mandates being met. Rebuilding Plans that do not meet these biological mandates cannot be adopted, even if doing so would improve projected socioeconomic outcomes.

6.2: The requirement to rebuild within 10 years, whenever possible according to the biology of the stock, reduces the flexibility to adapt rebuilding plans according to economic and social considerations.

6.3: Socioeconomic considerations influence the management of overfished stocks through the public participation process (e.g., public testimony to Councils regarding the magnitude of socioeconomic impacts). Stakeholder participation and concerns regarding the impacts of Rebuilding Plans can also result in ad hoc mitigation measures (e.g., disaster relief assistance) that operate outside the fishery management process. The design of these measures is not fully informed by social science, and their implications on other fisheries and on the long-run social and economic viability of coastal communities are not fully known.

6.4: The mandate that rebuilding targets are met with a certain minimum probability, along with the requirement to utilize most current stock assessments may lead to marked changes to Rebuilding Plans based on new data and/or models as they become available. These adjustments can cause economic and social impacts, potentially both positive (e.g., sooner rebuilding and increases in allowable catch) and negative (e.g., rebuilding behind schedule and decreases in allowable catch). Although, these adjustments may reflect best available science, they nonetheless can influence the perceived credibility of the science among stakeholders.

6.5: The guidance on economic and social methods to be used in the analysis of the alternative harvest control rules is consistent with best practices, but implementation is variable across plans and between Councils.

6.6: The treatment of uncertainty is not integrated across the ecological, economic, and social dimensions of Rebuilding Plans. Given the challenges of addressing the many types of risks and uncertainty in fishery management, the cumulative risk trade-offs are not well understood. Consequently, it is not clear whether the necessary precaution (or too much precaution) is being applied.

6.7: In considering different management alternatives for meeting rebuilding targets, the information provided to the Councils is most relevant for short-run economic impacts on the commercial and recreational fishing sectors and local communities. Although models may forecast socioeconomic outcomes over longer time periods, the simplifications and assumptions of these analyses limit their relevance to longer-term forecasting.

6.8: When evaluating socioeconomic outcomes of Rebuilding Plans, economic impacts on commercial, recreational, and related fishing industries are the primary focus of the Councils.

The analysis of different management options rarely quantifies impacts on nonmarket ecosystem services or non-fishery benefits.

6.9: Retrospective reviews of the broader socioeconomic impacts of Rebuilding Plans are rare, at least partially due to data availability. These socioeconomic impacts include changes in the structure of commercial fishing sector, economic returns, recreational values, fish processing industry, and culture of fishing in communities.

6.10: Methods exist and innovations are emerging in economic and social science approaches to characterize the breadth of economic and social impacts of Rebuilding Plans and factors that contribute to the success of these plans, although they have not yet been broadly applied, tested and refined to meet these information needs.

6.11: The nature of fisheries management, including in the United States, can lead to situations that exacerbate the economic and social impacts of meeting rebuilding targets by institutionalizing the specialization of the fishing industry (including fishing fleets, processing, and related support businesses). These constraints reduce the ability of the fishermen and community to absorb some of the costs associated with curtailing catches.

7

LOOKING FORWARD

Introduction

Our purpose in this chapter is to offer some ideas based on observations from the previous chapters on issues for consideration as part of long-term strategic planning for fisheries rebuilding and for potential application to ecosystem-based fisheries management (EBFM). We begin with the overarching issue of how to balance the tradeoff between the current prescriptive approach and an approach that would allow rebuilding plans to be more tailored to the specific circumstances of the fishery, the environment, and the scientific information available. We then cover seven topics that are directly or indirectly related to the overarching issue of prescriptive versus flexible approaches. These topics are: defining success of rebuilding plans; rebuilding and EBFM; rebuilding time frames; model predictions, projections, and data and knowledge limitations; mixed-stock fisheries; the role of biological science and socio-economic factors; and communication.

Overarching Issue: What is the Best Balance between Prescriptiveness and Flexibility? (Findings 2.2, 2.3, 3.10, 5.1, 6.1, 6.4)

The rebuilding approach, established by the current legal framework and MFSCMA guidelines, is highly prescriptive. Under this framework and guidelines, management of individual stocks are based on specified biomass thresholds and targets, a fishing mortality limit, catch reductions in consideration of various types of uncertainty, accountability measures, and a specified maximum time period within which rebuilding must occur with at least a 50% probability. The guidelines for implementing rebuilding also stipulate the process by which scientific advice is formulated and conveyed. Our comments on the prescriptive approach relate to the specifics of the current guidelines for rebuilding.

The benefits of taking a prescriptive approach are that it should act to reduce delays in taking corrective action when stocks become depleted; and provide clear specification of the steps involved in plan formulation, the required rebuilding targets, and how to track progress towards rebuilding. The prescriptive approach also limits the potential use of short-term socioeconomic costs as an argument to justify delay of rebuilding plans that would, if successful, provide long-term socioeconomic benefits. The prescriptive approach also can (but is not guaranteed to) ensure that scientific advice is followed. Clear rules and steps to follow should improve accountability because of clear tractability (e.g., identification of targets), and thus more straightforward communication of the status of fish stocks. The disadvantage of the highly prescriptive approach is that, by definition, it leaves little room for flexibility or innovation (e.g., use of alternative stock-specific reference points), and precludes tailoring rebuilding plans to the specifics of each stock and its fisheries. Further, satisfying the specific demands established by the rebuilding guidelines may divert attention from the broader goals of EBFM.

The tradeoff between flexibility and prescriptiveness within the current legal framework and MFSCMA guidelines for rebuilding underlies many of the issues discussed in this chapter. The present approach may not be flexible or adaptive enough in the face of complex ecosystem and fishery dynamics when data and knowledge are limiting. The high degree of prescriptiveness (and concomitant low flexibility) may create incompatibilities between single-species rebuilding plans and EBFM. Fixed rules for rebuilding times can result in inefficiencies and discontinuities of harvest-control rules, put unrealistic demands on models and data for stock assessment and forecasting, cause reduction in yield, especially in mixed-stock situations, and de-emphasize socio-economic factors in the formulation of rebuilding plans. The current approach specifies success of individual rebuilding plans in biological terms. It does not address evaluation of the success in socio-economic terms and at broader regional and national scales, and also does not ensure effective flow of information (communication) across regions. We expand on each of these issues below and discuss ways of increasing efficiency without weakening the rebuilding mandate.

Defining success of rebuilding fisheries
(Findings 2.1, 3.5, 3.6, 3.8, 5.9, 6.1, 6.2, 6.3, 6.9)

Our evaluation of rebuilding plans (Chapter 3) focused quantitatively on biological metrics, consistent with current legal mandates. Beyond those, there is a lack of consensus concerning what, specifically, is implied by overall (not just biological) rebuilding success. While this is a basic and simple question, determining the answer can be quite complicated. Ideally, rebuilding plans would balance the expected socioeconomic trade-offs with reductions in fishing pressure to rebuild stocks in a given time period, and no stocks would be classified as overfished in the future. However, this is unlikely to happen. So what is a realistic basis for judging the overall performance of individual rebuilding plans and what is a realistic standard for overall success at the national level?

Each rebuilding plan has an acknowledged possibility of failing to achieve a given rebuilding target by the specified time because rebuilding plans are acceptable if their associated probability of rebuilding, estimated at time of adoption, is 50% or greater. Even if rebuilding plans were designed to have a higher success rate (e.g., 90%), some of the plans would not achieve their objective on schedule. Stocks may rebuild faster or slower than expected because of environmental influences. Thus, even from just the biological perspective, nationwide "success" is possible with some failures of individual stocks to rebuild by the agreed time or even ever.

The approach to evaluating the success of rebuilding plans should examine the portfolio of stocks, and quantify how many are rebuilding ahead of their target dates and how many are rebuilding slower than expected in comparison to the probability of success they were designed to achieve. In addition, it should distinguish cases in which there was a failure to reduce fishing mortality as intended from those in which stocks are failing to show signs of rebuilding in spite of reducing fishing mortality(see Chapter 3). Our analysis of rebuilding plans indicated that there were only a few stocks for which biomass did not increase when fishing mortality was effectively reduced.

At the national level, there will always be some stocks classified as overfished and in need of rebuilding for several reasons: rebuilding plans are not designed to rebuild stocks on schedule with certainty (certainty is not attainable even with zero fishing), some stocks are incorrectly categorized as overfished because of scientific uncertainty (see Chapter 3), and changes in environmental and ecological conditions preclude rebuilding to targets set based on historical stock levels.

In addition to the biological benchmarks, the socioeconomic impacts of rebuilding are also an important component in judging success. Presently, only a subset of the total socioeconomic costs and benefits of rebuilding are typically quantified or systematically considered when evaluating rebuilding plans, the evaluation of these costs and benefits varies among the regional councils, and analyses often involve simplifying assumptions that limit their long-run applicability (Chapter 6). A national discussion on defining success in rebuilding plans, especially defining suitable and quantitative measures of performance, would help clarify overall goals of rebuilding, and enable the progress of rebuilding individual stocks to be viewed in a more general context than stock-by-stock. How should biological benchmarks and socio-economic factors be considered in overall success? While consensus is unlikely, regional discussions that feed into a national discussion could lead to greater understanding of stakeholder views and perspectives, which in turn would improve the perceived relevance and credibility of the science and the legitimacy of the decision-making process.

Rebuilding under ecosystem based fisheries management
(Findings 2.1, 5.1, 5.3-5.6, 6.8, 6.10)

The present focus of rebuilding plans on a stock-by-stock basis with MSY-based targets has advantages (e.g., clear benchmarks and tractability). In addition, the use of output controls, such as catch limits, provides a direct and measureable mechanism for controlling fishing mortality, albeit with uncertainty. However, how such stock-specific rebuilding measures fit within EBFM is unclear. It is conceivable that the focus on stock-specific rebuilding plans that rely on output controls can, in some situations, be difficult to mesh with, and even be detrimental to, EBFM objectives. EBFM is still only conceptually defined, although progress is being made to move from proof of concept to operational use (Chapter 5). Formal approaches to evaluating management strategies (Chapter 5) that are often based on multi-species or ecosystem models offer one promising approach for exploring the long-term performance of rebuilding plans and strategies beyond the responses of individual stocks. However, at this time, these highly parameterized multi-species and ecosystem models are essentially “best guesses” whose performance and skill are uncertain. Use of such analyses will require additional effort for their results to play a stronger role in strategic planning and informing specific rebuilding situations. There remains a gap between the approaches of stock-specific rebuilding and EBFM.

Rebuilding time frames
(Findings 2.2, 3.3, 3.7, 4.2, 4.6, 5.9, 6.2)

The idea of a simple fixed rule for determining the maximum number of years for rebuilding is, in principle, an effective way to ensure rebuilding occurs at a reasonable pace, but also can create inefficiencies in practice. Having a rule for determining the maximum time horizon associated with acceptable rebuilding plans clearly reduces the possibilities for delaying fishing reductions into the future.

However, there are also disadvantages to fixed rules for specifying the maximum time for rebuilding. First, problems may be associated with the specific formulation of the rule. The 10-year rule presently in place uses T_{MIN} to determine a minimum possible rebuilding time; this is useful as it takes account of initial stock condition and expected productivity. However, the way the rule determines T_{MAX} has a discontinuity at 10 years (see Fig. 4.1), which can lead to discontinuities in target dates for recovery (10 years to many decades) with potentially only small changes in estimates of stock size from assessments.

Second, a fixed maximum time for rebuilding also hinders the consideration of socio-economic tradeoffs, especially when the range of acceptable rebuilding periods (i.e., from T_{MIN} to T_{MAX}) is narrow. The allotted rebuilding time can lead to substantial increases in rebuilding costs if the incremental additional costs from rebuilding are sensitive to the rebuilding schedule. As described in Chapter 6, it is sometimes possible for modest changes in a rebuilding schedule to have non-trivial effects on net social benefits; such adjustments are often precluded under current requirements. Abrupt changes in management measures can have real economic and social impacts on communities, and influence the perceptions and attitudes of stakeholders and managers.

Finally, a fixed time to rebuilding can also be problematic when rebuilding is faster or slower than expected, causing over-reaction and misinterpretation of the causes. Rebuilding faster than expected can lead to pre-mature demands to lessen rebuilding measures and therefore the rate of rebuilding. Delays in rebuilding, on the other hand, can lead to severe reductions in target fishing mortality in an effort to achieve the rebuilding target by the pre-specified date. The reasons for rebuilding occurring slower than expected include unexpectedly low recruitment, an ecosystem change, or failure to reduce fishing mortality due to imprecise or inaccurate science, or to catches exceeding desired levels (fishing mortality is higher than the target level). When recruitment is below expectations (e.g., due to unfavorable environmental conditions), a control rule based on maintaining fishing mortality at some fraction of F_{MSY} may be more efficient than one that forces ever more severe controls to try to keep rebuilding on schedule; such a control rule could be formulated to ensure achievement of the goals of rebuilding as more favorable conditions return.

When discussing the goals and design of rebuilding plans in the future, the benefits and costs of introducing more flexibility in determining the time to rebuild should be considered so that new scientific information and socioeconomic tradeoffs can be more fully accounted for in rebuilding and community mitigation. Determining when and how within the rebuilding process to introduce additional flexibility that properly accommodates biological and socio-economic factors is a challenge. Experience from other countries (Chapter 3) indicates that legal mandates that are similarly strong in demanding reductions in fishing mortality as in place now in the U.S., but that allow greater flexibility in setting the time horizon for rebuilding can be effective. A caveat to the applicability of the international examples is that there are other aspects of the fishery management systems, such as the role of industry interests in decision-making, that differ among countries, which make direct comparisons difficult.

Model predictions, projections and data limitations
(Findings 2.1, 3.5, 3.9, 4.1, 4.4, 4.7, 4.8, 5.1, 5.5. 5.6-5.8, 6.3)

The situation of many stocks for which there are insufficient data and information to allow for model-based projections will remain a challenge to providing management advice in general, and for designing rebuilding plans in particular. Most of our analyses and commentary in Chapters 2 through 6 focused on the stocks for which projections of stock size and estimation of fishing mortality and biomass-based benchmarks are possible. However, many stocks can be characterized as data-poor, implying thatstock projections cannot be conducted and thus benchmarks for either status determination or rebuilding cannot be established. Indeed, the stock status of over half of the nation’s 479 managed stocks or stock complexes was unknown at the end of 2012 (Chapter 3).

Knowledge may be limited even for stocks for which data are abundant. For example, there may be several plausible alternative models to explain the data but they can still produce different predictions. How to deal with these data-poor and knowledge-poor stocks, both in assessing their status and then, if appropriate, in formulating rebuilding plans, has been a long-standing challenge. The present implementation of rebuilding in the MFSCMA can, in some situations, put demands on the available data, information, and modeling that are beyond the current capabilities and therefore introduces high uncertainty.

When data-poor stocks (and perhaps for some knowledge-poor stocks) require rebuilding, spatial and habitat-based approaches (e.g., marine zoning including MPAs), with empirical rules to adjust harvest controls in response to abundance trends and demographic indicators as well as ecosystem-level indicators (e.g., prey abundance), provide an alternative, less data-intensive strategy. When data are too limited to perform stock assessments and estimate biomass and fishing mortality with sufficient confidence, demographic indicators can be used to adjust management controls to ensure that fishing rates are reasonable and precautionary and that rebuilding is progressing. Ecosystem, habitat, and demographic indicators may be able to be used as a practical alternative, or in conjunction with other fisheries information, within a more flexible rebuilding protocol.

The current approach to rebuilding, which requires projections of stock biomass many years (often decades) into the future, results in interpretation of stock biomass projections to a degree that is on the edge or beyond the capability of current models. Stock biomass estimates and projections can vary greatly in response to alternative plausible assumptions (models) and parameter values used in simulations. The stable dynamical behavior often assumed in stock assessment models (e.g., spawner-recruit relationships) can result in an inability to replicate the nonlinear population dynamics observed in fisheries data, and therefore give inaccurate and uncertain model projections.

While the quality of the data and models vary among stocks, data and modeling capabilities are rarely sufficient to truly interpret model projections as forecasts of actual stock size to be expected over the next decades. Rather, model projections, especially in situations of alternative plausible models, can be effectively viewed as tools to explore performance of rebuilding strategies in scenario mode (comparative or relative outcomes), with an emphasis on adequate feedback and adaptive responses, rather than on predictions of future biomasses. However, use of model projections in these ways is insufficient to truly fulfill the requirements of the current approach, especially in terms of biomass-based metrics.

Increased flexibility could promote more rapid consideration and adaptation of new methods and potentially allow for the design of more robust rebuilding plans for knowledge-poor stocks. One idea is to replace the present rebuilding strategy based on biomass benchmarks over a defined time horizon with an equally rigorous approach based on controlling fishing in the near term (i.e., years not decades) and using the short-term projected *directions* of stock change or fishing rates relative to F_{MSY}. Projections of fishing mortality relative to F_{MSY} tend to be more robust than projections of biomass relative to biomass reference points (Chapter 4), although this assertion would need to be confirmed for any particular stock.

Short-term forecasts can be made using age-structured models and statistically-based methods, especially in cases where stock dynamics are not dominated by unpredictable recruitment or highly variable mortality; although general real-time validation of forecast performance is largely lacking. New empirical modeling techniques are becoming available that can be tailored to management metrics that operate with variable recruitment and mortality, and that focus on the inherent non-linearities in fish population and community responses to environmental and biological changes and short-term rates of change (e.g., Boxes 5.1 and 5.2). While no modeling technique is perfect, a shift toward rebuilding that considers more shorter-term forecasting and heavier reliance on fishing-mortality-related metrics could enable rebuilding and be more robust to some aspects of model uncertainty.

Mixed stocks
(Findings 4.5, 5.5, 5.6, 5.7)

Rebuilding and mixed-stock fisheries will continue to be challenging due to the need to weigh trade-offs among species. While the mixed-stock problem was acknowledged in the current guidelines of the MSFCA, the committee is unaware of any cases where the “Mixed-Stock Exception” has been applied in rebuilding plans.

Attempting to deal with the mixed-stock problem will require analyses and modeling of fisheries and economics data to identify appropriate solutions, and the flexibility to apply mixed-stock exceptions (where applicable). One challenge is the development of mixed-stock fisheries models that allow for evaluation of trade-offs. Such models have been proposed (Chapter 6), but they need further evaluation and testing. Use of the predictions from these models is limited by the availability of data on all of the fish populations, and on their biological and technical interactions, as well as the relevant socio-economic data, but more data are becoming available.

A second challenge is to design operational regulations and incentivize fishing practices that adequately protect weak stocks while providing fishing opportunities for healthy stocks. The MSFCMA requires that fishing mortality be kept below F_{MSY} for all stocks and that time-constrained rebuilding plans be implemented for all overfished stocks. This constraint requires forgoing benefits to achieve rebuilding goals for even the most insignificant stocks in terms of their value or ecosystem function. Such precaution can be needed when extinction is an issue. However, there is usually a wide range of choices between F_{MSY} and the rate of fishing mortality that increases risk of extinction to a noteworthy extent, but these choices in fishing mortality are not allowed under the MSFCMA. Rebuilding plans could be designed to allow harvesting of healthy stocks in mixed-stock situations, while preventing weak stocks from being driven to unacceptably low abundance. .

Role of biological science and socio-economic factors (Findings 6.1, 6.3-6.7, 6.9, 6.10)

The net economic and other social benefits of successful rebuilding are often (though not always) positive in the long run. There is also often a time lag between rebuilding fish stocks and rebuilding the fisheries that depend on them. Rebuilding plans necessarily involve a reduction in fishing pressure and the rebuilt fishery will require less fishing capacity than before. Only a subset of the total socioeconomic costs and benefits of rebuilding are typically quantified or systematically considered when evaluating rebuilding plans, the evaluation of these costs and benefits varies across Councils, and analyses often involve simplifying assumptions that limit their long-run applicability (Chapter 6). The existing rules (e.g., 10-year rule) and guidance, along with the lack of application of the Mixed-Stock Exception, prevent consideration of possible harvest options that could otherwise improve socioeconomic outcomes. This situation contributes to stakeholders contesting rebuilding plans due to the perceived and real socioeconomic impacts and to stakeholders appealing for and securing mitigation measures from Congress, NOAA, and other stakeholders.

Given the socioeconomic tradeoffs, a broader dialogue is needed as to how and when socioeconomic information should be introduced into the process of developing rebuilding plans. The deliberations would lead to greater mutual understanding among industry, managers, scientists, NGOs, and other stakeholders, improve transparency in decision-making, and enhance opportunities to apply rapidly advancing social science methods to select management and mitigation measures within rebuilding plans. Ultimately, explicit consideration of socioeconomic impacts is critical because the current process of evaluating socioeconomic outcomes contingent upon the prior establishment of biological parameters precludes consideration of potential rebuilding plans with superior socioeconomic properties (e.g., greater benefits or smaller costs).

The challenge is how to appropriately include socio-economic considerations while maintaining the tractability and mandate for action that many consider to be a positive aspect of current rebuilding plans. Coupled human-natural systems models are starting to be developed that could eventually play a role here (Chapter 6). Systems-based approaches, and formal models of decision-making under uncertainty, are some of the options that can help promote a more transparent, deliberative process for developing rebuilding plans, thereby encouraging better integration of biological and socioeconomic considerations.

A second challenge is how to provide scientific advice in situations of alternative models. The SSC and other review bodies can provide singular advice when there is a best model, the alternative models generate similar results, or it is scientifically appropriate to combine the results from multiple models. Ideally, a weighting could be applied to such an average based on rigorous out-of-sample testing. However this kind of validation is rarely practiced. In situations where there is no scientific basis for giving singular advice according to one model and not the other (or others), SSCs and other review bodies should present the results from multiple approaches (ideally including socio-economic aspects) to the managers. The implications of alternative management decisions on the different models considered plausible should be included in the advice, and be part of the deliberations of the managers as they weigh biological and socio-economic considerations.

Need for effective communication and stakeholder engagement
(Findings 6.3, 6.4, 6.6)

Finally, as is always the case with controversial issues when science and policy intersect, the importance of clear communication and effective stakeholder engagement is critical and the search for methods for effective communication must continue. Collaborative research and monitoring among fishers, managers, and scientists offers one avenue for communication and engagement (Chapter 6). Transparency in the capabilities of the modeling used for developing rebuilding plans, how and what sources of uncertainty were quantified, and how socio-economic factors were considered is also necessary for communication and for managers to be able to make informed decisions. A more formal description and implementation across rebuilding plans and Councils (see Chapter 6 for more details) would enhance the perceived credibility of the science and legitimacy of the decision-making process in the eyes of industry, NGOs, the public, U.S. Congress, and other stakeholders.

CONCLUDING REMARKS

The current implementation of the MSFMCA relies on a highly prescriptive approach that has resulted in demonstrated successes in identifying and rebuilding overfished stocks. Fishing mortality has generally been reduced for stocks that have been declared overfished and stock biomass has generally increased when fishing mortality was successfully reduced under rebuilding plans. Where they have been estimated, the long-term net economic benefits of rebuilding appear to be generally positive. Stocks that rebuilt or whose biomass increased appreciably were, in almost all cases in our analyses, experiencing fishing mortalities below F_{MSY}, and often less than 75% of F_{MSY}. More extreme reductions in target fishing mortalities have been implemented in situations in which rebuilding progress was slower than anticipated when the rebuilding plan was adopted, or the target year for rebuilding was approaching. The strong legal and prescriptive nature of rebuilding forces difficult decisions to be made, ensures a relatively high level of tractability, and can help prevent protracted debate over whether and how stocks should be rebuilt.

The present single-stock approach to rebuilding can, however, lead to inefficiencies. The perceived status of a stock can change with subsequent assessments. This can occur because of new data or assumptions in the more recent assessment that indicate that rebuilding is slower or faster than expected, or that there is a high probability that the stock was not actually overfished at the time it was declared overfished. Some stocks have not increased in biomass at the expected rate despite lowered fishing mortality rates, forcing more extreme reductions in fishing mortality to try to meet rebuilding demands. In some other cases, rebuilding plans have failed to reduce fishing mortality as much as intended, either due to overestimation of stock sizes or implementation issues, and rebuilding has been slow or not occurred. The inefficiencies that result from changes in stock assessments, lack of expected stock responses, or failure to achieve needed reductions in fishing mortality rates have, in cases, involved substantial negative biological and economic consequences (e.g., too low stock biomasses, lost future yields). Subsequent adjustments to rebuilding plans that are required in such cases can cause further substantial economic and social impacts (e.g., highly restricted fishing). In addition, the current approach is not as effective for the many data-poor stocks for which overfished status is unknown (over half of the nation's stocks). Even with well-studied stocks, the present approach forces reliance on forecasts and biomass-based reference points, which are sometimes highly uncertain. Some stocks may not conform to their rebuilding plans because of environmental variability, ecosystem interactions, or failure of the stock modeling to adequately account for nonlinear dynamics. Further, the stock-specific approach to rebuilding creates situations of foregone benefits when there are mixed stocks. In general, the present requirements of rebuilding have led to socio-economic considerations playing a secondary role in the design of rebuilding plans. Finally, there is a lack of standardization across geographic regions in how rebuilding plans are developed and implemented.

The committee used the evaluation, discussion, and findings in Chapters 2 through 6 as a basis for this final chapter, and looked forward with a long-term view at further improving the current approach to stock rebuilding. While Chapters 2 through 6 focused on rebuilding within the current legal framework, this chapter describes avenues for long-term planning in the development of stock rebuilding strategies over the next decades. We focused on seven aspects that directly or indirectly are related to the overarching issue of what is the appropriate degree of flexibility in stock rebuilding. We offer alternatives that could be more effective for developing rebuilding plans than the current approach, given the complex and variable nature of ecosystems, having to deal with coupled natural-human systems, and the considerable uncertainty in fishery science. Many of our comments could serve as suggestions for research and application to future revisions of National Standard Guidelines to improve the overall performance of stock rebuilding programs and thereby enhance the benefits derived from fisheries in the future.

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Appendix A

Committee and Staff Biographies

COMMITTEE

Ana M. Parma (*Co-Chair*) is a research scientist with CONICET – the Argentine Council for Science & Technology of Argentina. She earned her Ph.D. in Fisheries Science in 1989 from the University of Washington, and worked as an assessment scientist at the International Pacific Halibut Commission until 2000, when she returned to Argentina, her home country. Dr. Parma's research interests include fish stock assessment, population dynamics and adaptive management of fisheries resources. The main focus of her current work is on small-scale coastal shellfish fisheries, where she is involved in the evaluation of assessment and management approaches in several fisheries in South America. For this work she received support from a PEW Fellowship in Marine Conservation in 2003, and a Guggenheim Fellowship in 2008. She has participated as an independent scientist in many scientific and policy advisory groups and review panels. She is currently a member of the advisory panel of the Commission for the Conservation of Southern Bluefin Tuna, where she coordinated the development of a management procedure designed to rebuild the southern bluefin tuna stock. Dr. Parma has also served on four NRC committees, including the Committee on Ecosystem Effects of Fishing: Phase II -- Assessments of the Extent of Ecosystem Change and the Implications for Policy, the Committee on Evaluation, Design, and Monitoring of Marine Reserves and Protected Areas in the United States, the Committee on Fish Stock Assessment Methods, and Committee to Review Northeast Fishery Stock Assessments.

Patrick J. Sullivan (*Co-Chair*) is an Associate Professor of Quantitative Population and Community Dynamics at Cornell University's Department of Natural Resources. He earned his PhD in Biostatistics and his MS in Fisheries Science from the University of Washington. His research focuses on understanding what drives the spatial and temporal dynamics of natural populations and how these populations respond to anthropogenic influences. He also contributes to other research areas including the practical issues surrounding survey design and analysis, database management, and fisheries stock assessment as well as some more philosophical issues such as identifying what is the nature of good science, determining better ways for communicating and utilizing science and statistics, and clarifying scientific responsibility in issues of governance. Dr. Sullivan has served on several NRC committees, including chairing both the Committee on Review of Recreational Fisheries Survey Methods and the Committee on Improving the Collection and Use of Marine Fisheries Data. He also served on the Committee on Science and Its Role in the National Marine Fisheries Service.

Jeremy Collie is a professor of oceanography at the University of Rhode Island. He received his Ph.D. in biological oceanography from the joint program in oceanography with the Massachusetts Institute of Technology and Woods Hole Oceanographic Institution. He is a quantitative ecologist who specializes in fish population dynamics. Dr. Collie also studies the impacts of disturbance on benthic communities, predator-prey interactions, stock assessment and fisheries management. Furthermore, he has been a member of the Rhode Island Ocean Special Area Management Plan, Habitat Advisory Board; the U.S. National Oceanic and Atmospheric Administration, Science Advisory Board, Ecosystem Sciences and Management Working Group; and the New England Fisheries Management Council's Habitat Technical Team. Dr. Collie has served on two previous NRC committees; the Committee on Fish Stock Assessment Methods, and the Committee on Evaluating the Effects of Bottom Trawling on Seafloor Habitats.

Troy W. Hartley is a Research Associate Professor of Marine Science and Public Policy at the Virginia Institute of Marine Science (VIMS) and the Thomas Jefferson Public Policy Program, The College of William & Mary. He is a public administration/public policy scholar and his research considers the theory and practice of governance networks and collaboration, particularly in ecosystem-based management, fisheries management, and regional coastal and marine governance-primarily in the U.S. Northeast and Mid-Atlantic, and Canadian east coast. Dr. Hartley has conducted research on the design, implementation, stakeholder attitudes, and benefits of collaborative fisheries research, and is examining the role of collaborative research in social, ecological, and economic sustainability and resiliency. Dr. Hartley is the Director of Virginia Sea Grant, Past-President of the socioeconomic section of the American Fisheries Society (having served as its president for five years until 2011), and a member of the Commonwealth of Virginia's advisory Coastal Policy Team. He has advised the National Oceanic and Atmospheric Administration (NOAA) on measuring NOAA's societal impacts, setting the research agenda of ecosystem-based management, and understanding the land-sea governance network interactions for the Chesapeake Bay Program. Prior to joining VIMS in 2008, Dr. Hartley administered the Northeast Consortium, a collaborative fisheries research organization serving the Gulf of Maine and Georges Bank. He has a Ph.D. in environmental and natural resource policy from the University of Michigan, an M.A.I.S. in environmental dispute resolution from George Mason University, and a B.S. in zoology from the University of Vermont.

William Heyman is an associate professor in the Department of Geography at Texas A&M University, College Station. He is a marine ecologist by training and received his Ph.D. in marine science from the University of South Carolina. Dr. Heyman’s research focuses on the conservation and sustainable management of tropical coastal and marine resources. A core area of his research centers on the reproductive ecology of reef fish spawning aggregations and strategies for their conservation and management. He has worked in the Bahamas, Turks and Caicos Islands, the Marshall Islands, Honduras, Guatemala, Mexico, the Cayman Islands, Venezuela, and most extensively in Belize over the past 20 years. Given his position in TAMU’s Geography Department, Dr. Heyman also studies the relationship between humans, their resource use patterns, and their environment and utilizes multi-disciplinary approaches to study these interactions. In addition to the science, he is dedicated to generating practical and useful guidance for natural resource managers and local conservationists. In order to generate relevant information for managers, Dr. Heyman collaborates with colleagues from a wide set of disciplines including hydrology, oceanography, fisheries and marine ecology, physical geography, anthropology, economics, and utilizes tools of remote sensing and GIS.

Robert Johnston is Director of the George Perkins Marsh Institute and Professor of Economics at Clark University. He has a PhD in the economics of marine resources from the University of Rhode Island and a BA in economics from Williams College. Dr. Johnston's research addresses such topics as the valuation of non-market commodities and aquatic ecosystem services; benefit transfer and meta-analysis; and the management of aquatic resources, fisheries, and tourism. Over the past two decades he has authored hundreds of articles, chapters, books and other scientific and policy papers. He has worked with numerous international organizations, government agencies and non-profit organizations to assist in the appropriate use of economic information to guide natural resource policy development. His work has contributed to national, state and local policy in the US, Canada and elsewhere. Among other appointments on advisory, scientific and review committees, Dr. Johnston currently serves on advisory boards for the Marine Resource Economics Foundation, the Charles Darwin Foundation, the Communication Partnership for Science and the Sea, the Gulf of Maine Regional Ocean Science Council, Connecticut Sea Grant, and New York Sea Grant.

André E. Punt is a professor and current Director for the School of Aquatic & Fishery Sciences at the University of Washington. Dr. Punt is a mathematician with a B.Sc., M.Sc. and Ph.D. degrees in applied mathematics from the University of Cape Town, South Africa. He and his lab develop approaches to providing quantitative scientific advice for fisheries management. His research is primarily focuses on new methods for assessing fish and marine mammal populations and includes Bayesian assessment and risk analysis methods. Dr. Punt also is involved in evaluating the performance of existing methods for assessing and managing renewable resource populations. He has published nearly 200 peer-reviewed articles on a spectrum of fisheries related subjects including population modeling, fisheries management, stock assessment methodologies, assessment models, and quantitative ecology of marine resources.

Kenneth A. Rose is the E. L. Abraham Distinguished Professor in Louisiana Environmental Sciences in the Department of Oceanography and Coastal Sciences at Louisiana State University. He earned his Ph.D. in Fisheries from the University of Washington. Dr. Rose's research interests include developing and applying mathematical and simulation models to better understand and forecast the effects of natural and anthropogenic factors on aquatic populations. Other interests include the use of models in resource management, fisheries stock assessment and risk assessment. He has published extensively on the challenges of modeling fish population dynamics and their relationship to resources, stressors, site-specific factors and life history characteristics. He has served in a number of capacities with the Gulf of Mexico Fisheries Management Council since the late 1990s. Dr. Rose has also served on a recent NRC study Committee on Sustainable Water and Environmental Management in the California Bay-Delta.

James Sanchirico is a professor in the Department of Environmental Science and Policy at the University of California, Davis. Dr. Sanchirico is a natural resource economist by training, having earned his PhD in Agricultural and Resource economics from UCD. His research applies quantitative methods to study the design and evaluation of policy instruments for the conservation of natural resources. Specifically, he has worked on the management of marine populations and habitats, land-use, biodiversity conservation, invasive species management, provision of ecosystem services, and the design of market based policies, such as individual fishing quota systems. Dr. Sanchirico employs a variety of tools that include optimal control theory, differential equations, constrained optimization, household surveys, spatial statistics, and time series and cross-sectional econometric techniques. Some of his most recent work involves the design and analysis of catch share programs. Dr. Sanchirico has served as a reviewer for several NRC studies and also served on the NRC Committee to Review JSOST U.S. Ocean Research Priorities Plan.

Michael P. Sissenwine is the former Director of Research and Chief Science Advisory of the National Marine Fisheries Service (2002-2005). He was responsible for about 30 Laboratories and 1,400 staff. NMFS provides the scientific basis for conservation and management of marine living resources and ecosystems. During his 30 year career with the Agency, he also served as a research scientist, Director of the Northeast Fisheries Science Center (1996-2002), and the Agency's Senior Scientist (1990-1996). He was the President of the International Council for Exploration of the Sea (2004-2006) and chair of the committee which advises European countries on ocean issues (2008-2010). He is currently a Visiting Scholar at the Woods Hole Oceanographic Institution and a marine science consultant. He earned a Ph.D. in oceanography form the University of Rhode Island. Dr. Sissenwine has authored over 100 scientific papers on a wide range of topics. He serves on the Scientific and Statistical Committee of two Fisheries Management Councils, and he has served on, or led, numerous delegations to international scientific and management organizations. Dr. Sissenwine is the recipient of several prestigious awards including a Presidential Meritorious Rank Award and ICES and American Fisheries Society lifetime achievement awards. He has served on the OSB and BISO Boards, on four NRC or NAS committees (Coastal Ocean Science, Ecosystem Management for Sustainable Marine Fisheries, International Capacity Building for the Protection and Sustainable Use of Oceans and Coasts, National Committee for the Pacific Sciences Association as chair), and he has lead Delegations on behalf of the NAS.

George Sugihara is a professor and department chair at SIO at the University of California, San Diego. He earned his Ph.D. in Mathematical Biology from Princeton. His diverse research interests include complexity theory, nonlinear dynamics, food web structure, species abundance patterns, conservation biology, biological control, empirical climate modeling, fisheries forecasting, and the design and implementation of derivative markets for fisheries. One of his most interdisciplinary contributions involves the work he developed with Robert May concerning methods for forecasting nonlinear and chaotic systems. This took him into the arena of investment banking, where he took a five-year leave from SIO to become Managing Director for Deutsche Bank. There he made a successful application of these theoretical methods to forecast erratic market behavior. Most of Dr. Sugihara's early work was motivated exclusively by pure science and the later work more by pragmatic utility and environmental concerns. Nearly all of it is based on extracting information from observational data (turning data into information). His initial work on fisheries as complex, chaotic systems led to work on financial networks and prediction of chaotic systems. Dr. Sugihara serves on the Board on Mathematical Sciences and their Applications here at the NRC and also served on the Planning Committee for a Workshop on Technical Capabilities Required for Regulation of Systemic Risk.

STAFF

Kim Waddell is a senior program officer with the Ocean Studies Board. He received his Ph.D. in the Biological Sciences from the University of South Carolina and his B.A. in Environmental Studies from the University of California, Santa Cruz. Dr. Waddell recently rejoined the NRC after a 6-year hiatus during which he was a research associate professor at the University of the Virgin Islands and Texas A&M University working to build marine and environmental research capacity in the Caribbean region. During his previous tenure with the NRC, Dr. Waddell directed a number of studies for the Board on Agriculture and Natural Resources including California Agricultural Research Priorities: Pierce's Disease (2004), Biological Confinement of Genetically Engineered Organisms (2004), Animal Biotechnology; Science-based Concerns (2002), The Environmental Effects of Transgenic Plants (2002), Exploring Horizons for Domestic Animal Genomics (2002), and The Future Role of Pesticides in US Agriculture (2000).

Heather Chiarello joined the U.S. National Academy of Sciences in July 2008. She graduated magna cum laude from Central Michigan University in 2007 with a B.S. in political science with a concentration in public administration. Ms. Chiarello is currently a senior program assistant with the Ocean Studies Board in the Division on Earth and Life Sciences of the National Academies. She is pursuing a Master's degree in sociology and public policy analysis at The Catholic University of America in Washington, D.C.

Constance (Stacee) Karras joined the U.S. National Academy of Science in September 2012. She received her B.A. in Marine Affairs and Policy with concentrations in Biology and Political Science from the University of Miami in 2007. The following year she received an M.A. in Marine Affairs and Policy from the University of Miami's Rosenstiel School of Marine and Atmospheric Science. Most recently, she earned her J.D. from the University of Virginia School of Law. Ms. Karras is now serving as a Post-Graduate Intern with the Ocean Studies Board in the Division of Earth and Life Sciences of the National Academies.

Appendix B

List of Acronyms

ABC	Acceptable Biological Catch, a catch that is less than the OFL to account for scientific uncertainty
ACL	Annual Catch Limit, typically specified in units of tons
ACT	Annual Catch Target
AM	Accountability Measures
AOA	Analysis of Alternatives
ASMFC	Atlantic States Marine Fisheries Commission
B	Generic reference to biomass (usually measured in terms of spawning stock biomass but other units such as egg production may be preferred)
B_0	The unfished equilibrium biomass
$B_{x\%}$	The average biomass corresponding to fishing at a rate of $F_{x\%}$.
B_t	Biomass in year t (usually measured in terms of spawning stock biomass but other units such as egg production may be preferred)
B_{MSY}	Biomass corresponding to Maximum Sustainable Yield (often determined using a proxy such as 40% of B_0)
CFP	Common Fisheries Policy
CMFC	Caribbean Fishery Management Council
CPUE	Catch Per Unit Effort
CRA	"Crayfish," New Zealand management area designation for spiny red rock lobster
DFO	Fisheries and Oceans Canada
EBFM	Ecosystem-Based Fisheries Management
EC	European Commission
EEZ	Exclusive Economic Zone
EFH	Essential Fish Habitat
EIS	Environmental Impact Statement
EU	European Union
F	Generic reference to fishing mortality
$F_{x\%}$	The fishing mortality rate at which spawner biomass-per-recruit is reduced to x% of its unfished level.
F_{ACL}	Target fishing mortality
F_{MSY}	Fishing mortality rate corresponding to Maximum Sustainable Yield
F_{REBUILD}	Fishing mortality that achieves a 50% probability of rebuilding by $\mathrm{T}_{\mathrm{TARGET}}$
F_{THREAT}	Fishing mortality that drives a component population to threatened or endangered status

FAO	Food and Agriculture Organization of the United Nations
FCMA	Fishery Conservation and Management Act (See also MFCMA, MSFCMA, MSA and SFA)
FIS	Fishery Impact Statements
FMP	Fishery Management Plan
FSRP	Fish Stock Rebuilding Plan
FSSI	Fish Stock Sustainability Index, a list of stocks and their status relative to being overfished and subject to overfishing.
GMFMC	Gulf of Mexico Fishery Management Council
HMS	Highly Migratory Species
IATTC	Inter-American Tropical Tuna Commission
ICES	International Council for the Exploration of the Sea (needs to be spelled out on page 3-46ish)
ICNAF	International Commission for the North Atlantic Fisheries
IPHC	International Pacific Halibut Commission
M	Coefficient of natural mortality
MAFMC	Mid-Atlantic Fishery Management Council
MSST	Minimum Stock Size Threshold, the level of biomass at which a stock is declared to be overfished, often set at 50% of the B_{MSY} (or its proxy).
MCY	Maximum Constant Yield
MEY	Maximum Economic Yield
MFCMA	Magnuson Fishery Conservation and Management Act (as amended in 1980)
MFMT	Maximum Fishing Mortality Threshold (cannot exceed F_{MSY})
MSA	Magnuson Stevens Act (See also FCMA, MFCMA, MSFCMA, and SFA)
MSE	Management Strategy Evaluations
MSY	Maximum Sustainable Yield
MSFCMA	Magnuson-Stevens Fishery Conservation and Management Act (as amended in 1996; see also FCMA, MFCMA, and SFA)
MSST	Minimum Stock Size Threshold
MWPT	Mean Weight Per Tow
MT	Metric Tons
NAS	National Academy of Science
NEFMC	New England Fishery Management Council
NEPA	National Environmental Policy Act
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NPFMC	North Pacific Fishery Management Council
NRC	National Research Council
NS1G	National Standard 1 Guidelines
OECD	Organisation for Economic Co-operation and Development
OFL	Overfishing Limit, the biomass of catch corresponding to a catch under a F_{MSY} harvest strategy.
OY	Optimum Yield
PFMC	Pacific Fishery Management Council
RFMC	Regional Fishery Management Council
RIR	Regulatory Impact Review

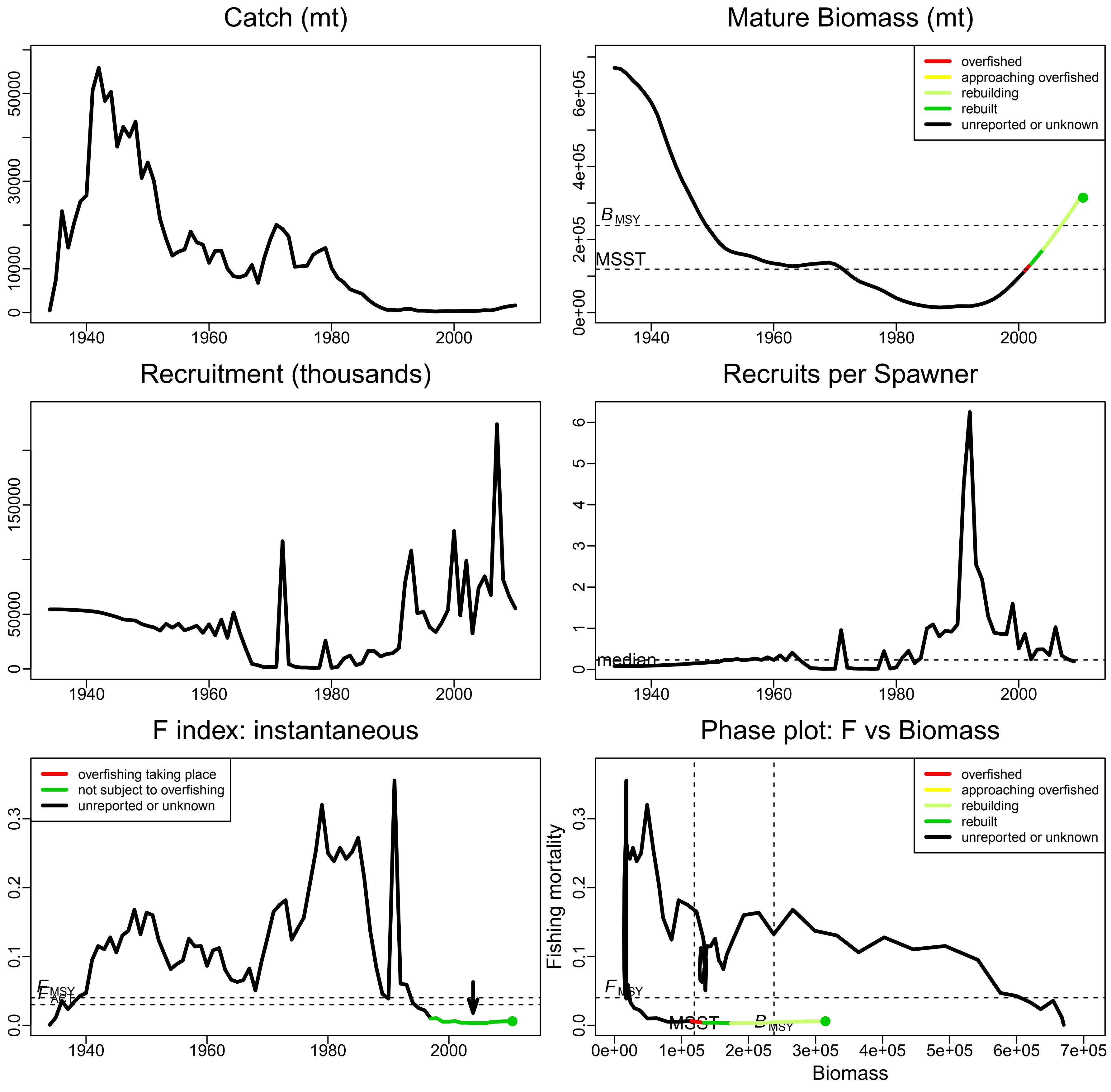
SAFE	Stock Assessment and Fishery Evaluation
SAFMC	South Atlantic Fishery Management Council
SFA	Sustainable Fisheries Act (1996), an amendment to the MFCMA
SSB	Spawning Stock Biomass
SSC	Scientific and Statistical Committee
SPR	Spawning Potential Ratio
SPR_{TARGET}	Target spawning potential ratio (equivalent to the target fishing mortality rate if the mix of fishing gears does not change over time)
Steepness	Fraction of unfished recruitment expected when the stock is depleted to 20% of B_0.
T	Generic reference to time
T_{MIN}	Minimum time to rebuild to B_{MSY} (with 50% probability)
T_{MAX}	Maximum permissible time to rebuild to B_{MSY} (10 years unless the biology of the species does not allow the stock to rebuild in ten years)
T_{TARGET}	Target time for rebuilding - must lie between T_{MIN} and T_{MAX}
TAC	Total Allowable Catch
UNFA	United Nations *Agreement on Straddling and Highly Migratory Fish Stocks*
USR	Upper Stock Reference
WPFMC	Western Pacific Fishery Management Council
WSSD	World Summit on Sustainable Development (Needs to be spelled out like this and abbreviated when mentioned the first time, on 3-46ish)
YD	Year a stock was declared to be overfished

Appendix C

Time Series Plots

Time series of total catch, stock size, fishing mortality, recruitment and recruits per spawning biomass for all stocks for which estimates were made available for this report. Results correspond to the most recent assessments as of September 2012. Reference points for stock size or biomass (B_{MSY} or proxy), minimum stock size threshold (MSST) and fishing mortality (F_{MSY} or proxy) are indicated by dashed lines. When available, the target fishing mortality ($F_{\mathrm{ACT})}$ used to calculate the catch limit for 2012 is also shown. Definitions of stock size and fishing mortality differ for the different stocks; for some stocks fishing mortality corresponds to 1-SPR (one minus the reduction in spawning biomass per recruit). The vertical arrow in the fishing mortality plot indicates the first year of the Rebuilding Plan. Lines are colored according to overfishing status since 1997 (for B or F), as classified in the annual reports to Congress. Mismatches between the line color and the values of B or F relative to reference points (e.g., red instead of green when biomass is larger than B_{MSY}) are due to differences between the initial assessment (used for overfishing status classification, shown by the color code) and the most recent assessment. The point at the end of each time series is colored according to overfishing status in the report to Congress of September 2012. SOURCE: estimates provided by NMFS, complemented in some cases by information provided by the assessment authors, and obtained from assessment reports.

C.1: Acadian redfish − Gulf of Maine / Georges Bank



Catch (mt)

Mature Biomass (mt)

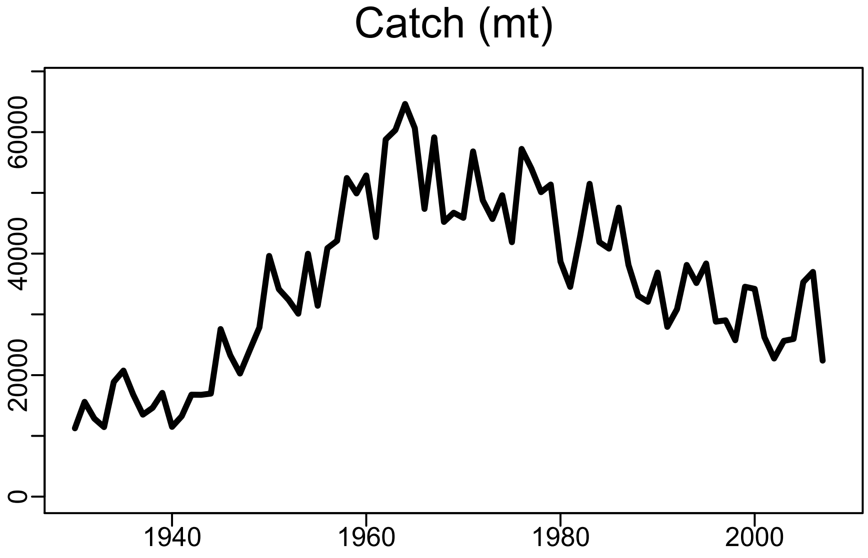
Recruitment (thousands)

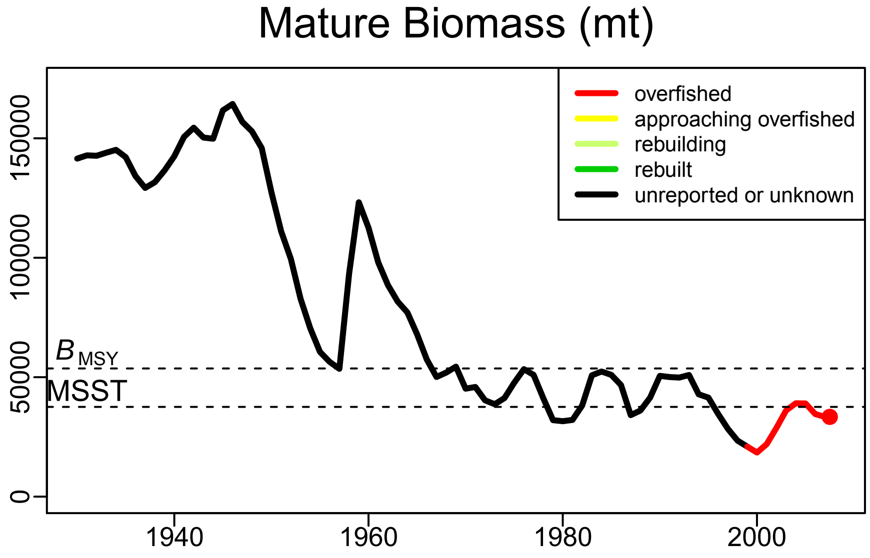
Recruits per Spawner

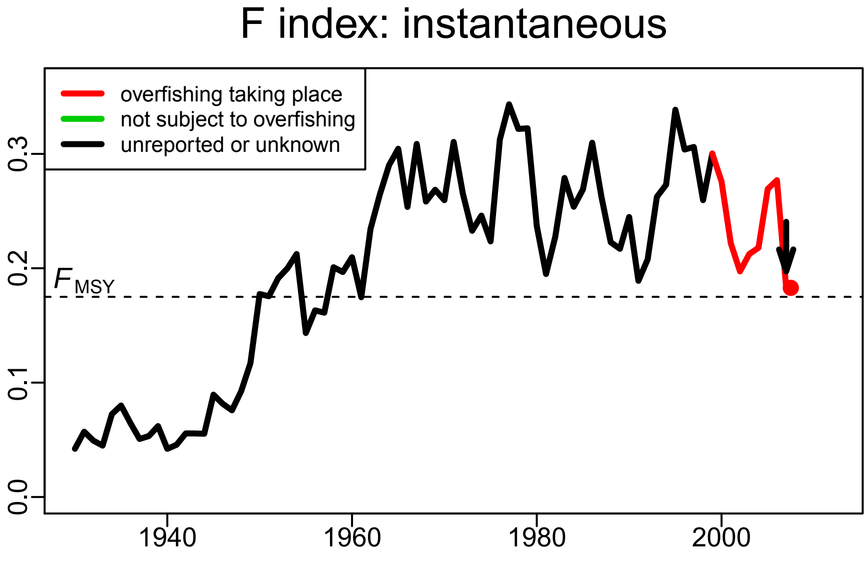
F index: instantaneous

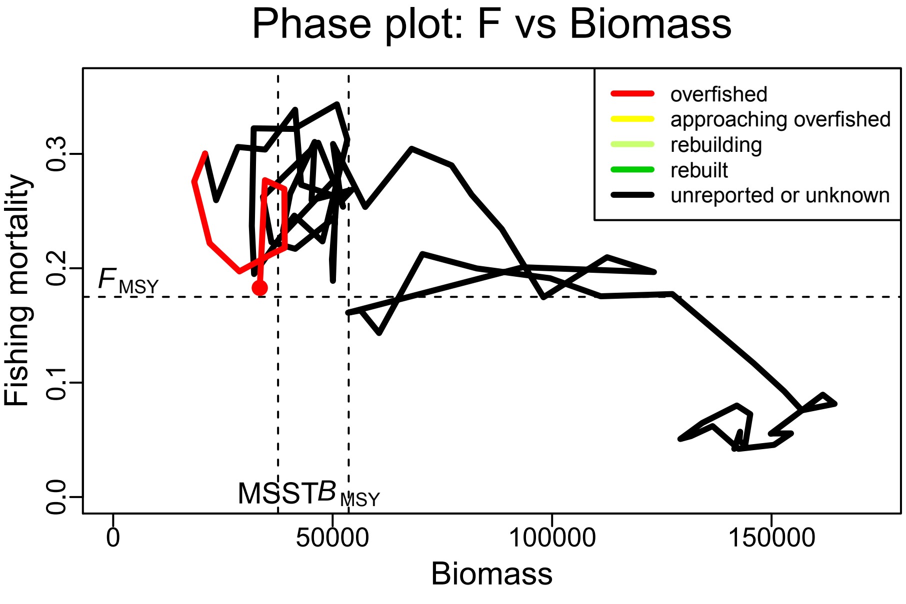
Phase plot: F vs Biomass

C.2: Albacore − North Atlantic

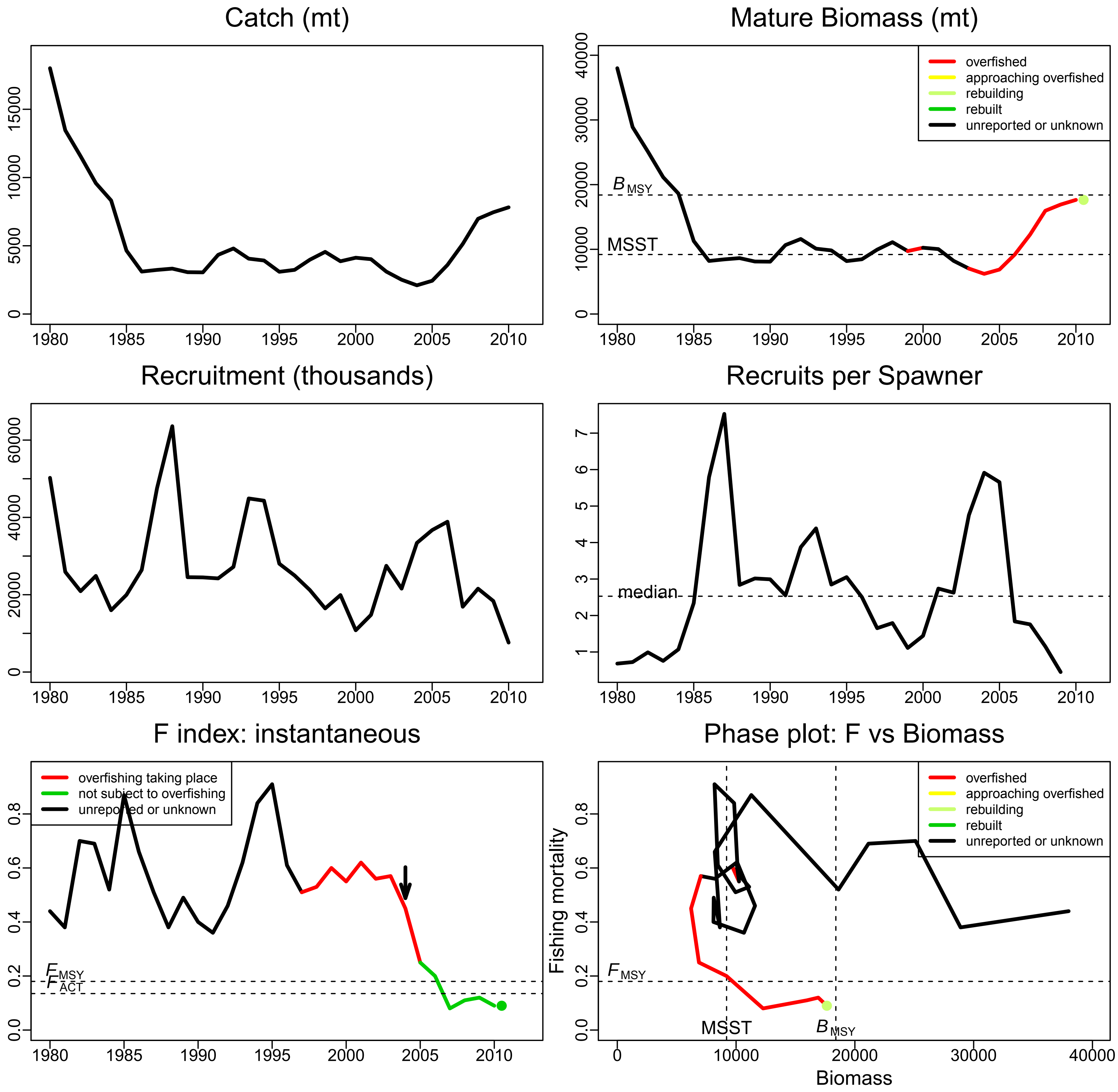

Catch (mt)


Mature Biomass (mt)

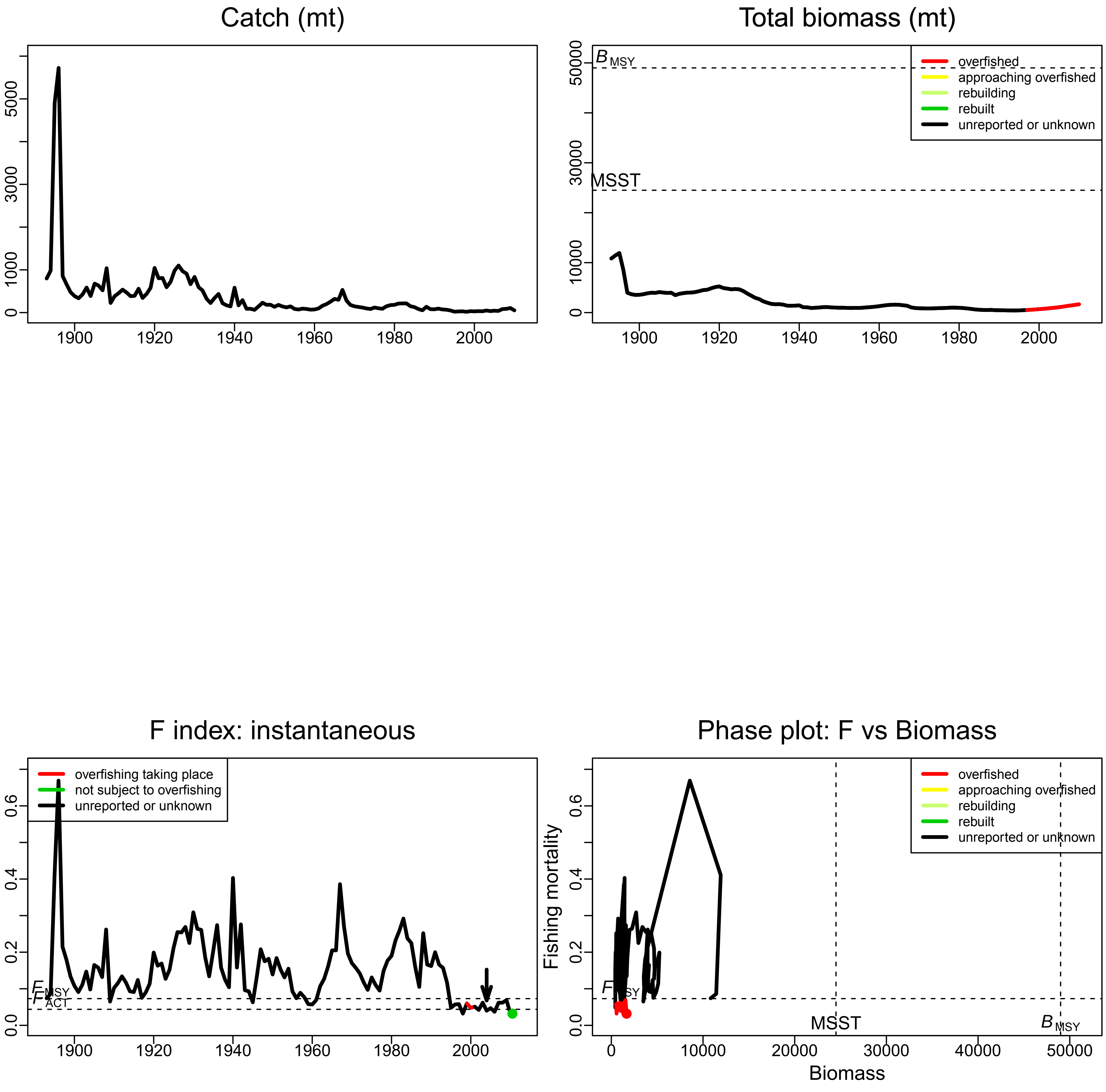

F index: instantaneous


Phase plot: F vs Biomass

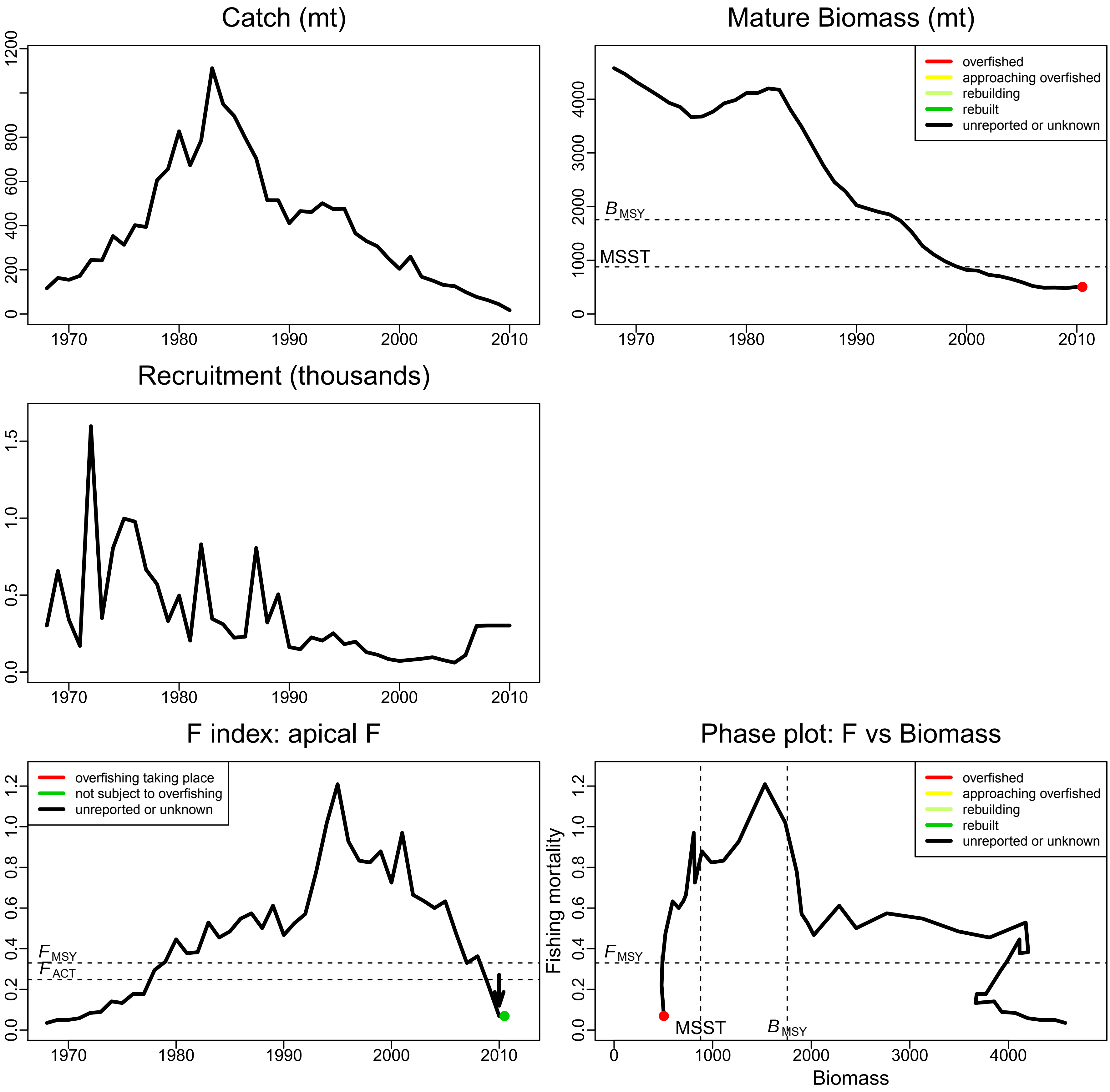
C.3: American plaice − Gulf of Maine / Georges Bank



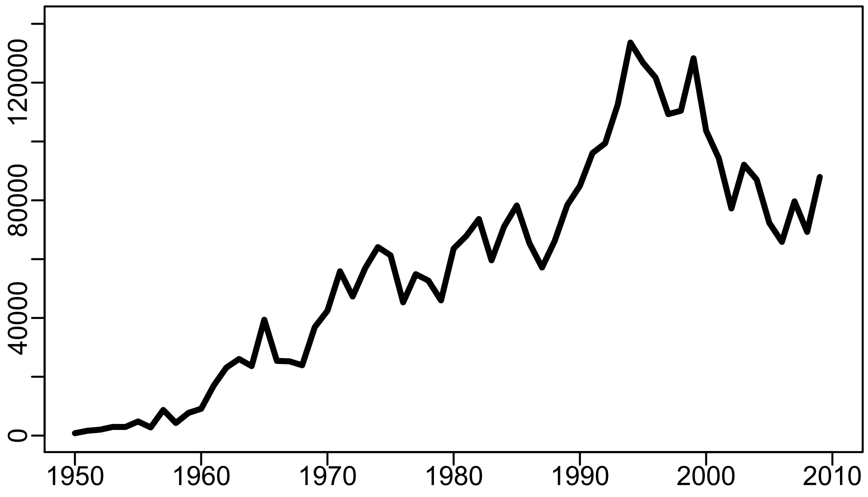
C.4: Atlantic halibut − Northwestern Atlantic Coast



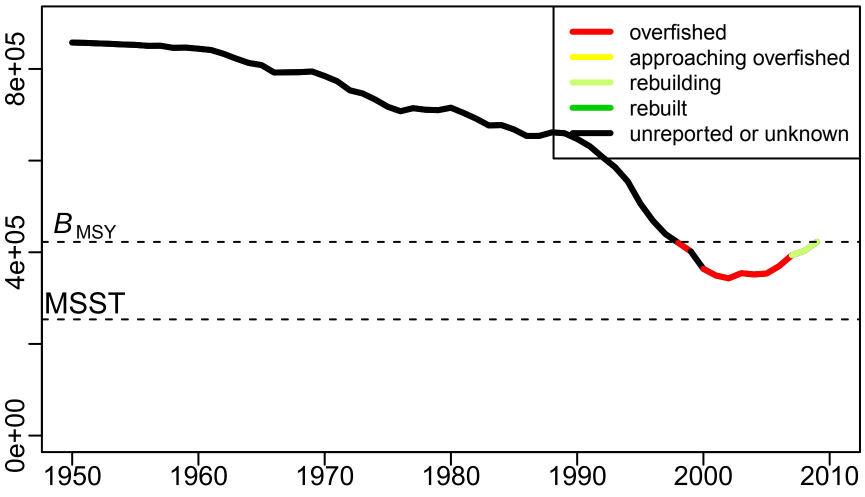
C.5: Atlantic wolffish − Gulf of Maine / Georges Bank



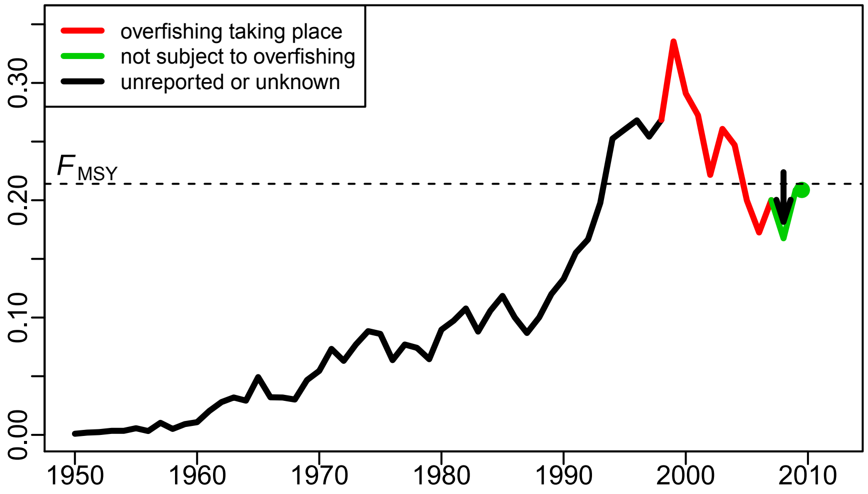
C.6: Bigeye tuna − Atlantic



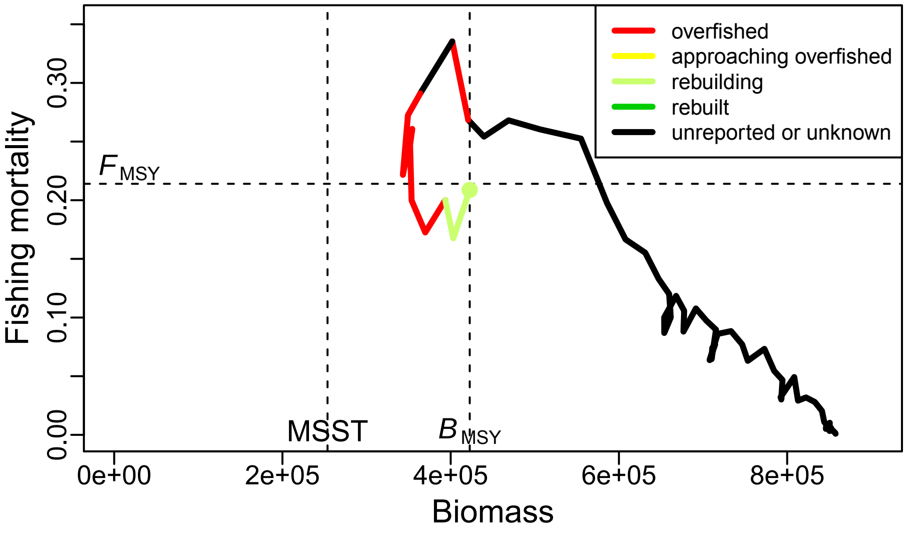
Catch (mt)



Total biomass (mt)

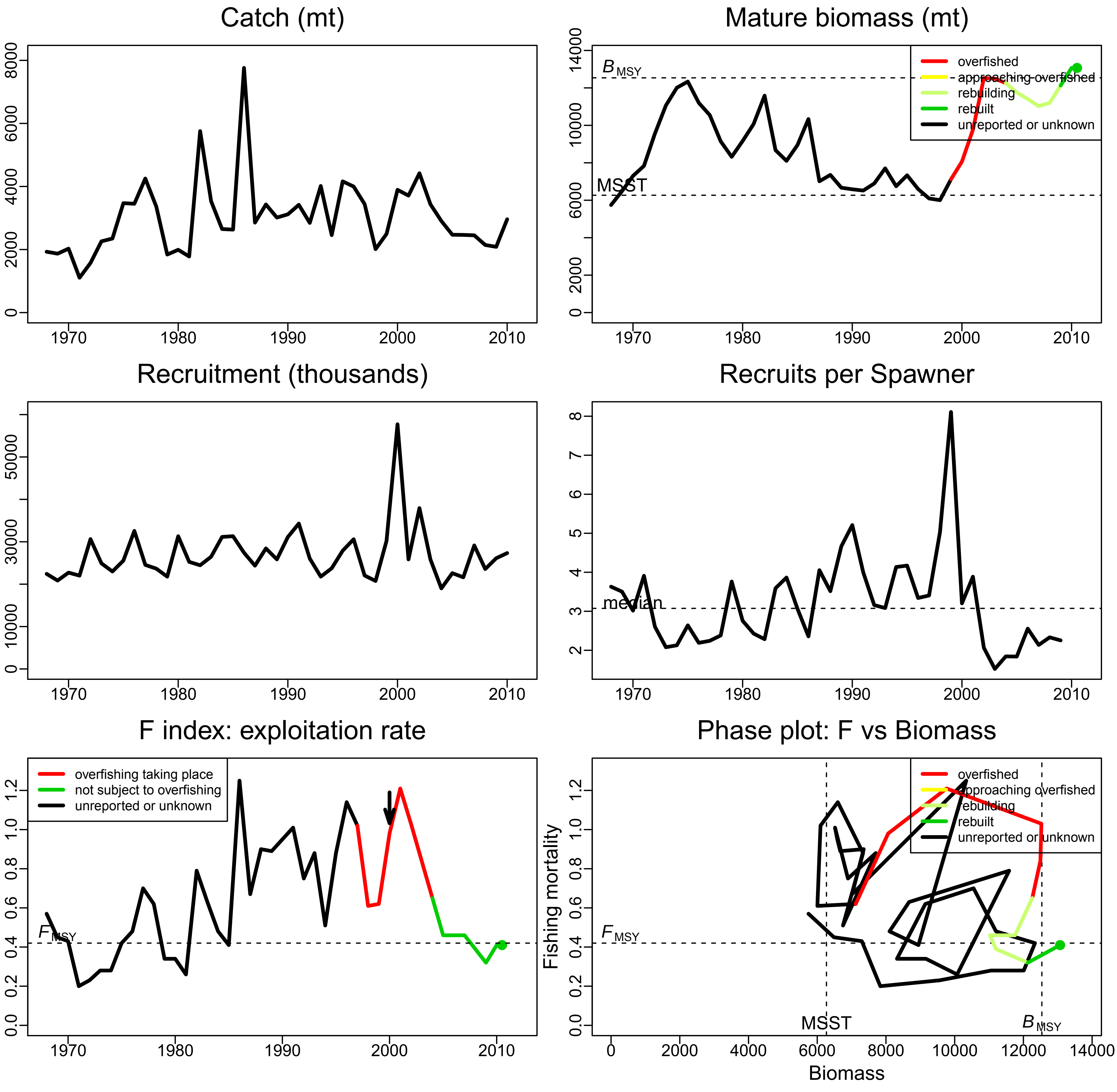


F index: exploitation rate

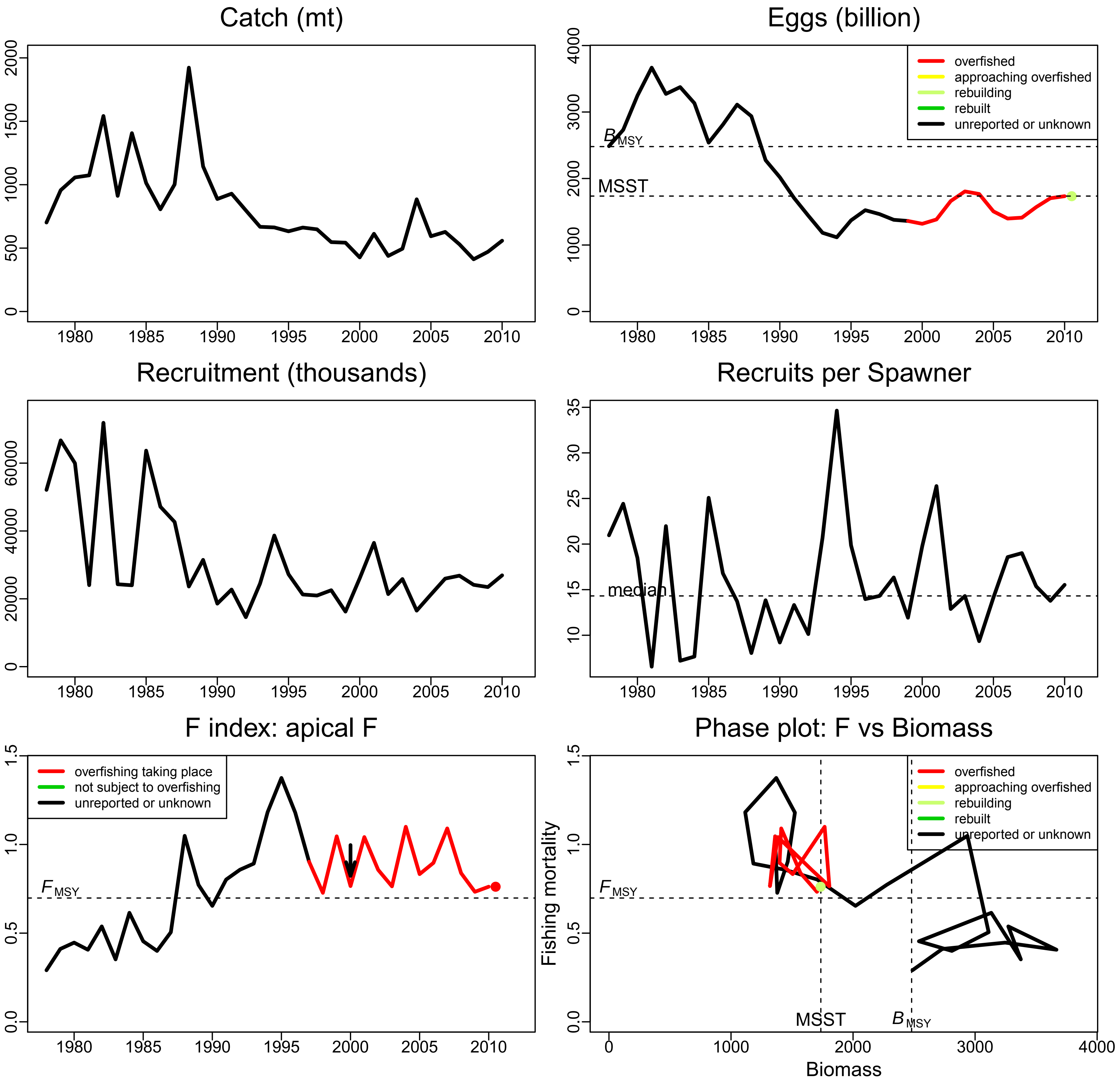


Phase plot: F vs Biomass

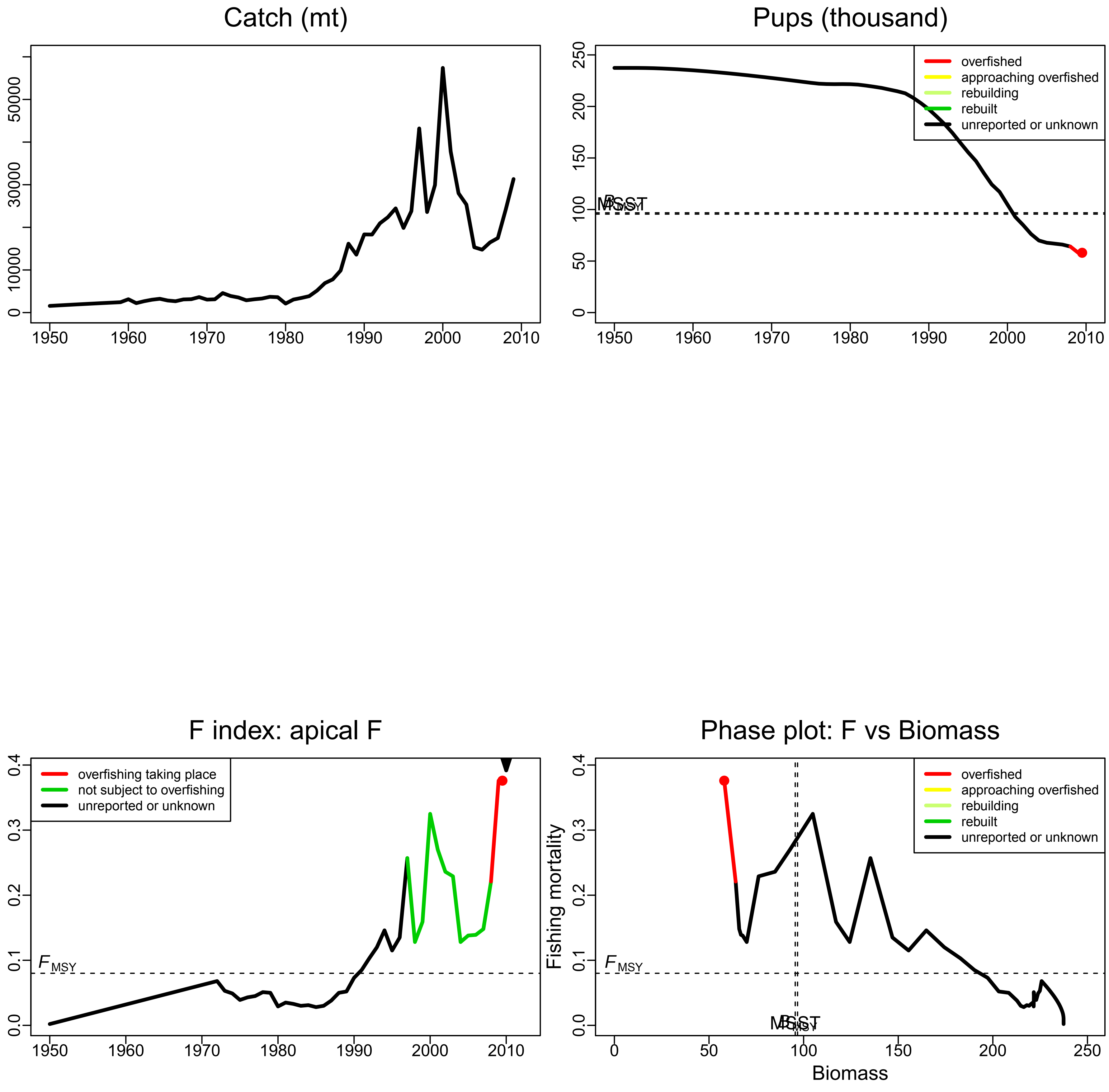
C.7: Black sea bass − Mid−Atlantic Coast



C.8: Black sea bass − Southern Atlantic Coast



C.9: Blacknose shark − Atlantic



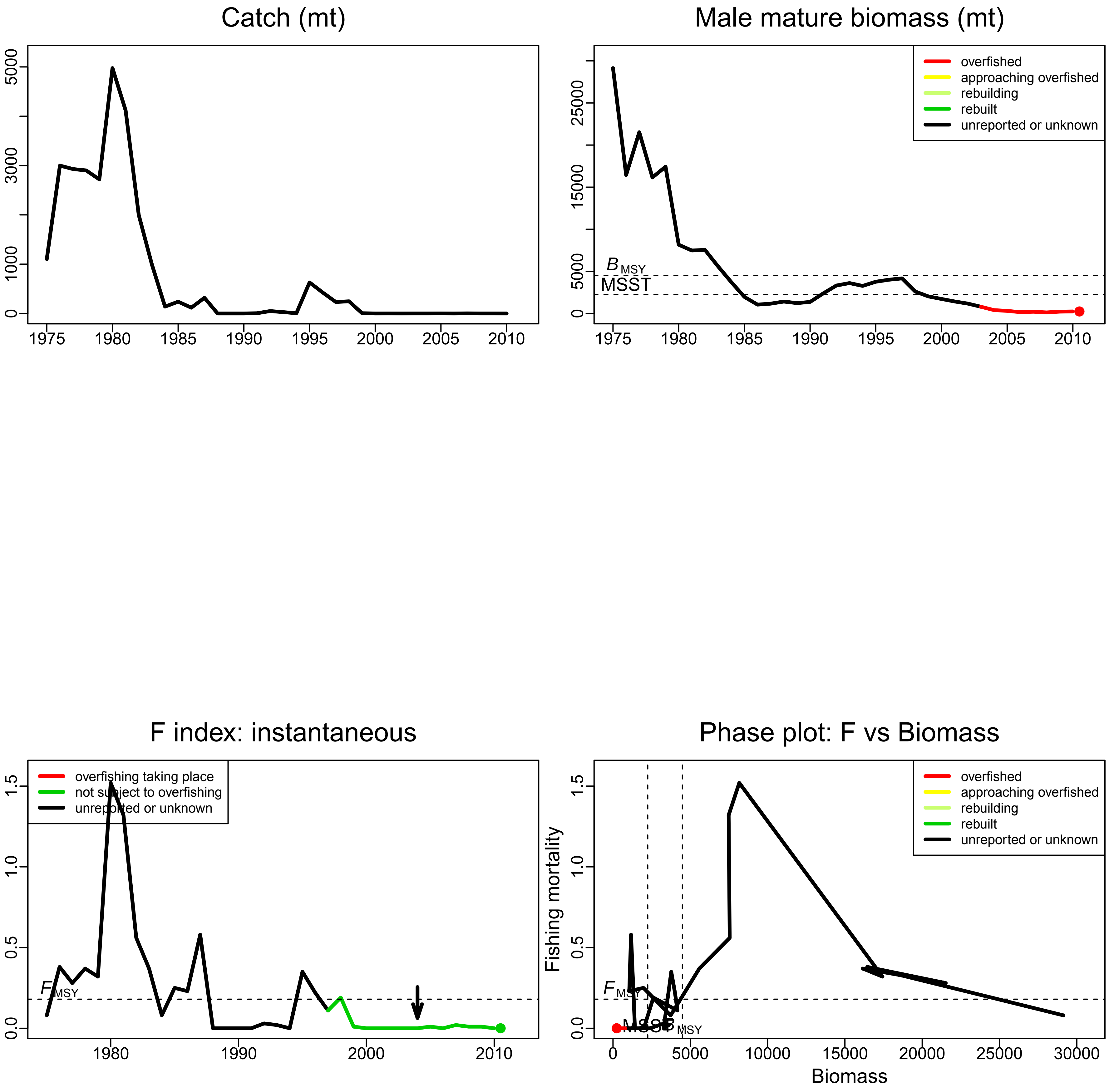
Catch (mt)

Pups (thousand)

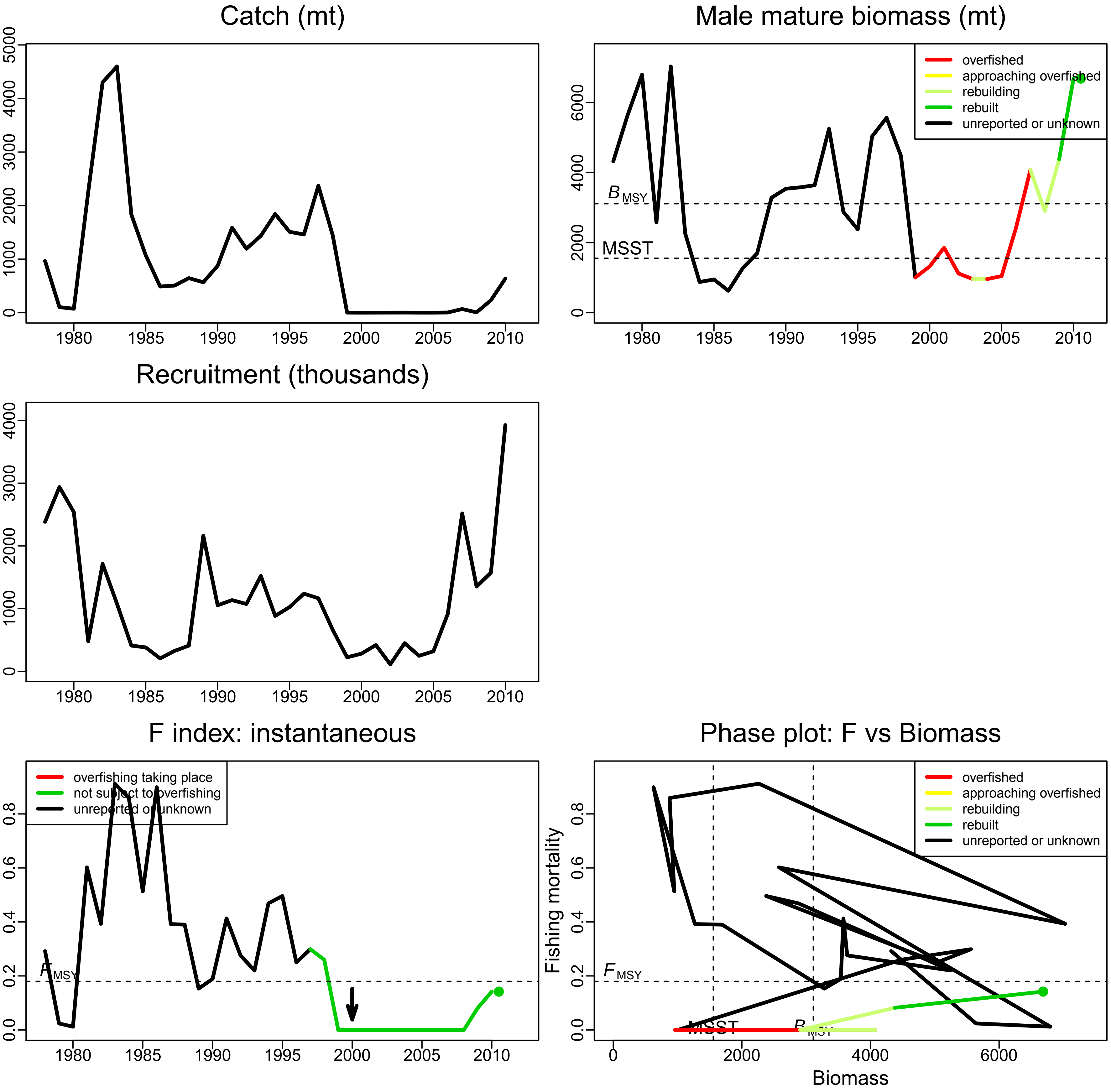
F index: apical F

Phase plot: F vs Biomass

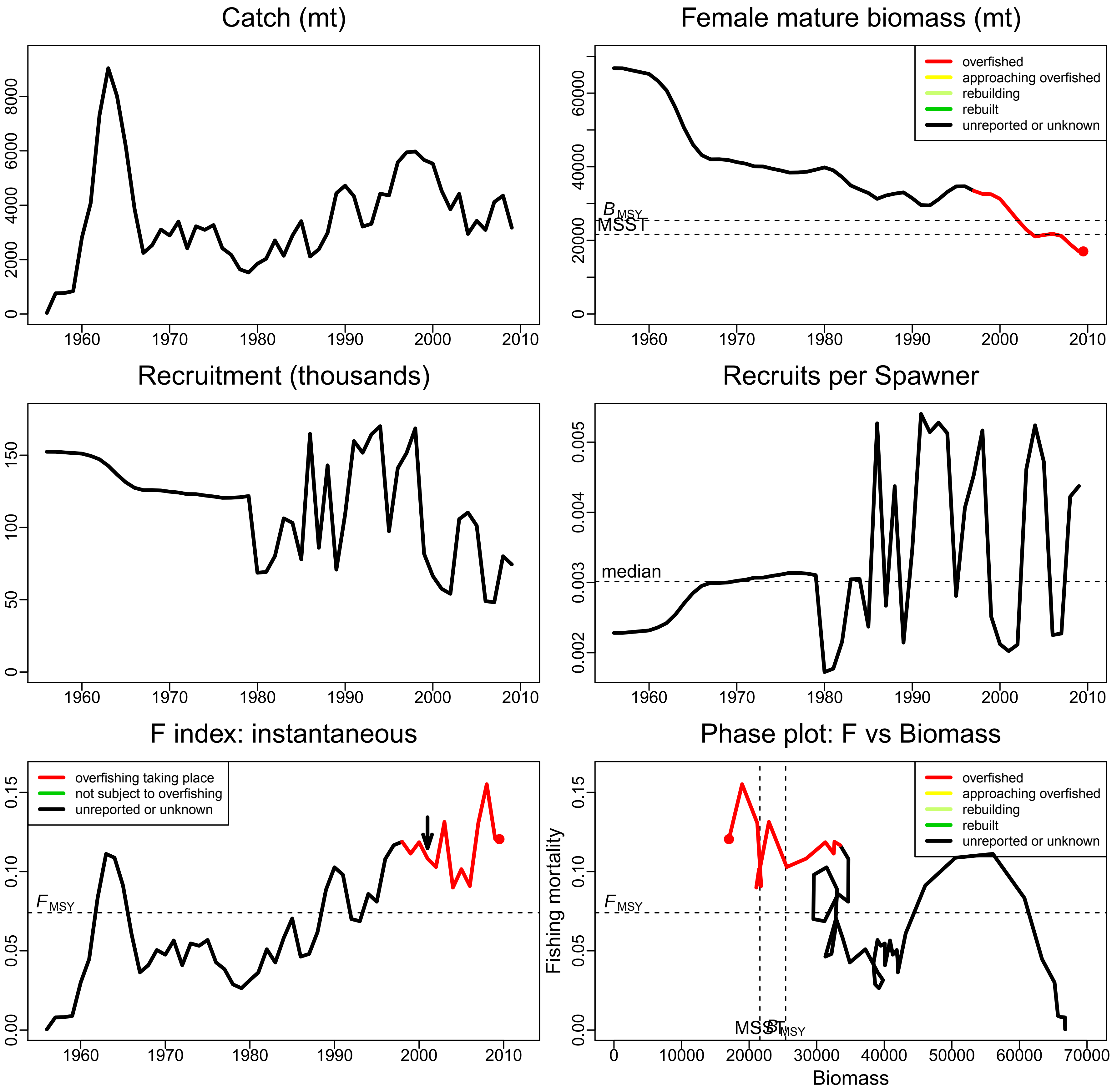
C.10: Blue king crab − Pribilof Islands



C.11: Blue king crab − Saint Matthew Island



C.12: Blue marlin − Atlantic


Catch (mt)

Female mature biomass (mt)

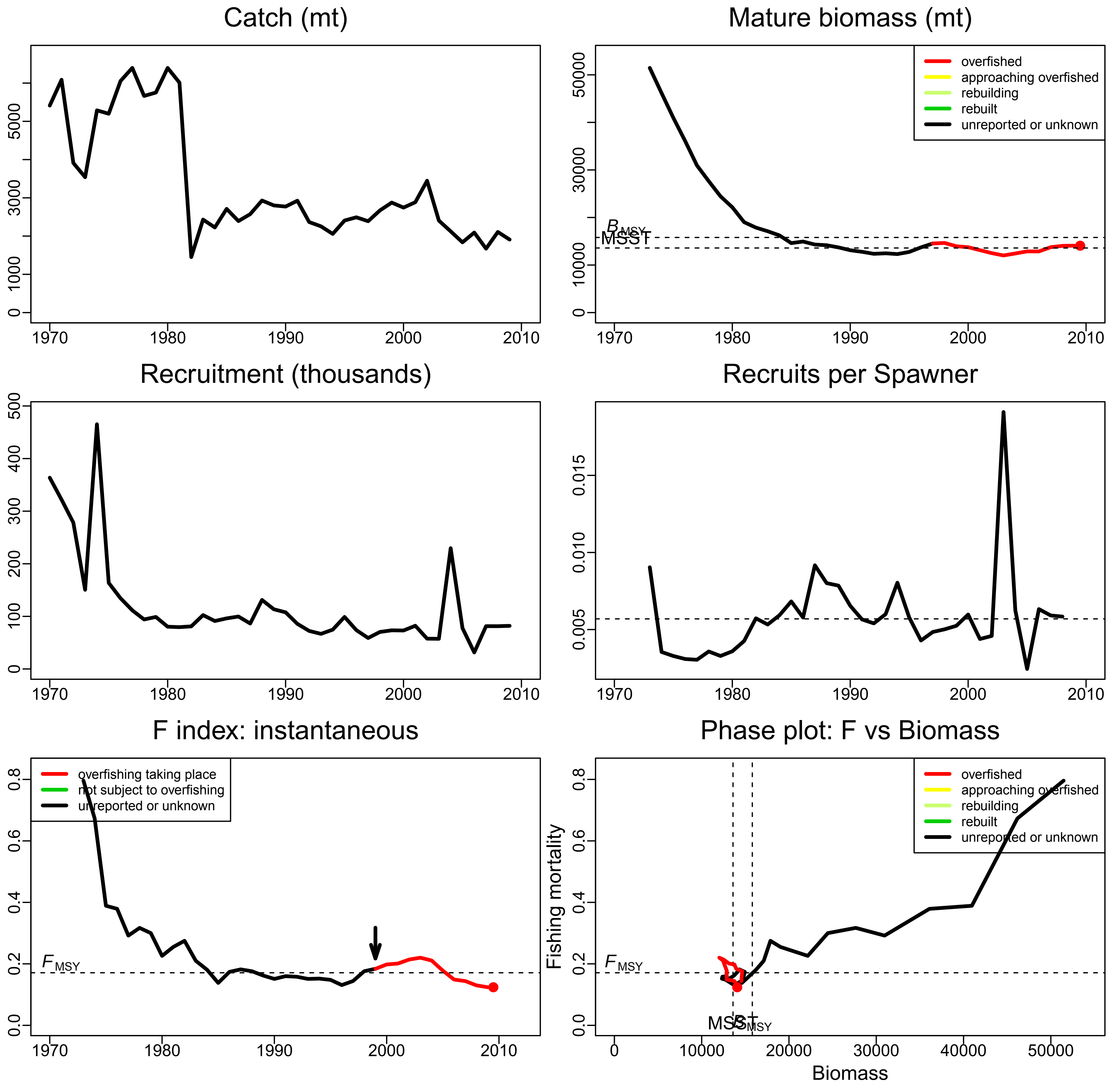
Recruitment (thousands)

Recruits per Spawner

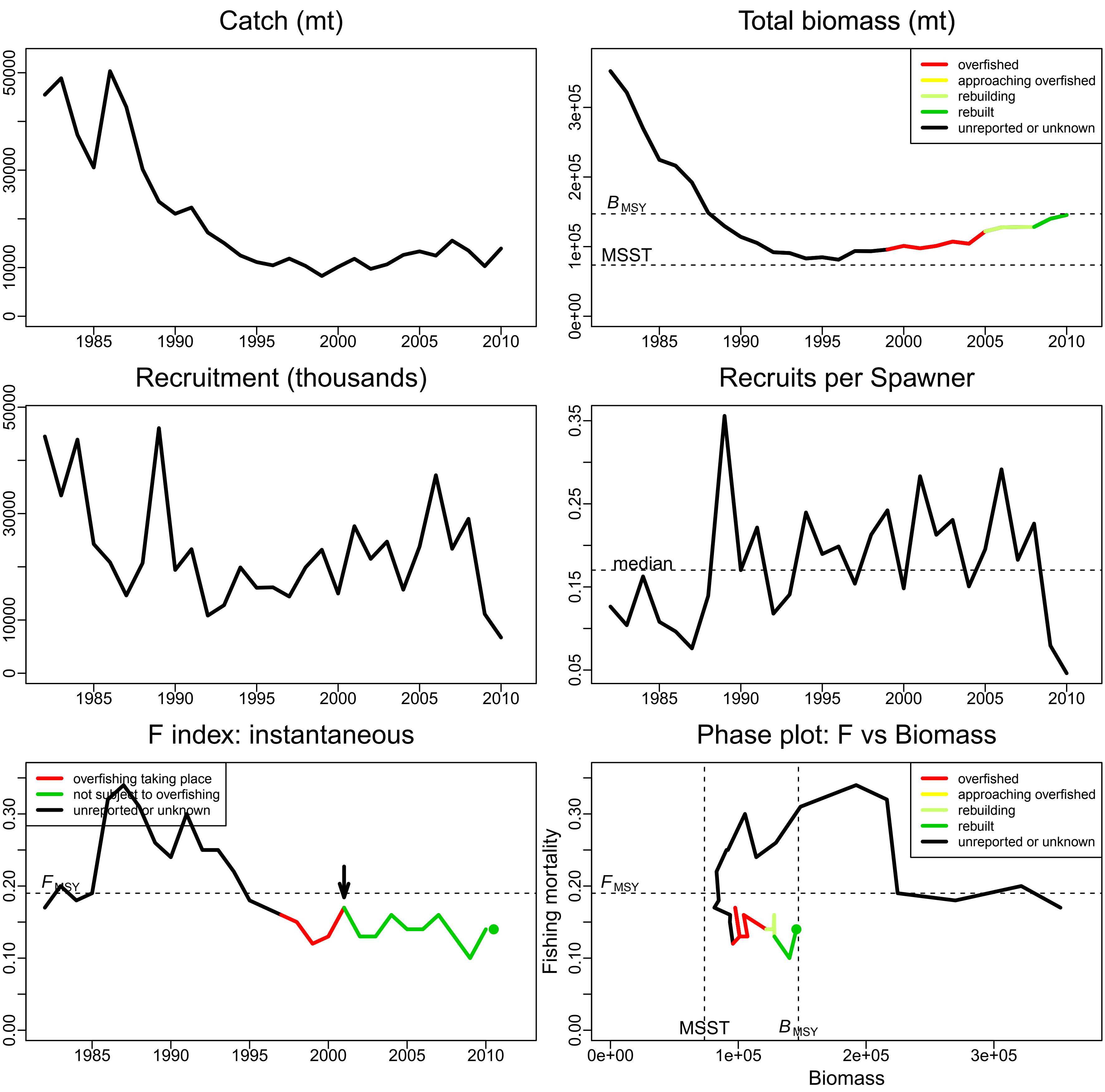
F index: instantaneous

Phase plot: F vs Biomass

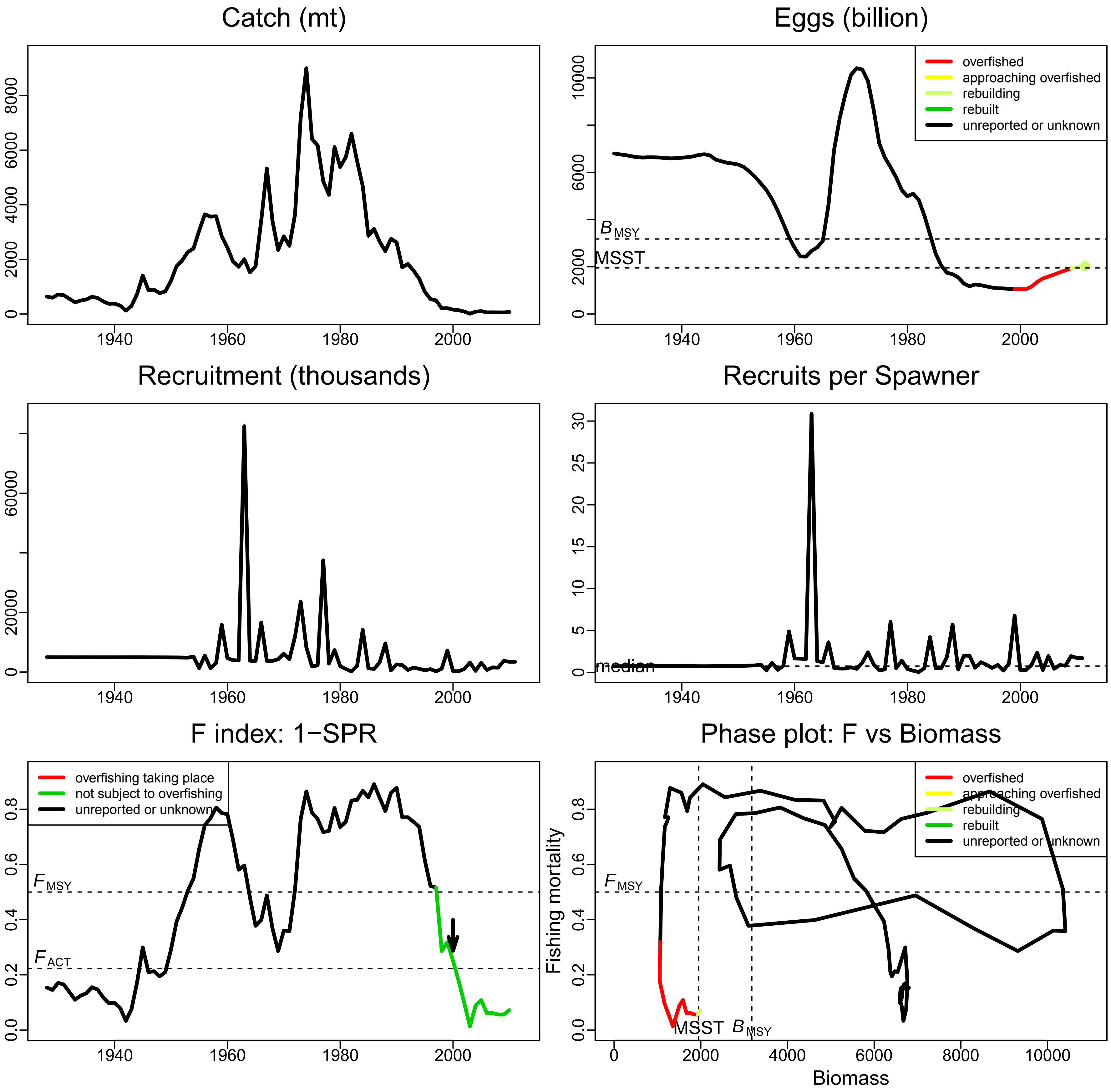
C.13: Bluefin tuna − Western Atlantic



C.14: Bluefish − Atlantic Coast



C.15: Bocaccio − Southern Pacific Coast



Catch (mt)

Eggs (billion)

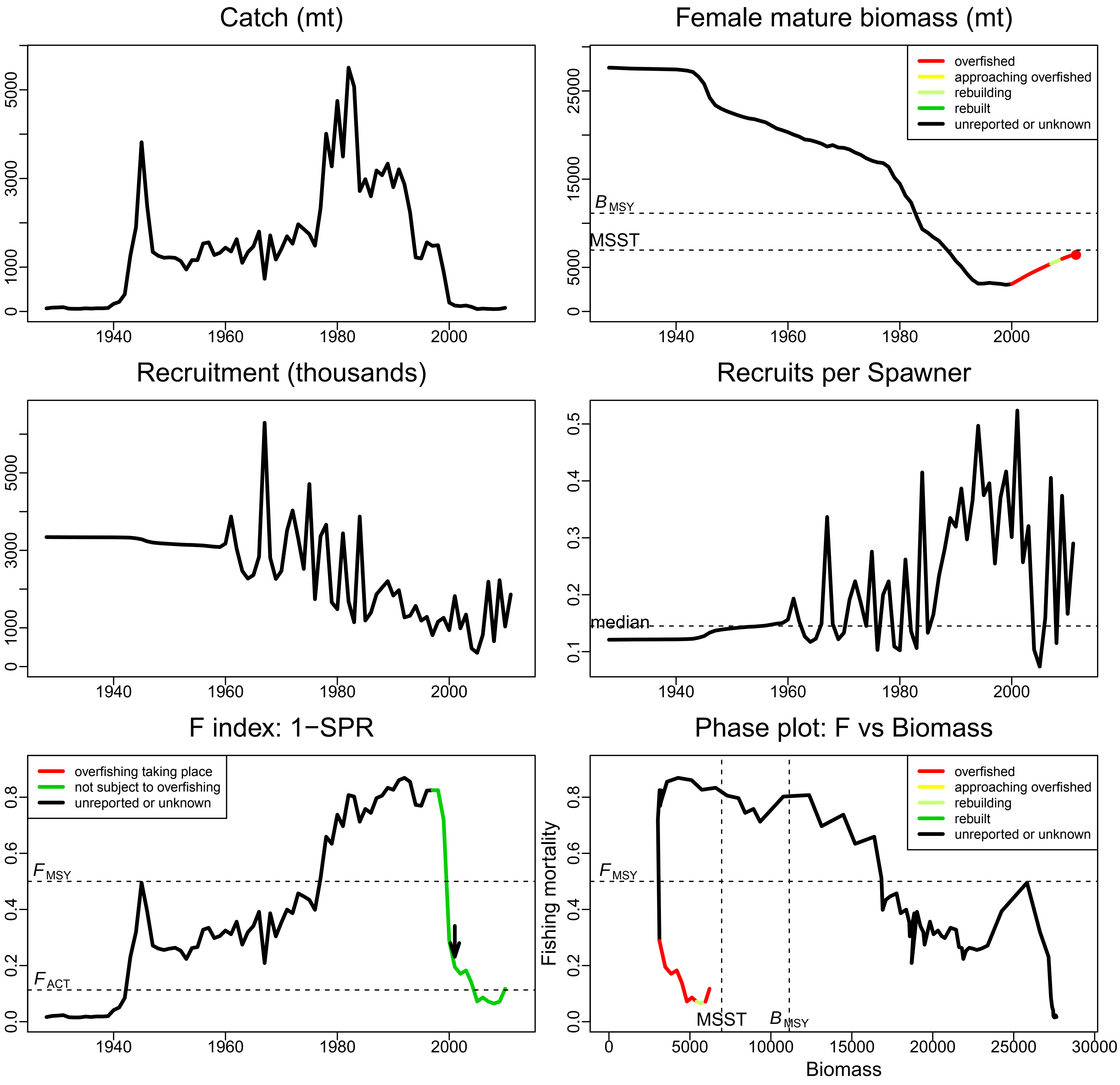
Recruitment (thousands)

Recruits per Spawner

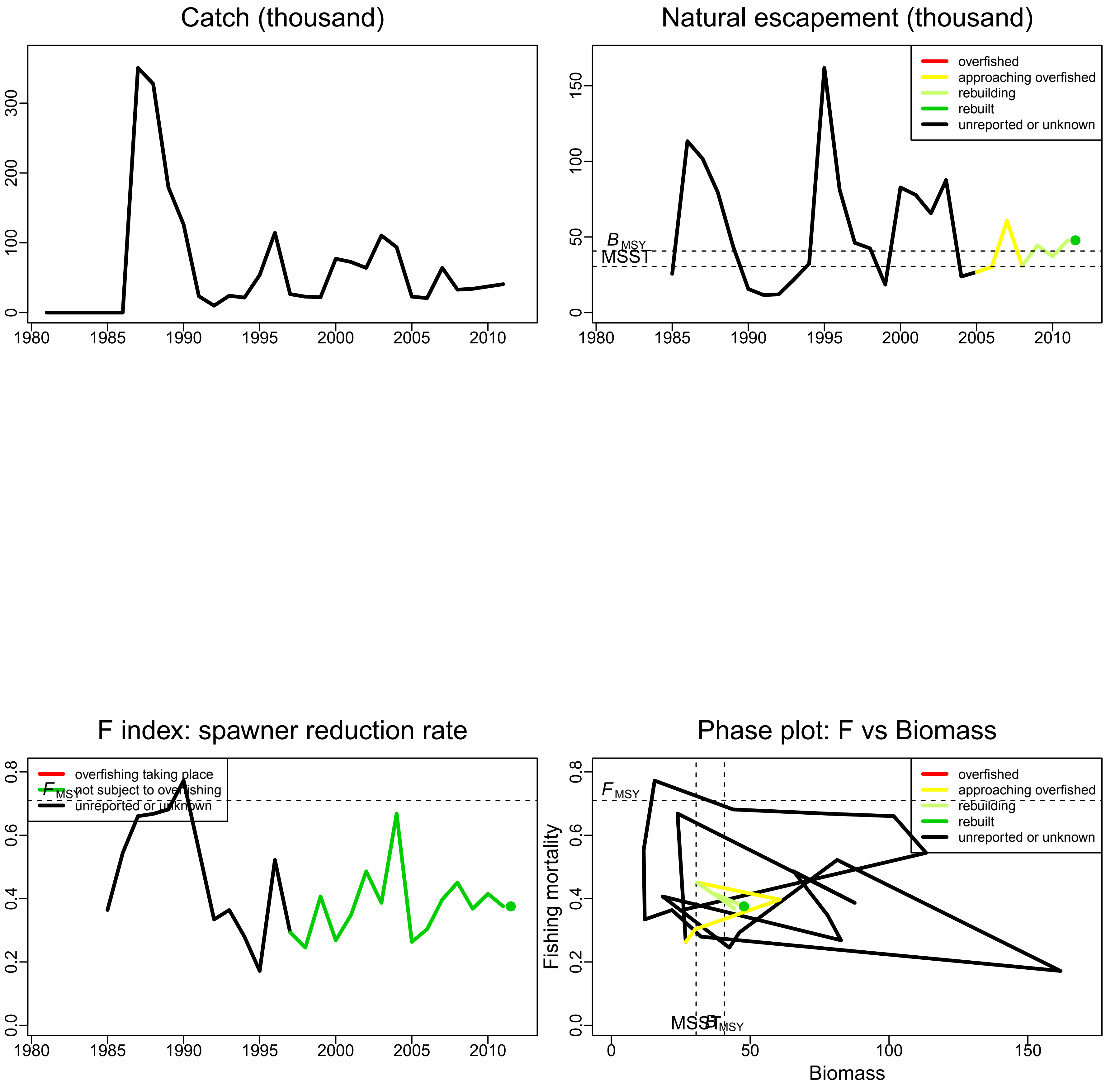
F index: 1−SPR

Phase plot: F vs Biomass

C.16: Canary rockfish − Pacific Coast



C.17: Chinook salmon − Northern California Coast: Klamath (fall)



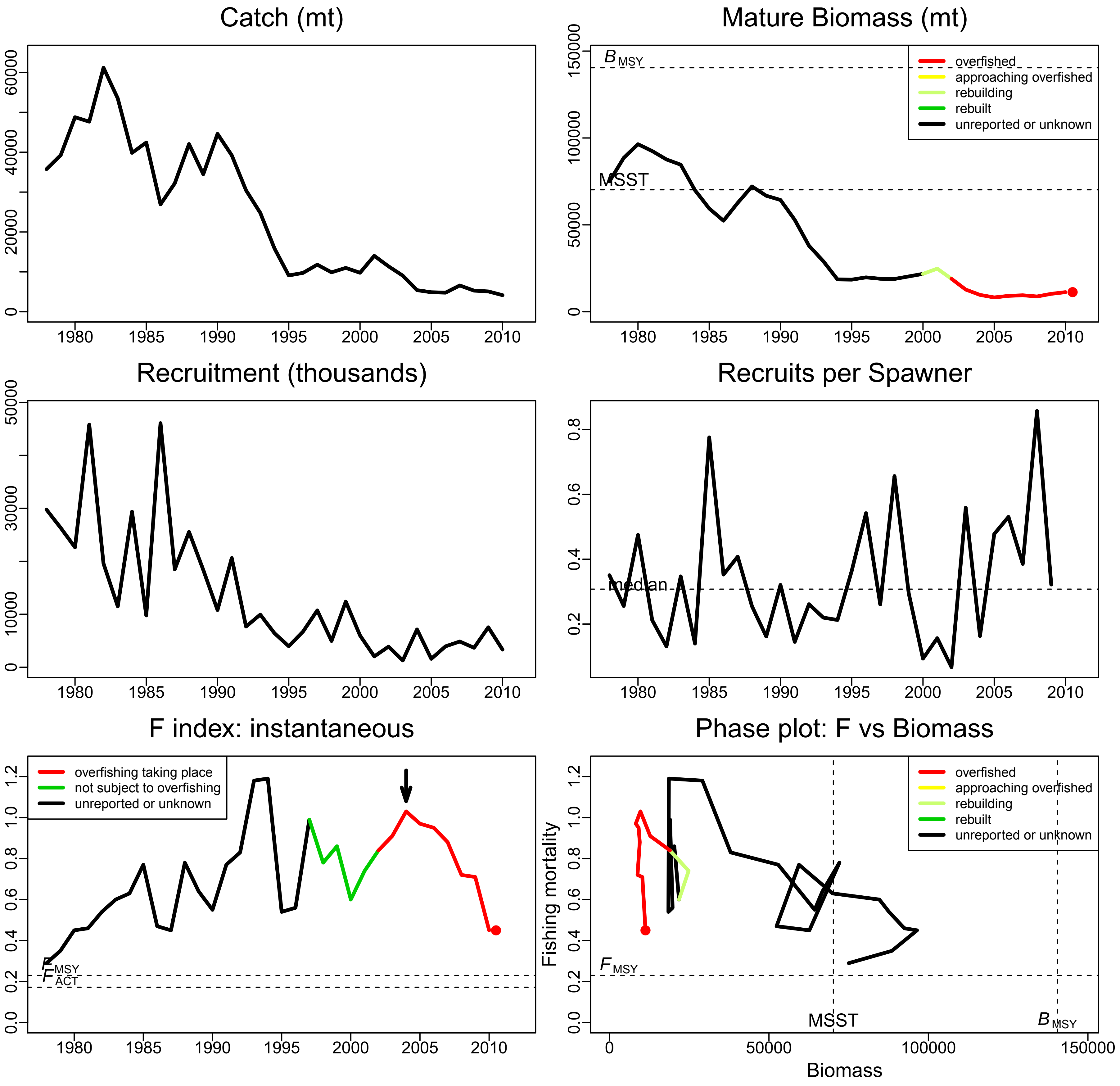
Catch (thousand)

Natural escapement (thousand)

F index: spawner reduction rate

Phase plot: F vs Biomass

C.18: Atlantic cod − Georges Bank



Catch (mt)

Mature Biomass (mt)

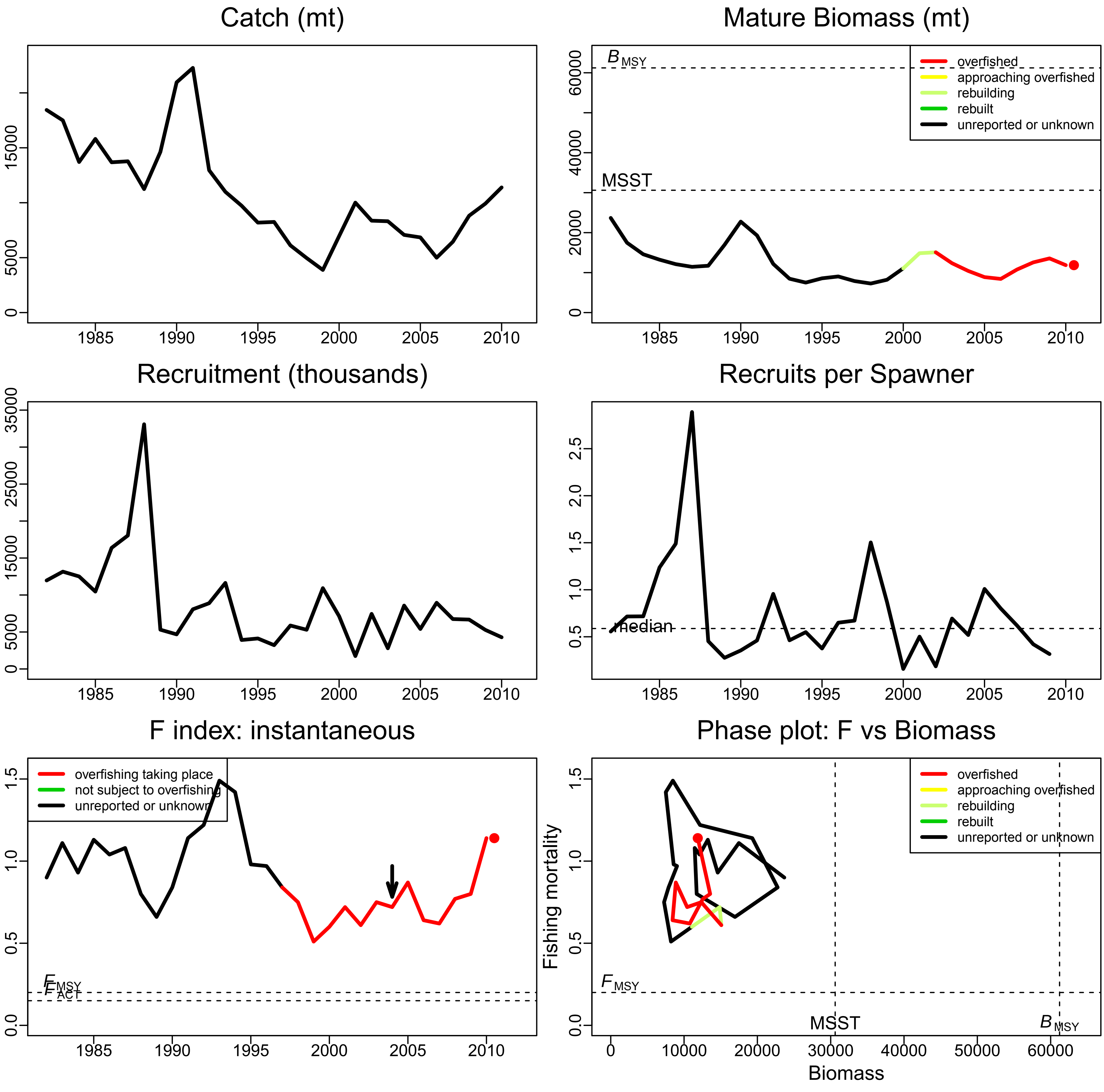
Recruitment (thousands)

Recruits per Spawner

F index: instantaneous

Phase plot: F vs Biomass

C.19: Atlantic cod − Gulf of Maine



Catch (mt)

Mature Biomass (mt)

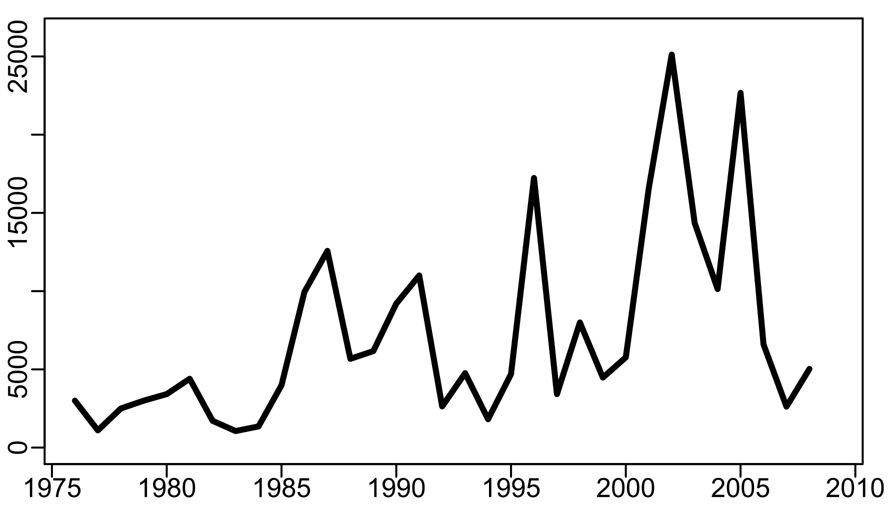
Recruitment (thousands)

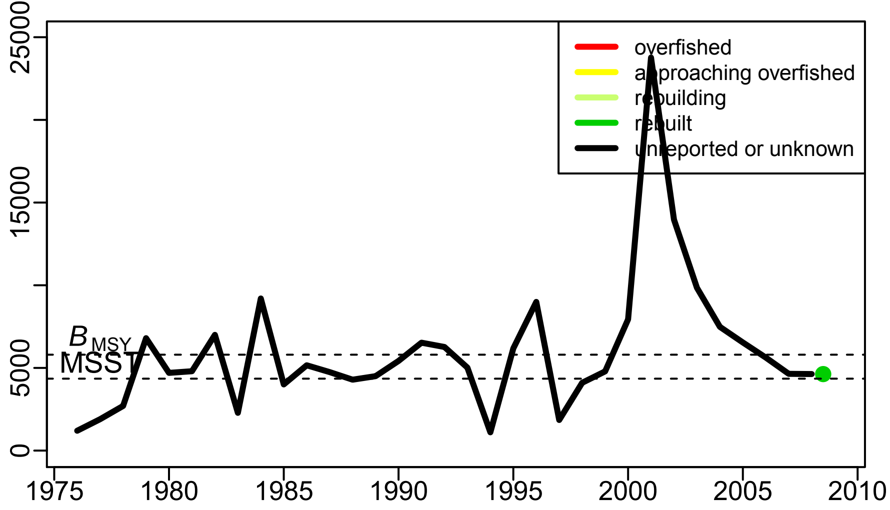
Recruits per Spawner

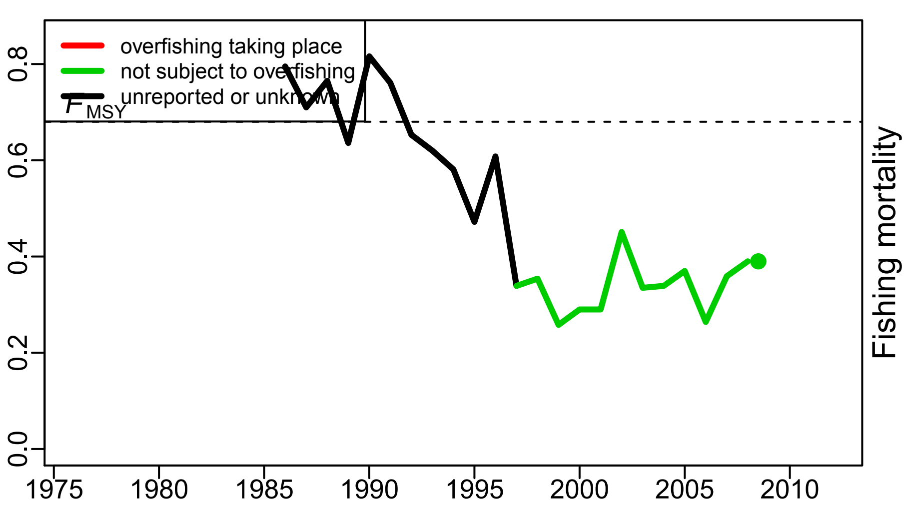
F index: instantaneous

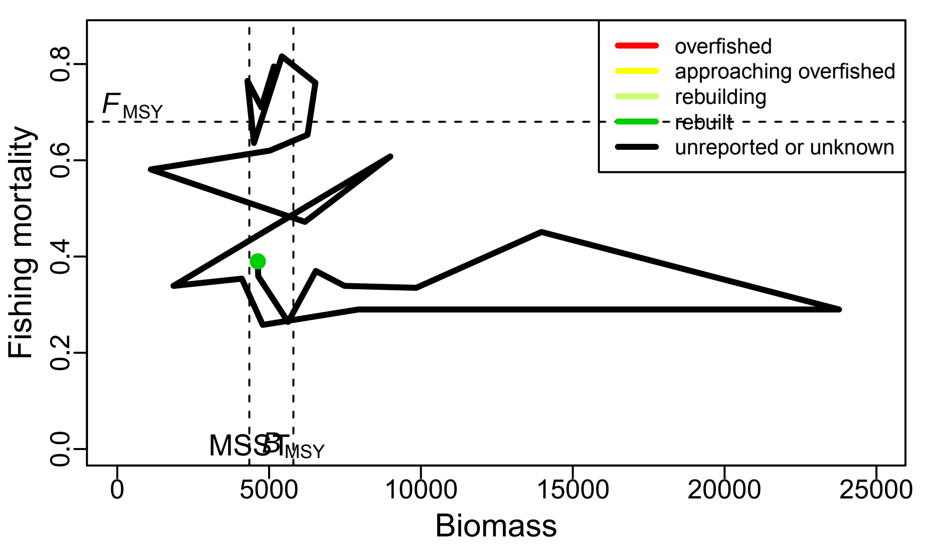
Phase plot: F vs Biomass

C.20: Coho salmon − Washington Coast: Queets

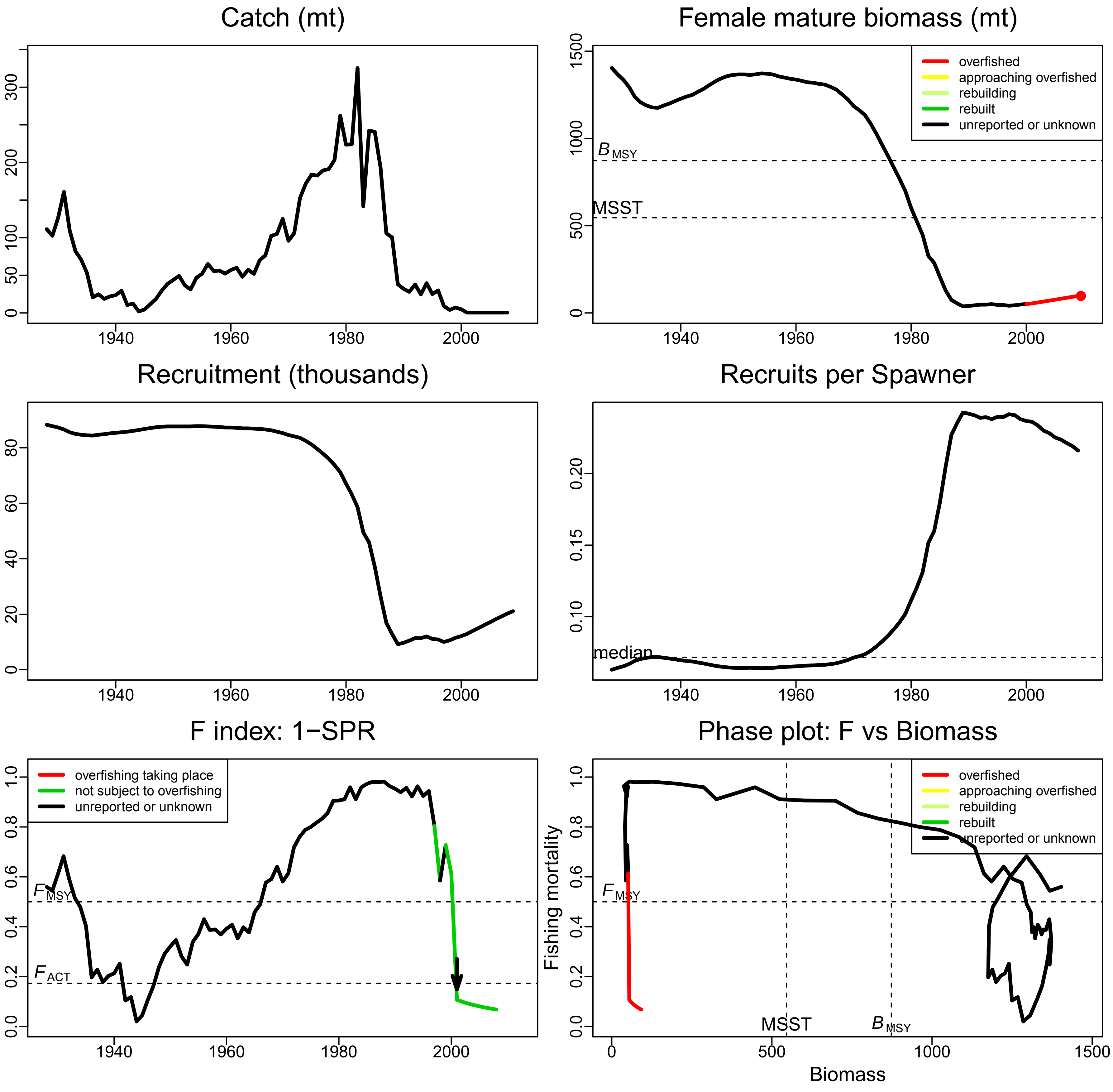

Catch (units)


Natural escapement (units)

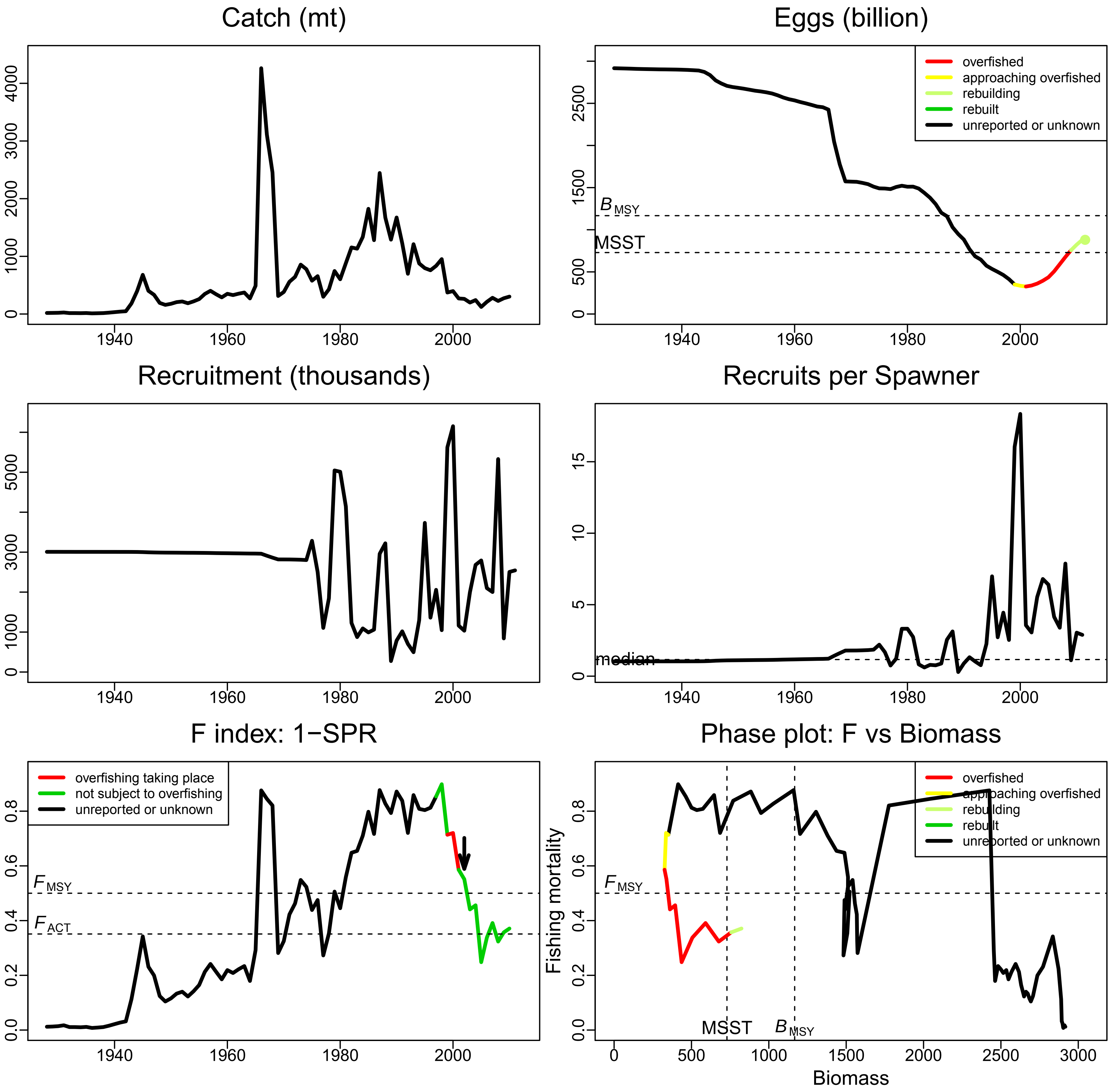

F index: exploitation rate


Phase plot: F vs Biomass

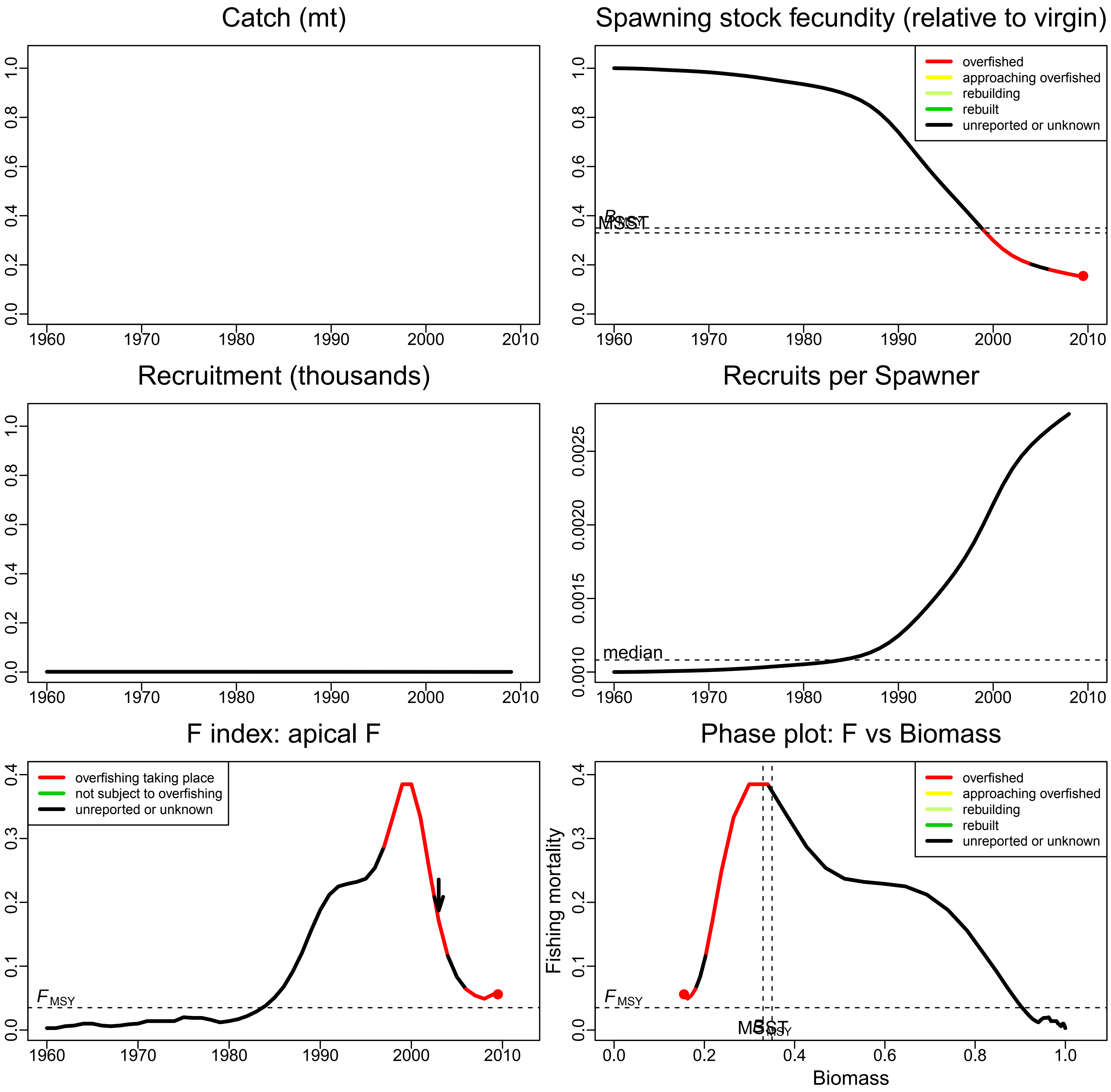
C.21: Cowcod − Southern California



C.22: Darkblotched rockfish − Pacific Coast



C.23: Dusky shark − Atlantic



Catch (mt)

Spawning stock fecundity (relative to virgin)

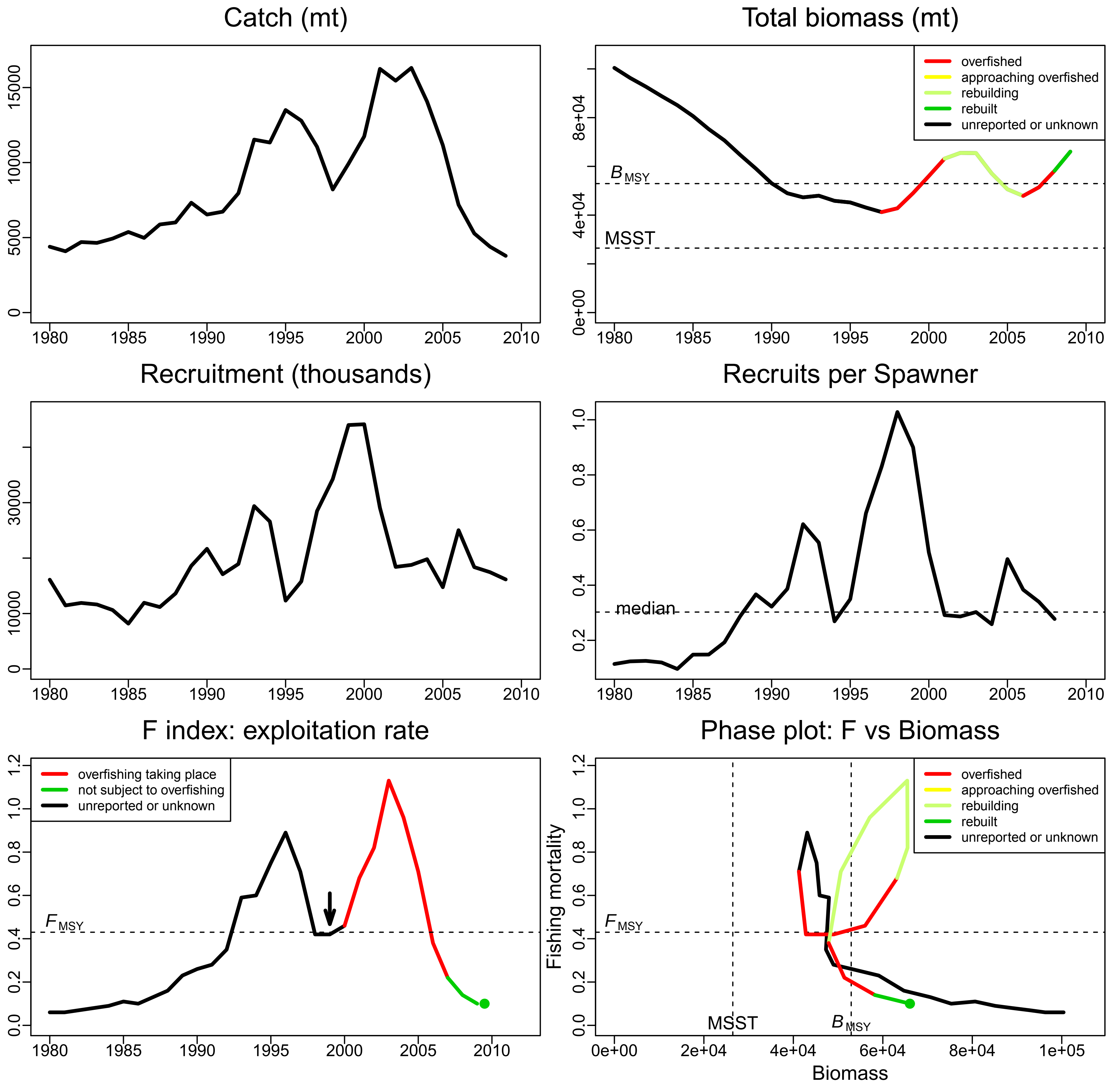
Recruitment (thousands)

Recruits per Spawner

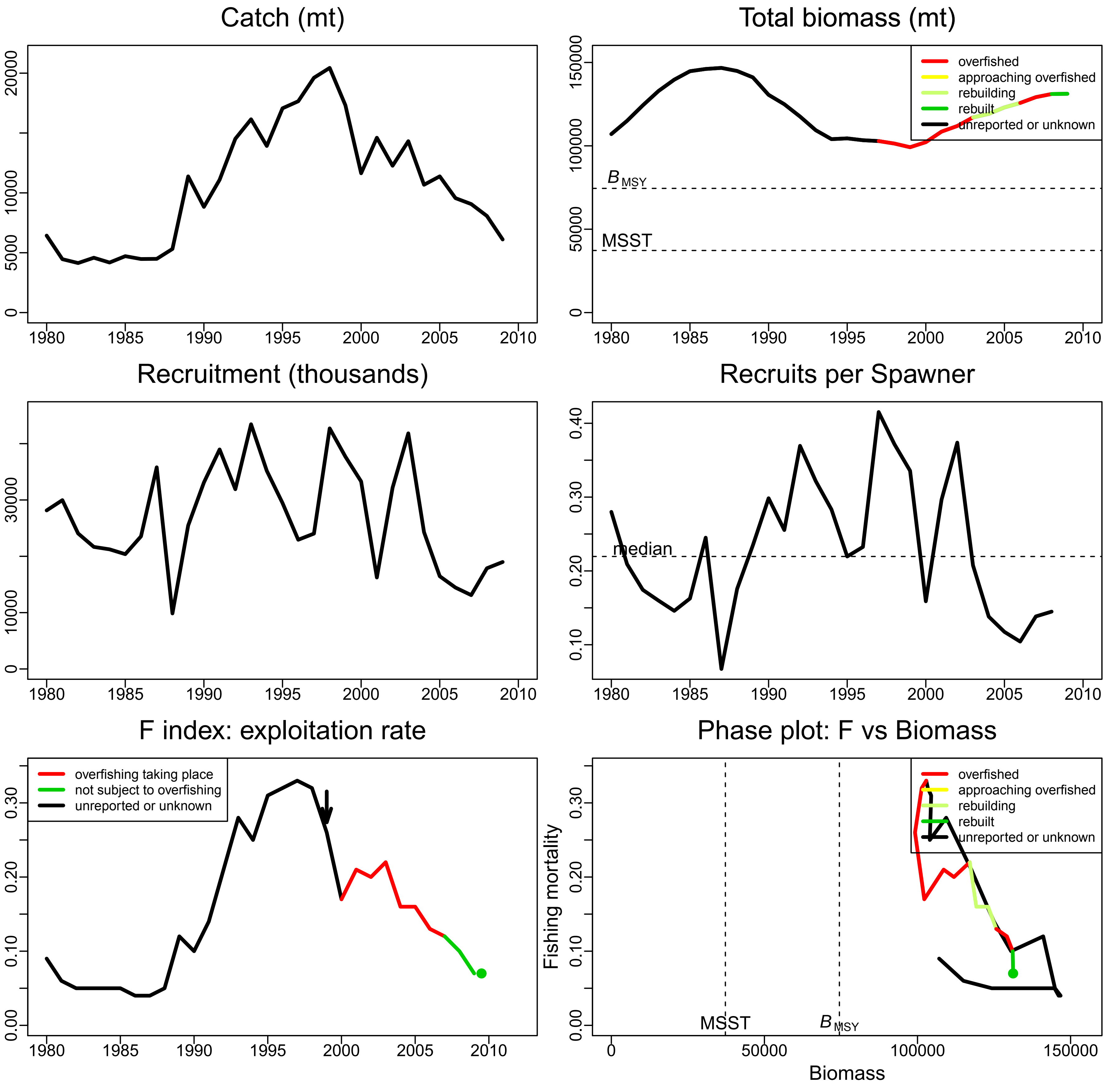
F index: apical F

Phase plot: F vs Biomass

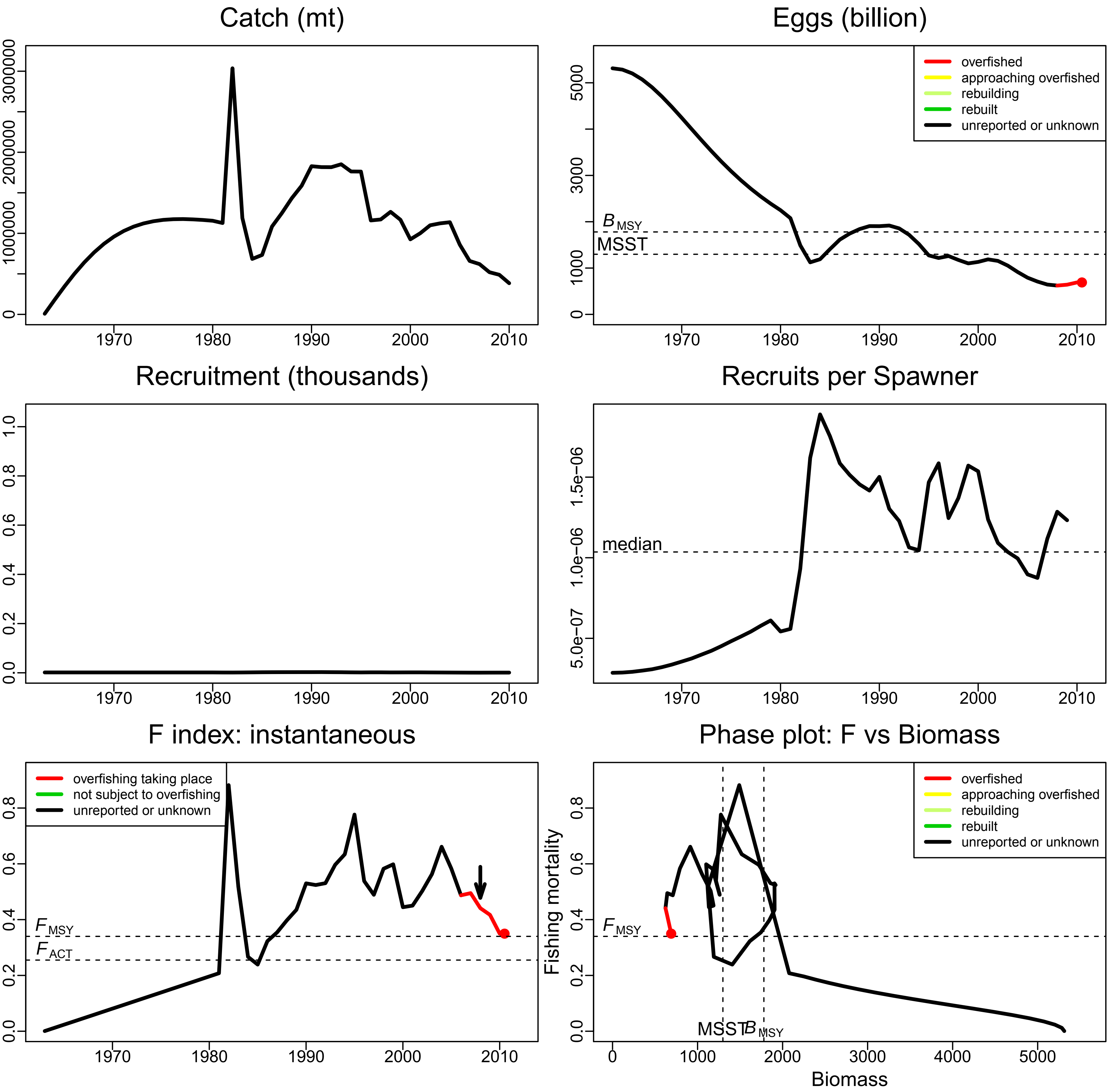
C.24: Goosefish − Gulf of Maine / Northern Georges Bank



C.25: Goosefish − Southern Georges Bank / Mid−Atlantic

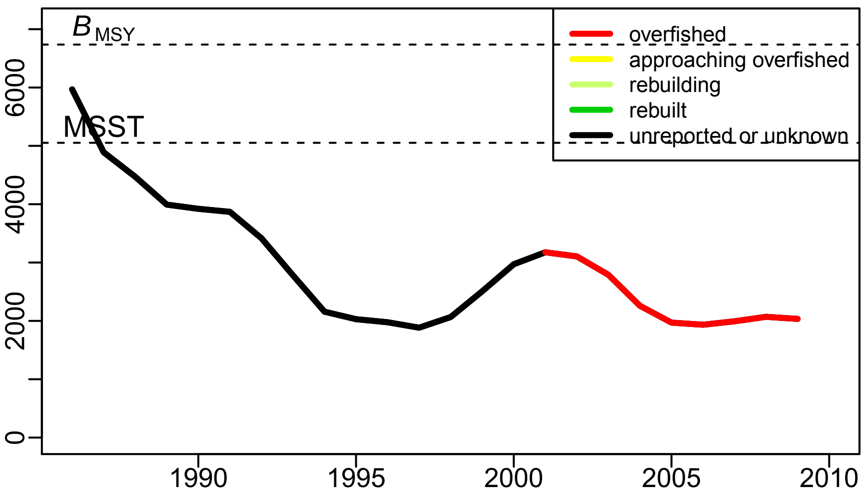


C.26: Gray triggerfish − Gulf of Mexico



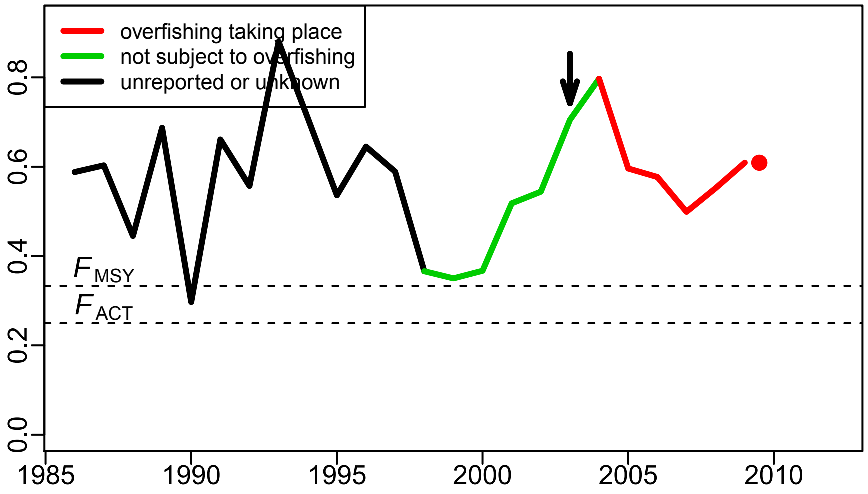
C.27: Greater amberjack − Gulf of Mexico

Catch (mt)

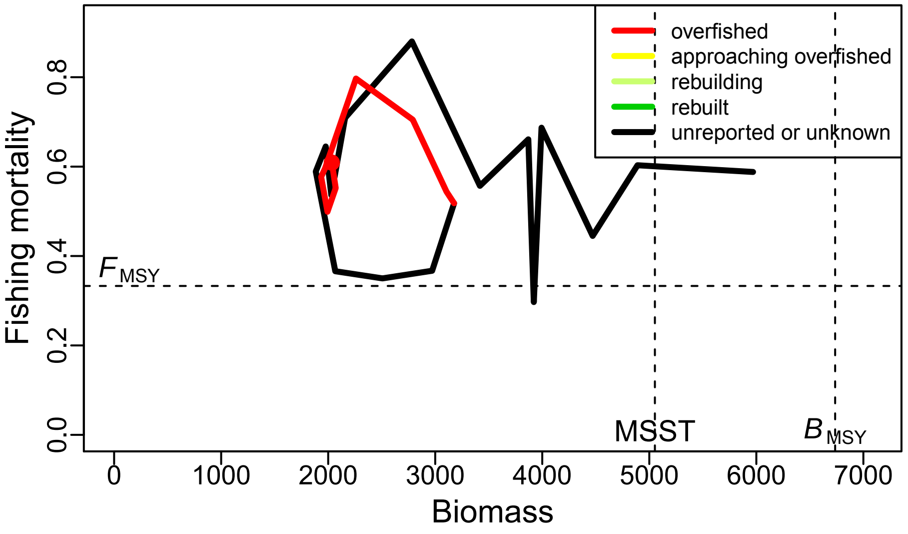


Total biomass (mt)

Exploitable biomass (mt)

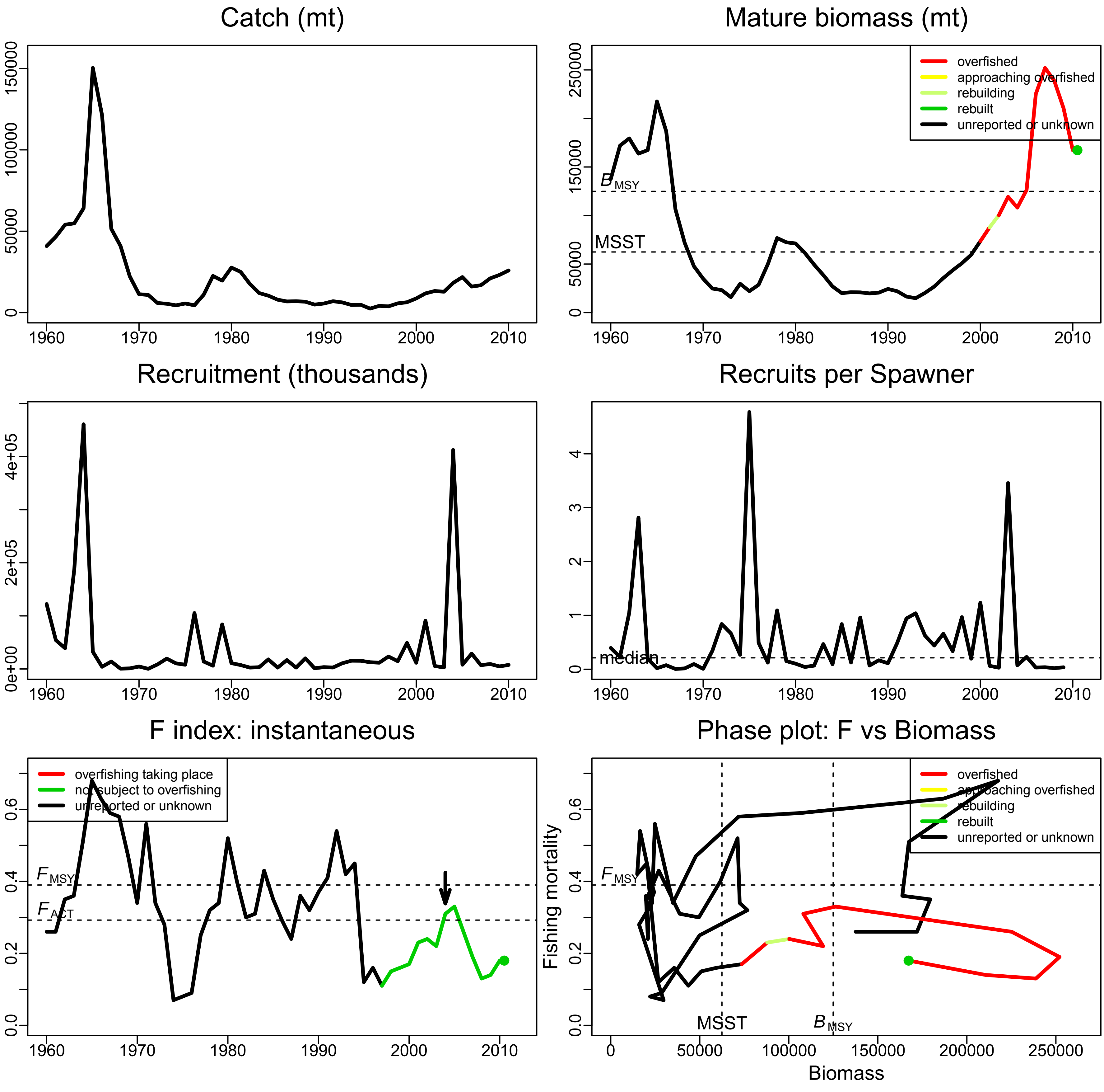


F index: instantaneous



Phase plot: F vs Biomass

C.28: Haddock − Georges Bank



Catch (mt)

Mature biomass (mt)

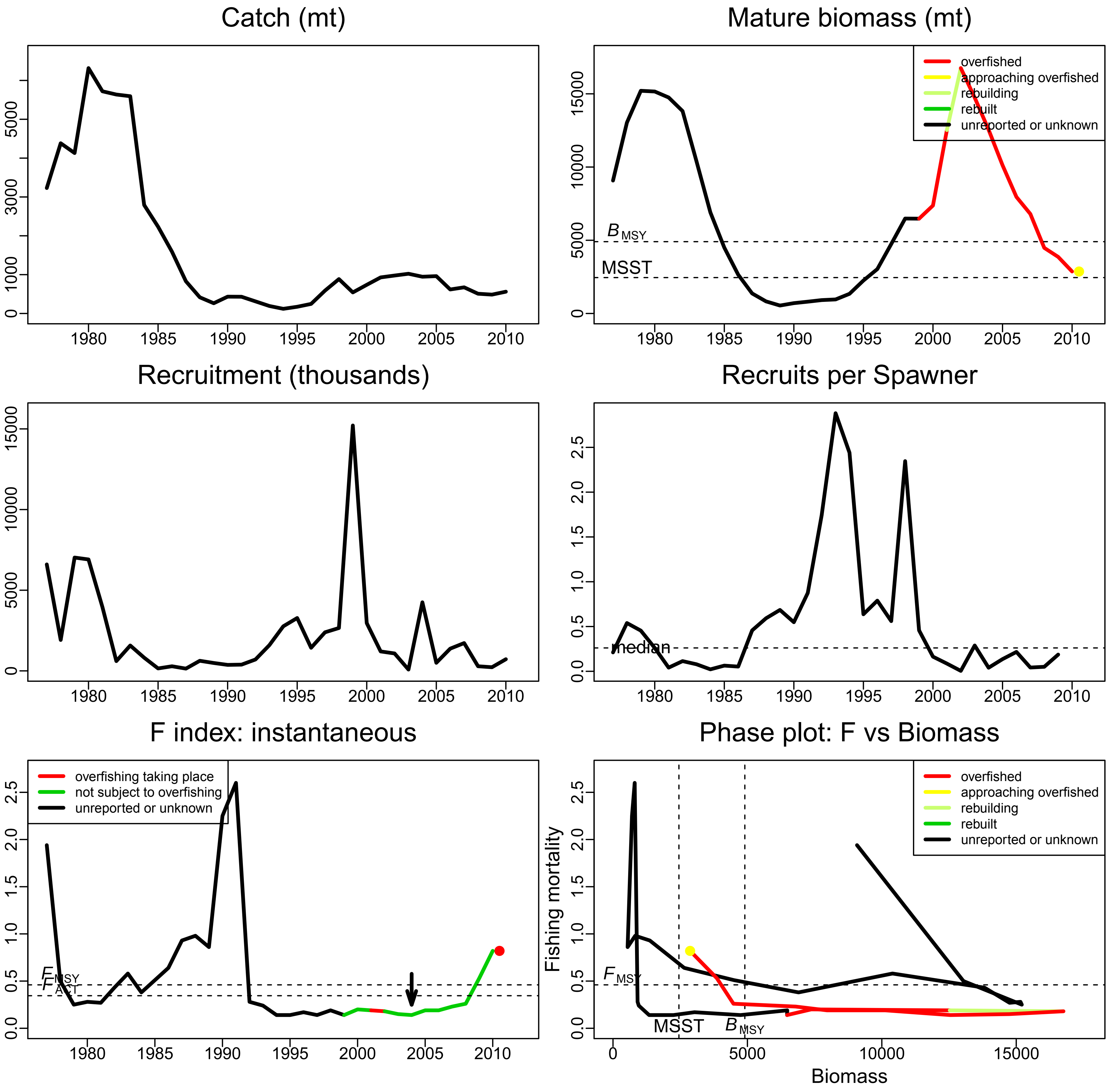
Recruitment (thousands)

Recruits per Spawner

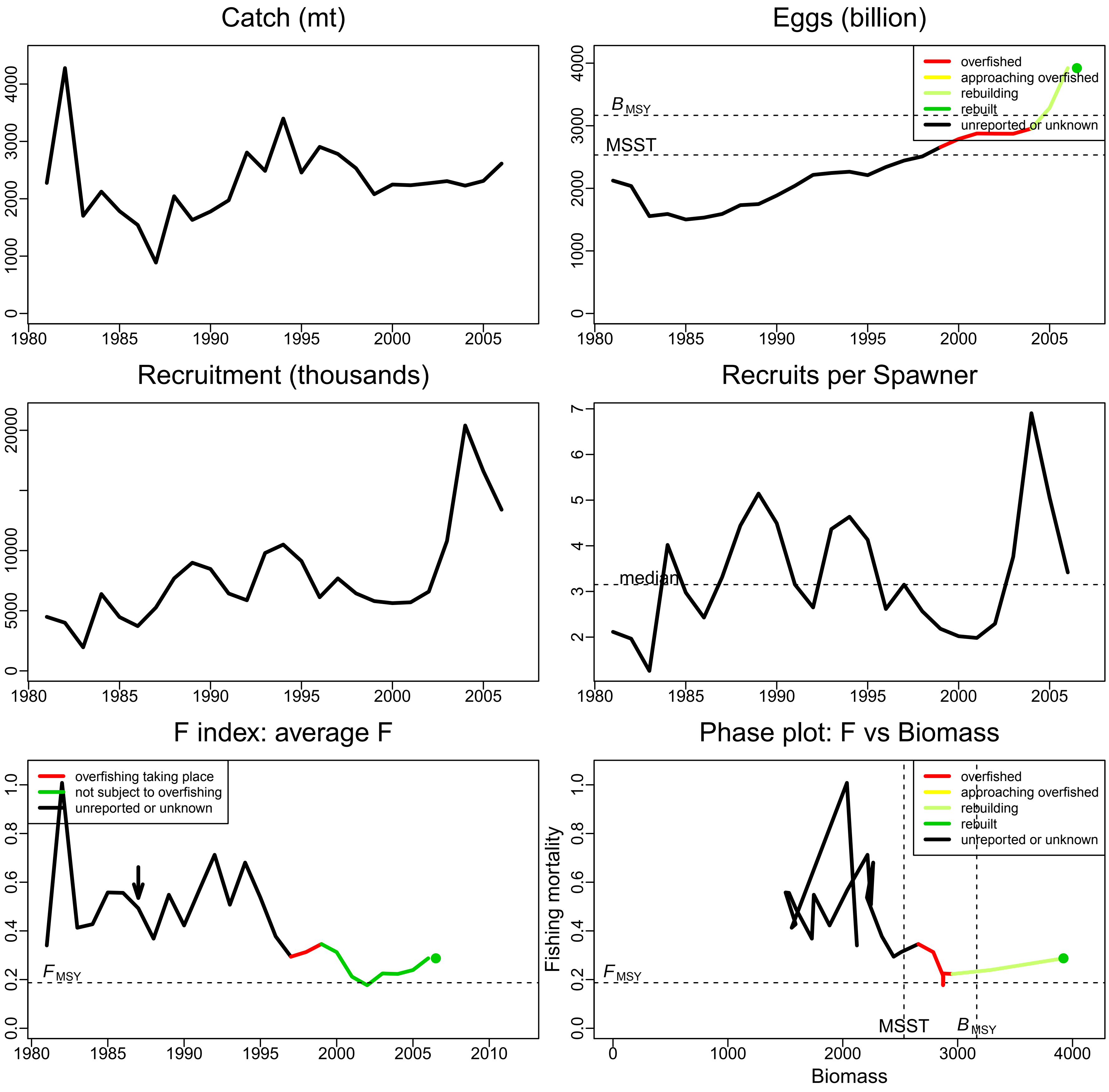
F index: instantaneous

Phase plot: F vs Biomass

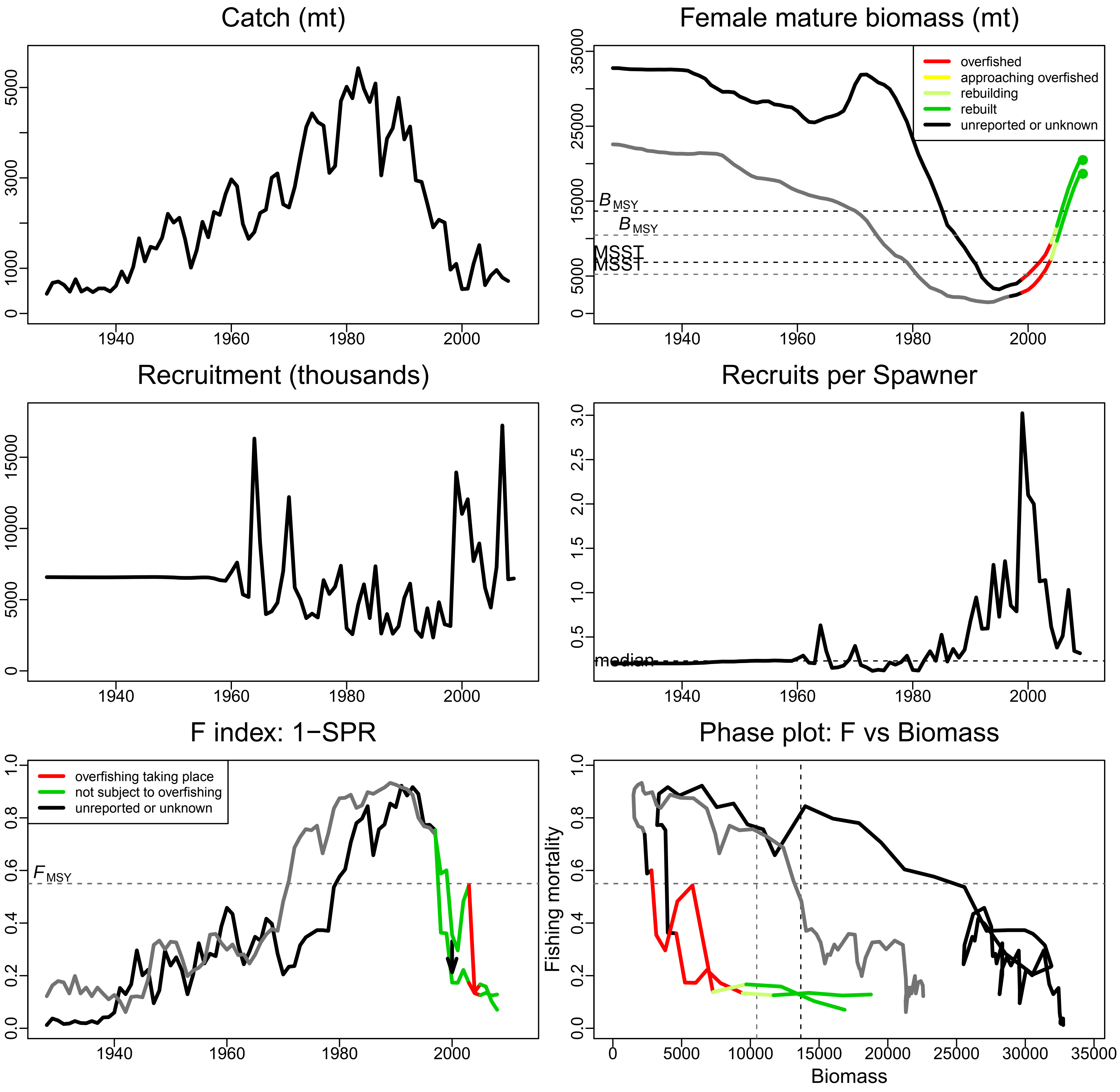
C.29: Haddock − Gulf of Maine



C.30: King mackerel − Gulf of Mexico



C.31: Lingcod − Pacific Coast



Catch (mt)

Female mature biomass (mt)

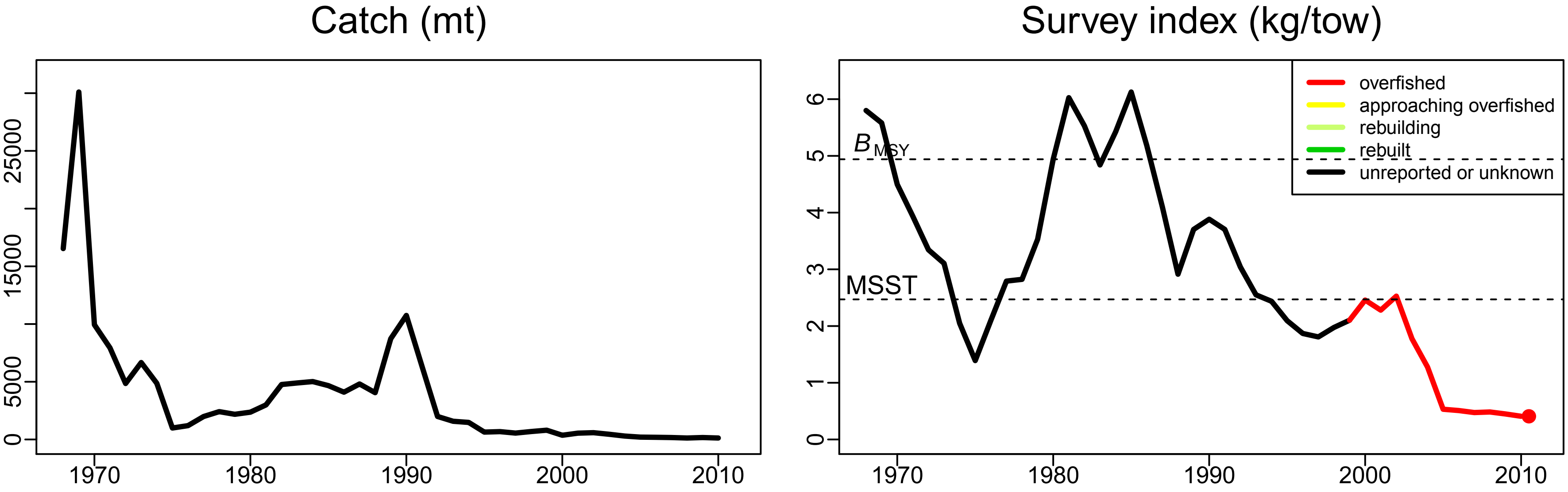
Recruitment (thousands)

Recruits per Spawner

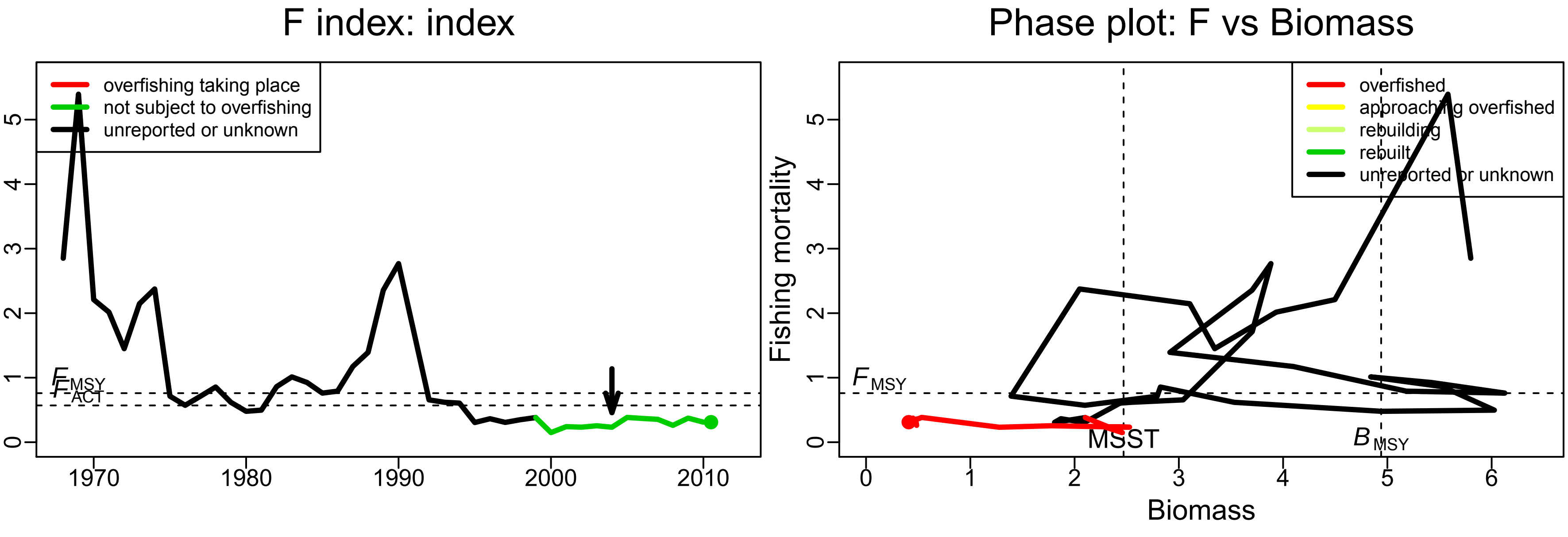
F index: 1−SPR

Phase plot: F vs Biomass

C.32: Ocean pout − Northwestern Atlantic Coast

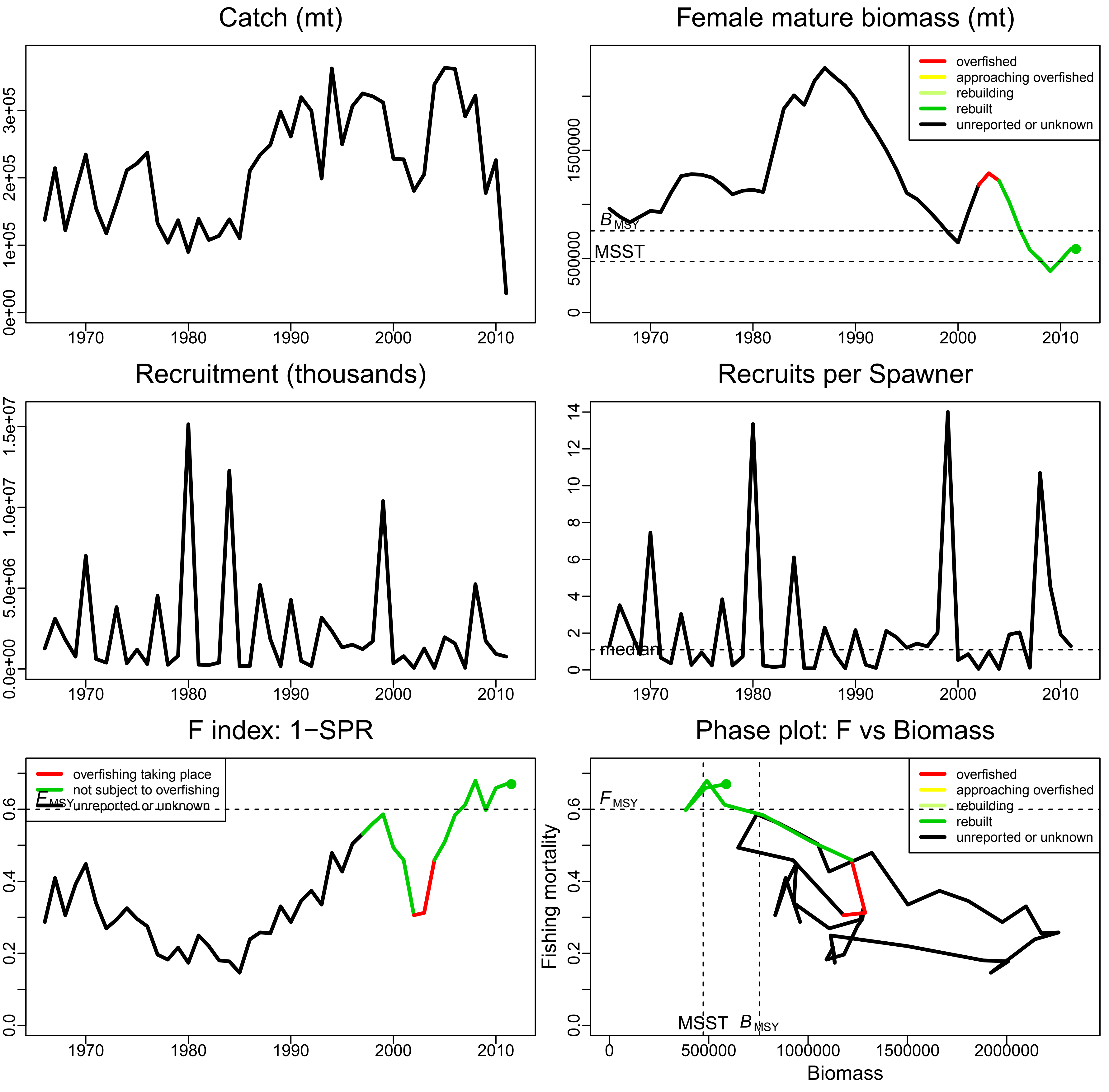

Catch (mt)

Survey index (kg/tow)


F index: index

Phase plot: F vs Biomass

C.33: Pacific hake − Pacific Coast



Catch (mt)

Female mature biomass (mt)

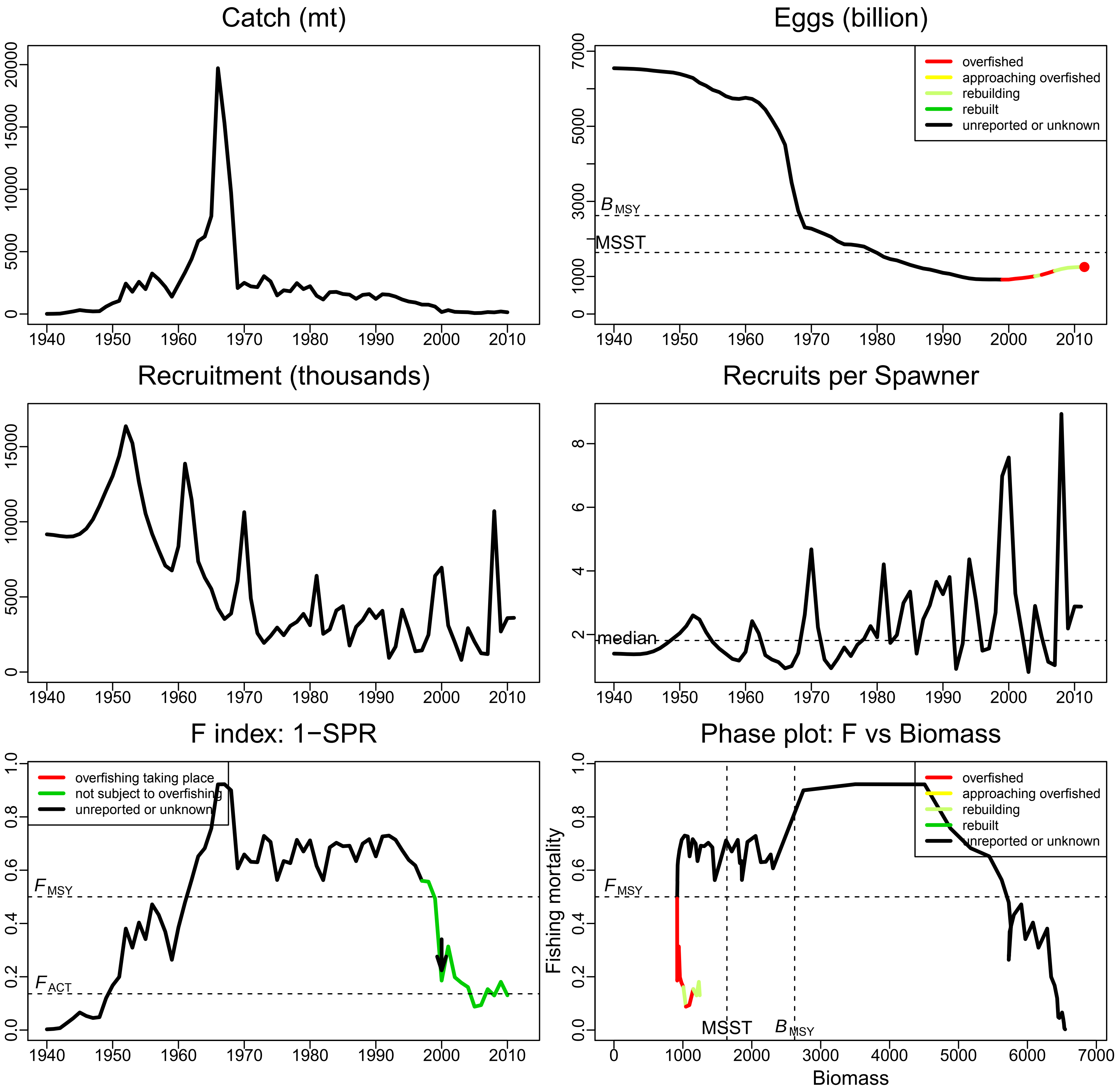
Recruitment (thousands)

Recruits per Spawner

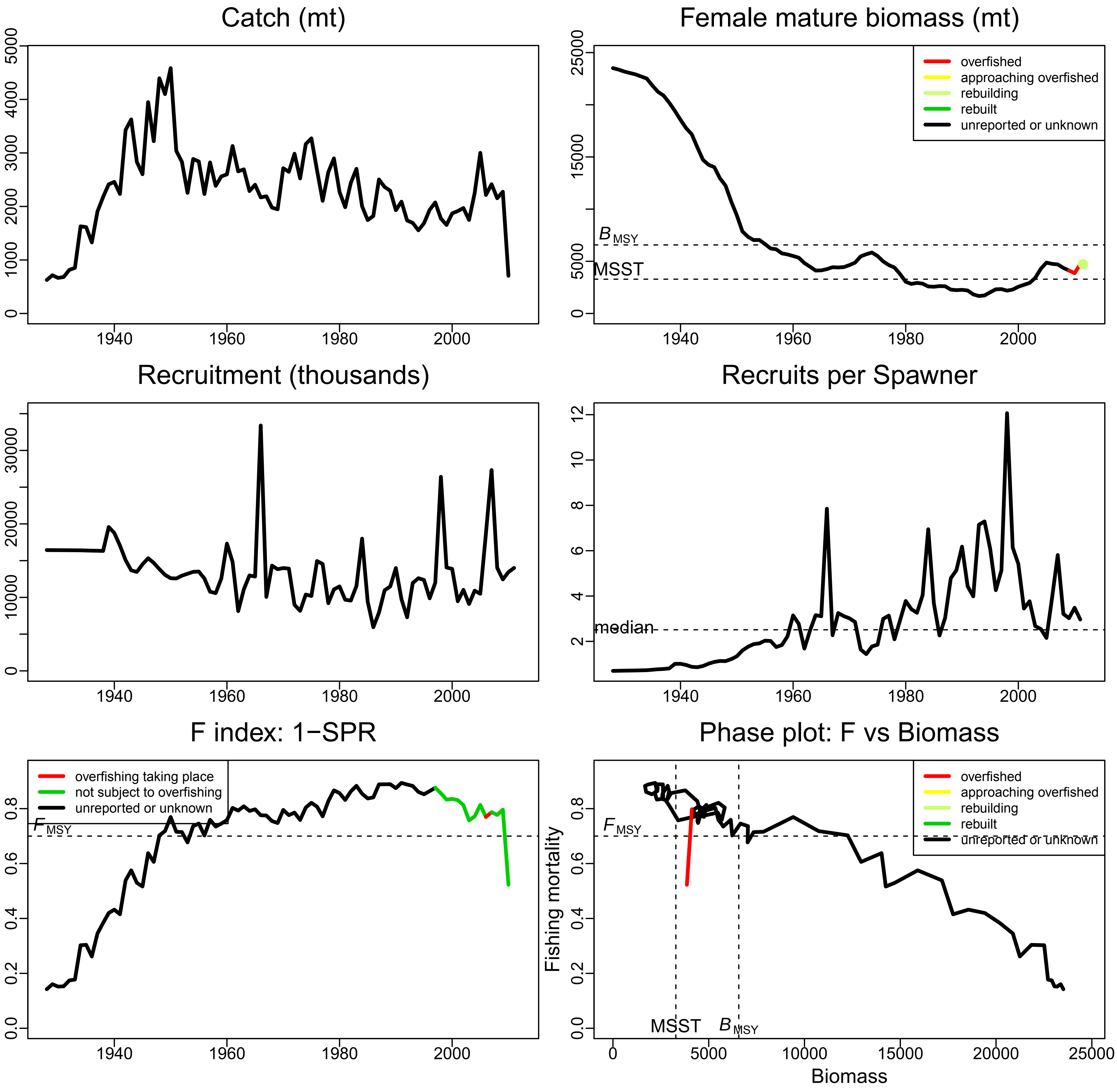
F index: 1−SPR

Phase plot: F vs Biomass

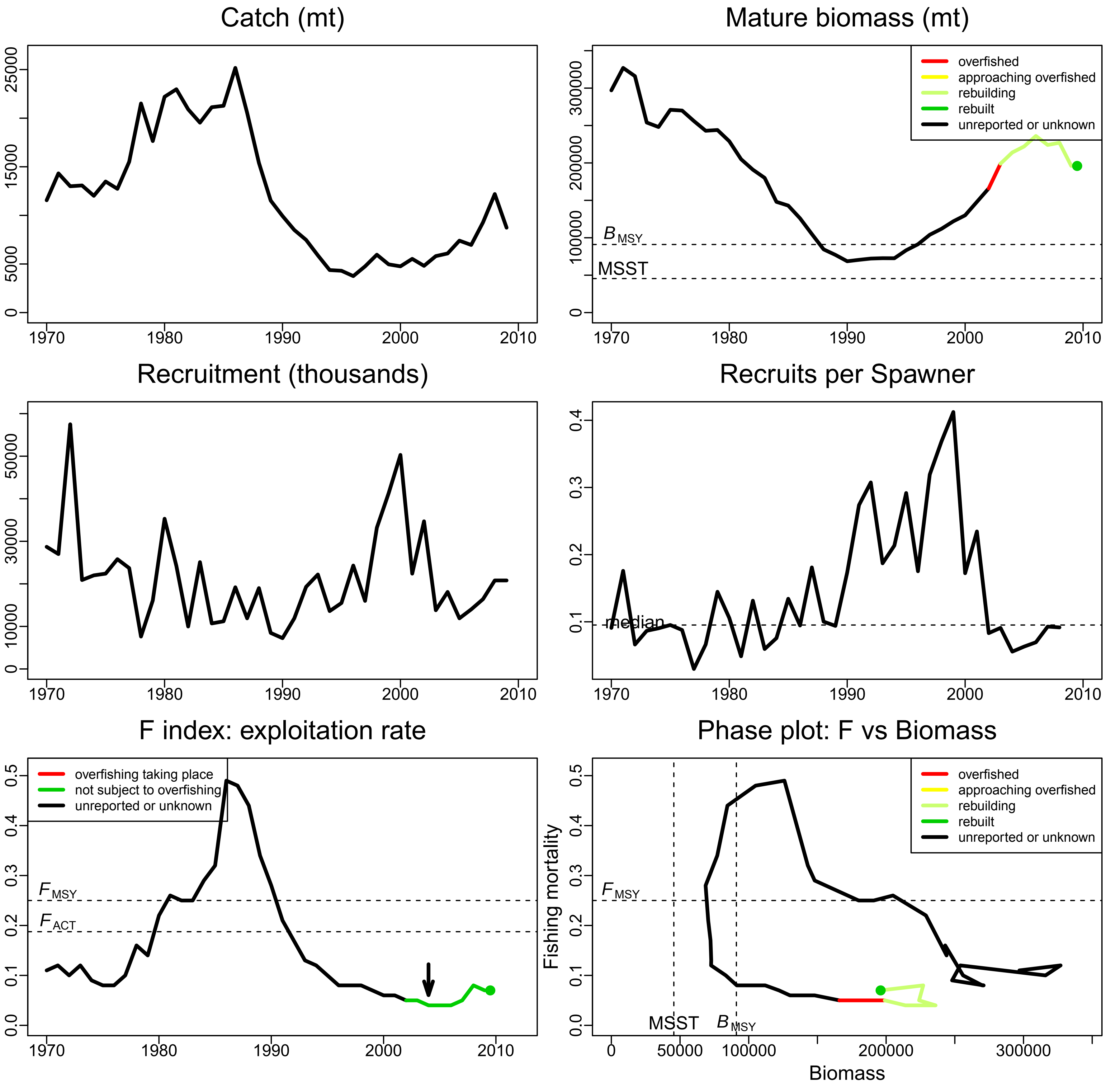
C.34: Pacific ocean perch − Pacific Coast



C.35: Petrale sole − Pacific Coast



C.36: Pollock − Gulf of Maine / Georges Bank



Catch (mt)

Mature biomass (mt)

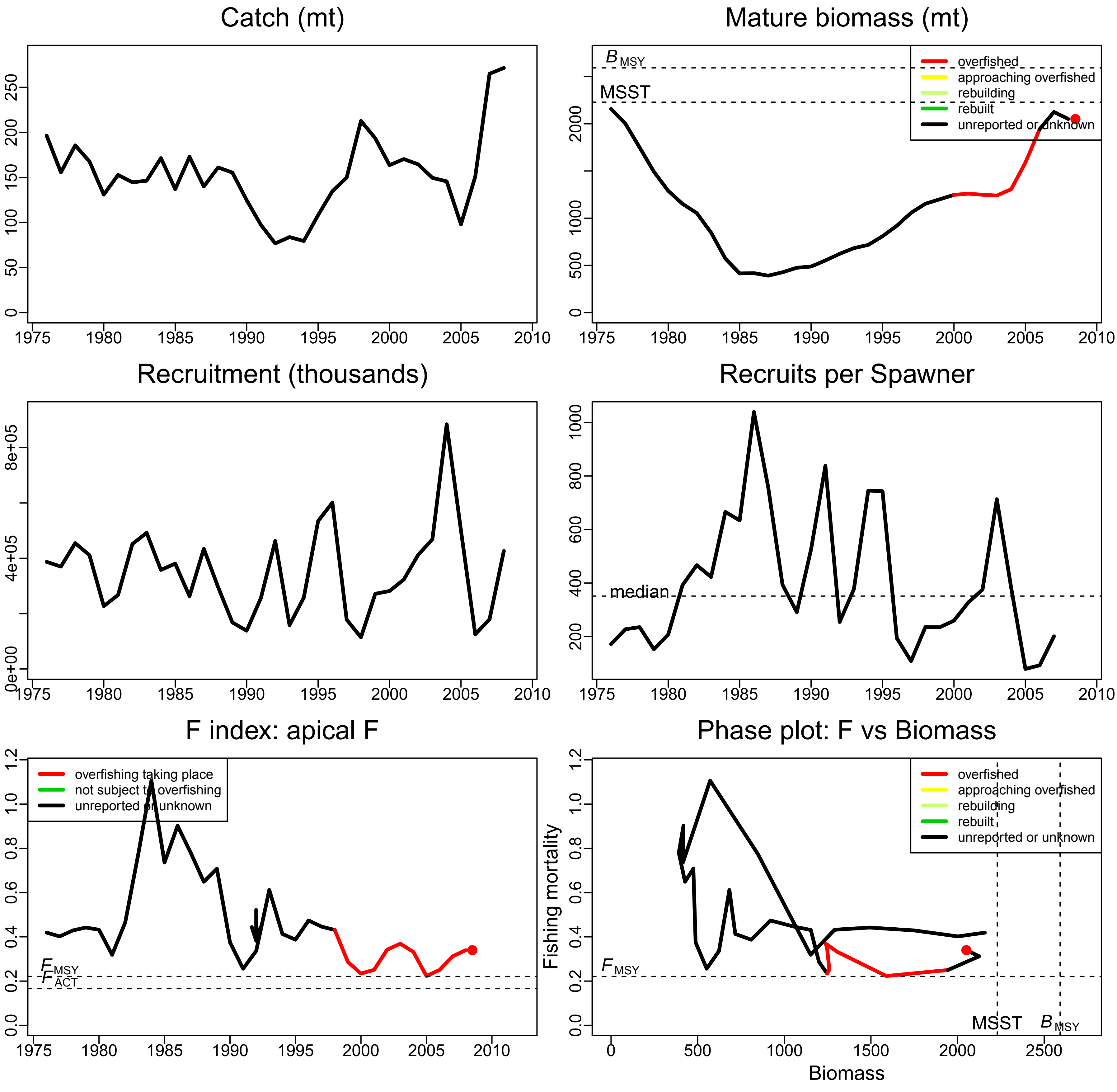
Recruitment (thousands)

Recruits per Spawner

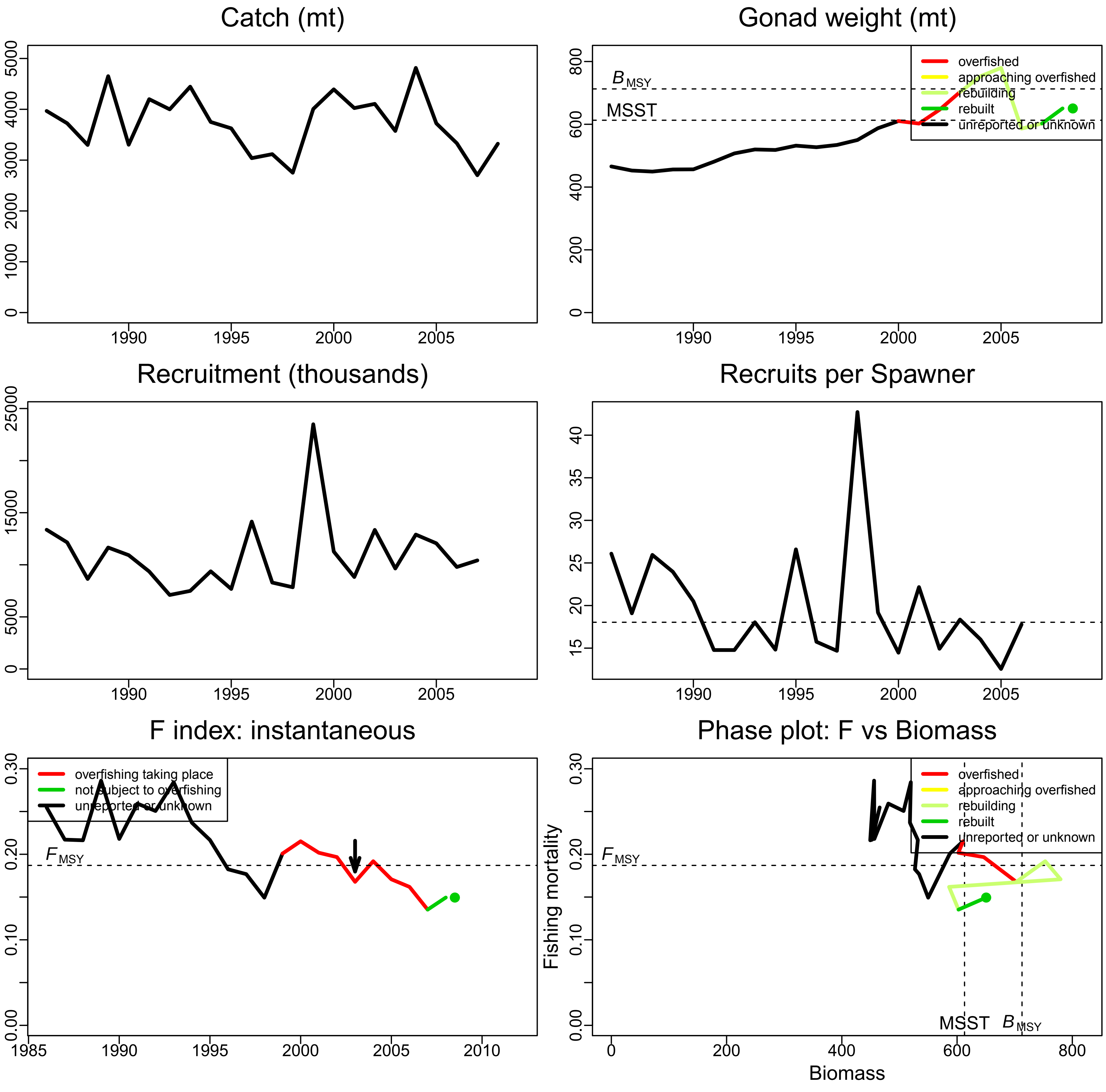
F index: exploitation rate

Phase plot: F vs Biomass

C.37: Red grouper − Southern Atlantic Coast



C.38: Red grouper − Gulf of Mexico



Catch (mt)

Gonad weight (mt)

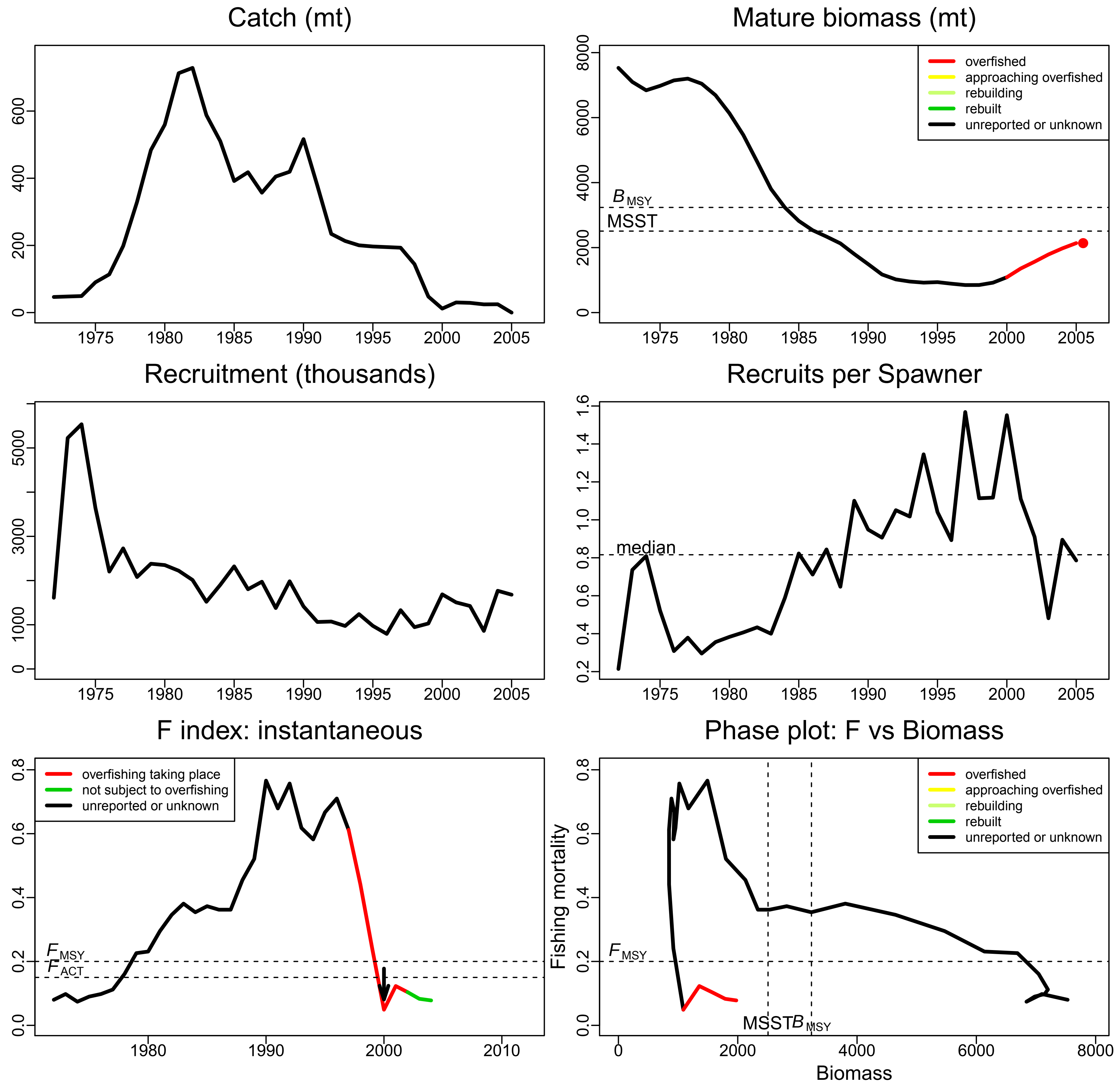
Recruitment (thousands)

Recruits per Spawner

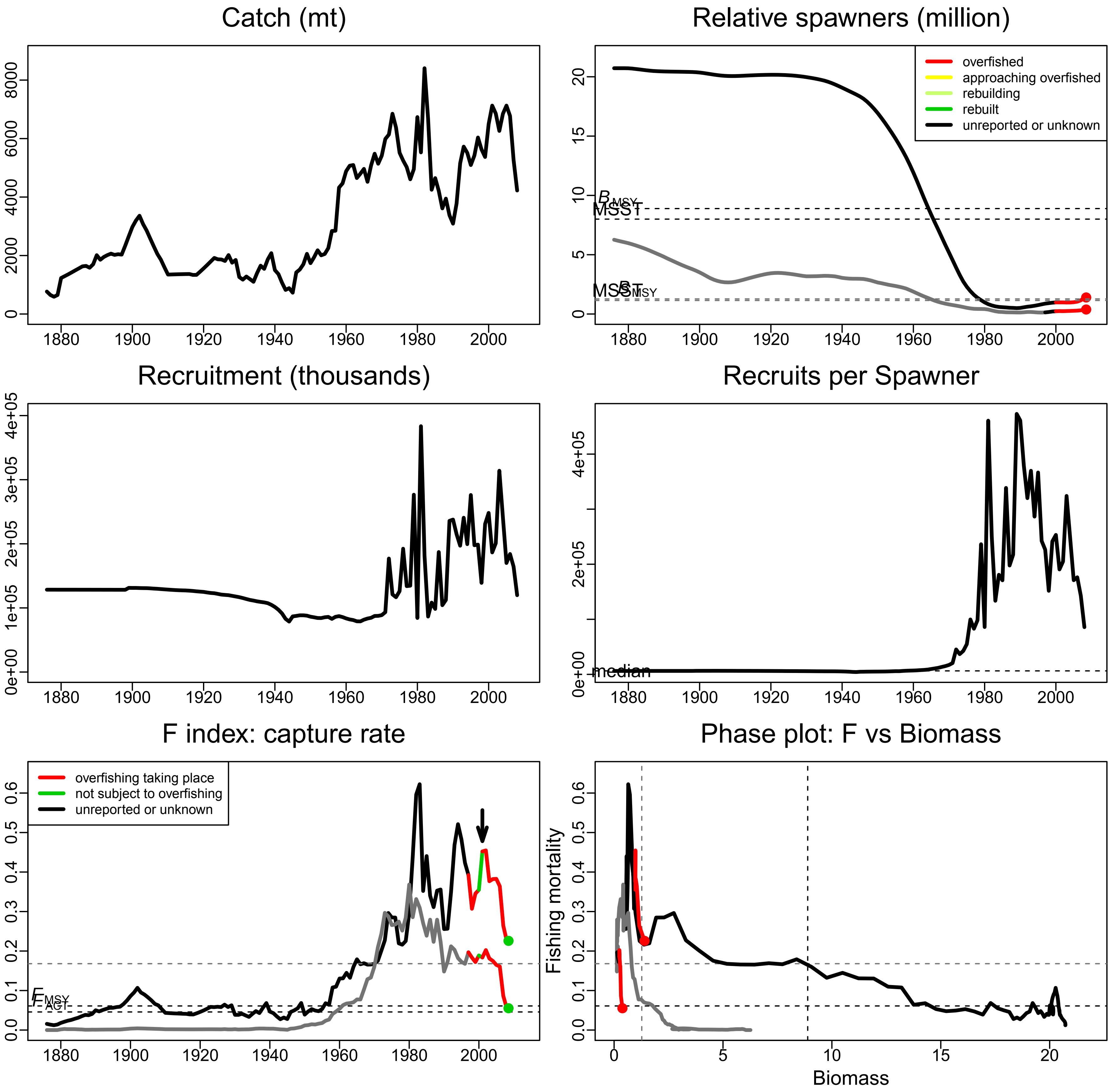
F index: instantaneous

Phase plot: F vs Biomass

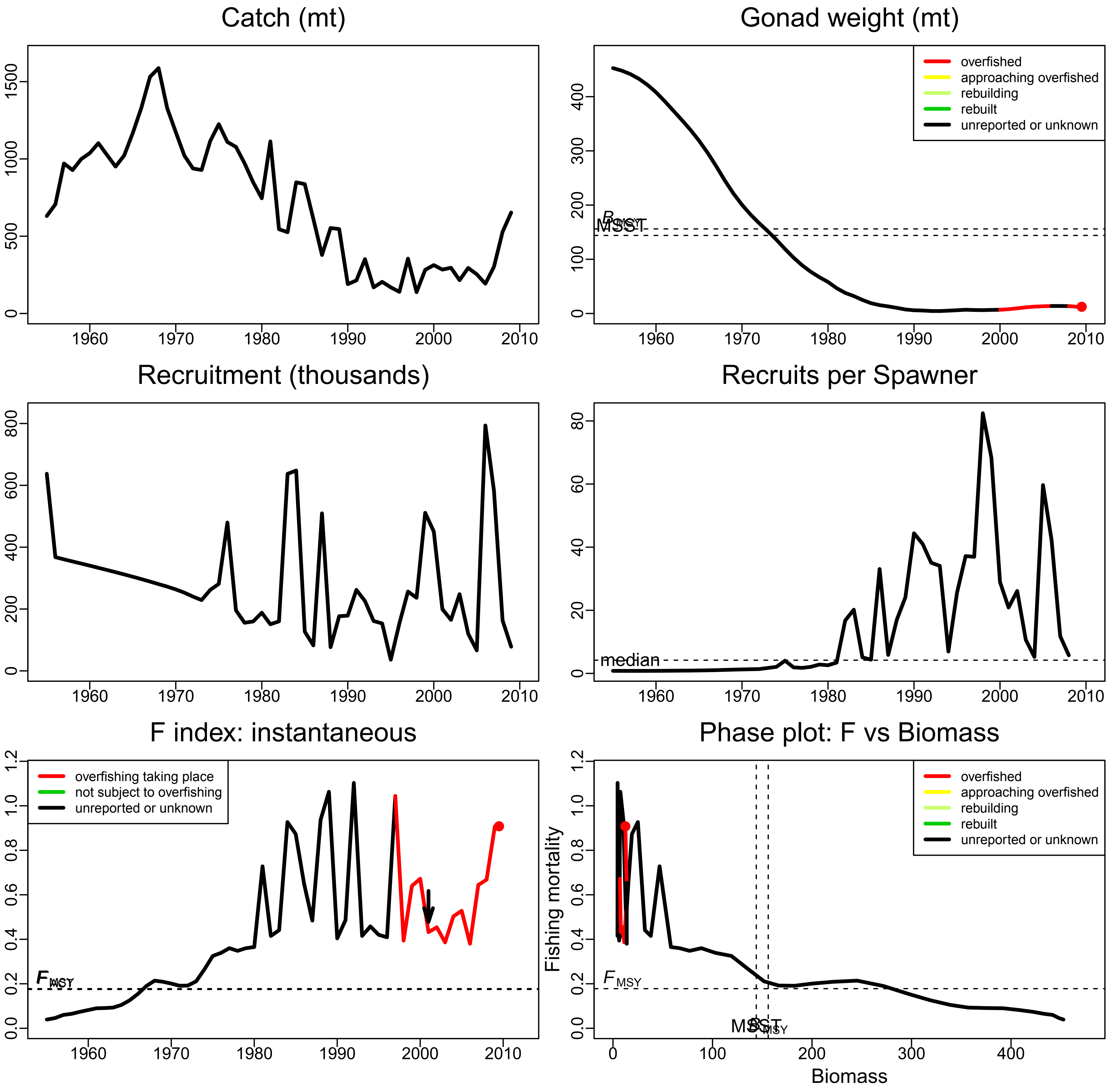
C.39: Red porgy − Southern Atlantic Coast



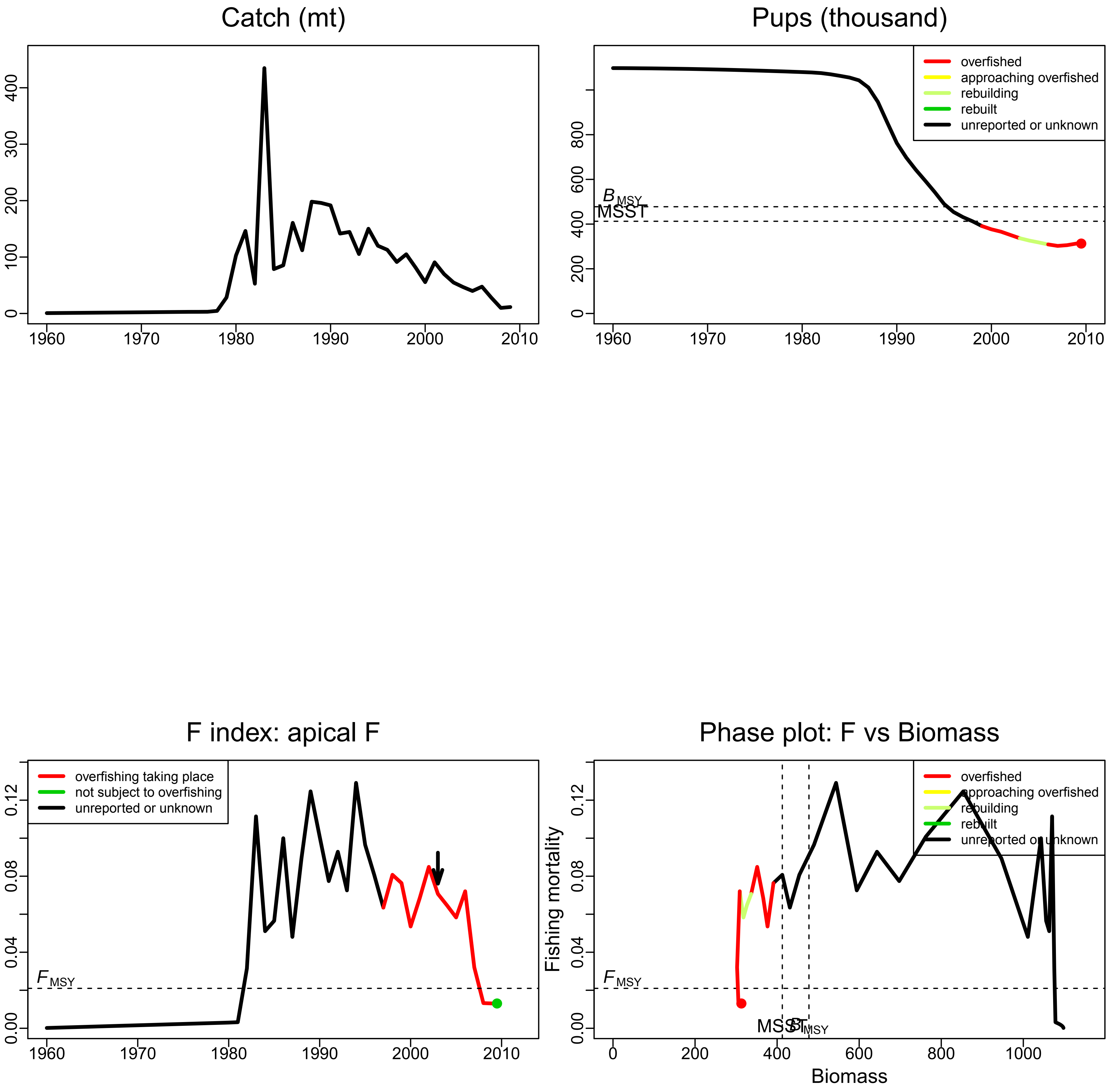
C.40: Red snapper − Gulf of Mexico



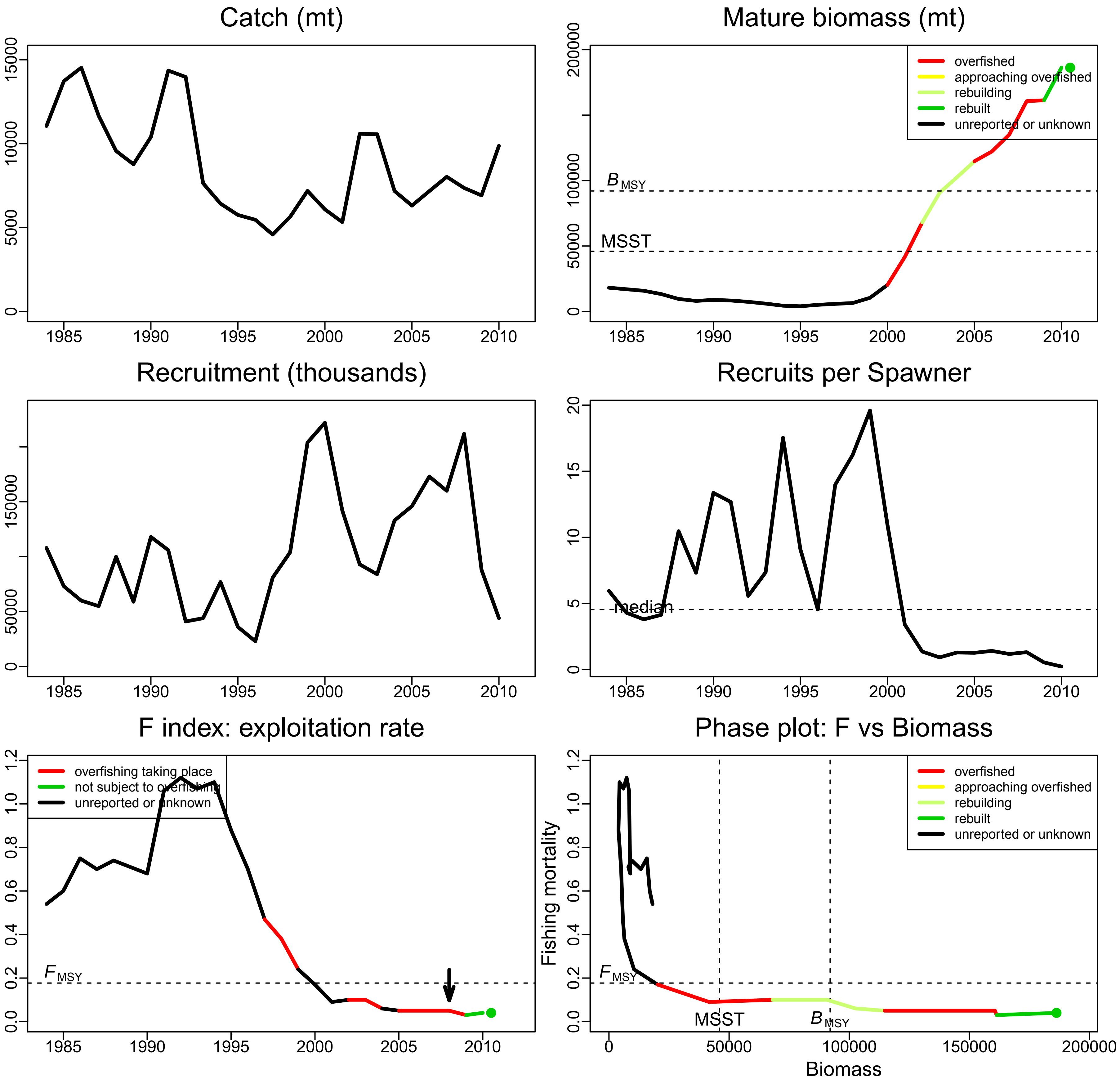
C.41: Red snapper − Southern Atlantic Coast



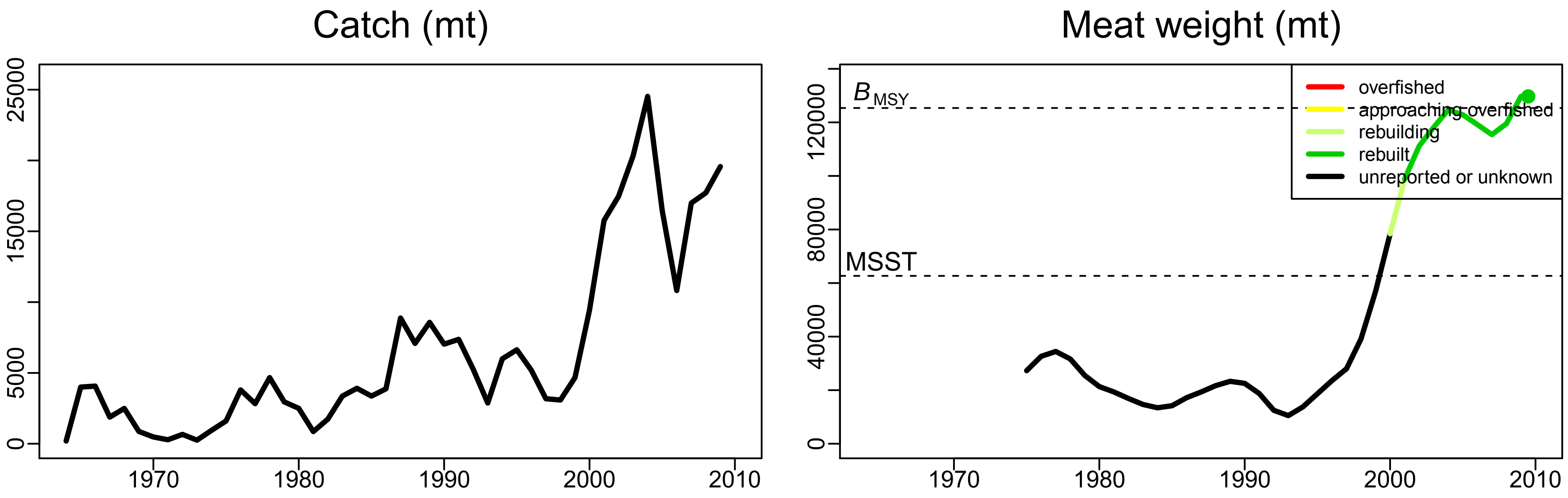
C.42: Sandbar shark − Atlantic

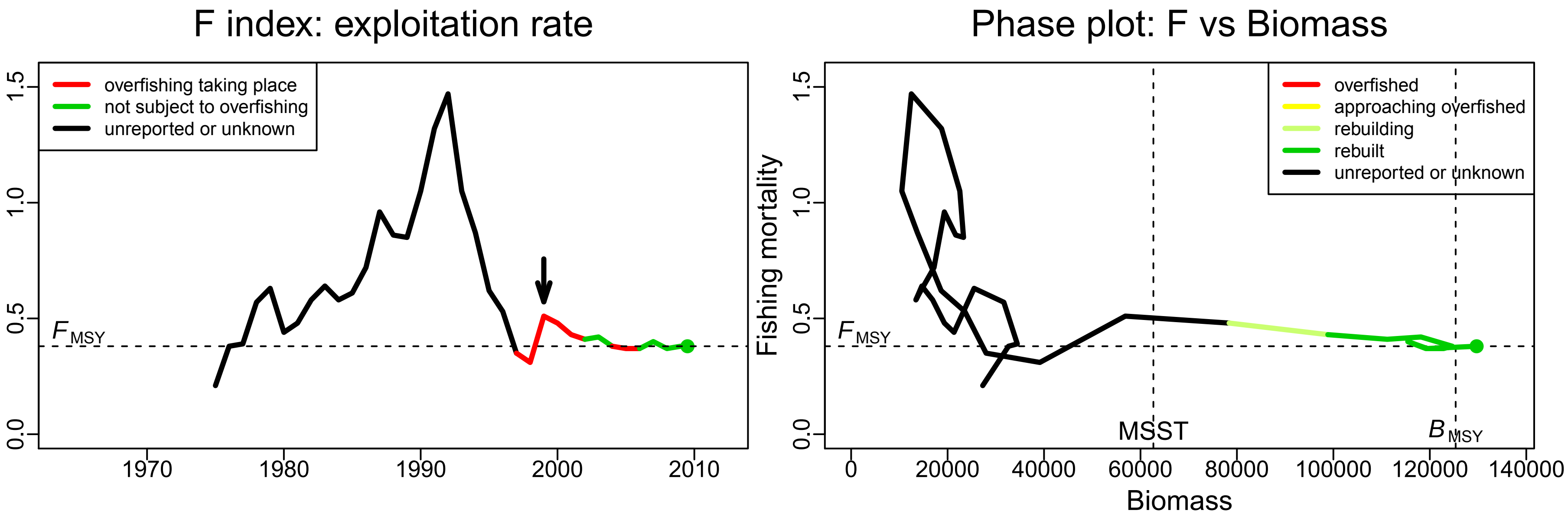


C.43: Scup − Atlantic Coast

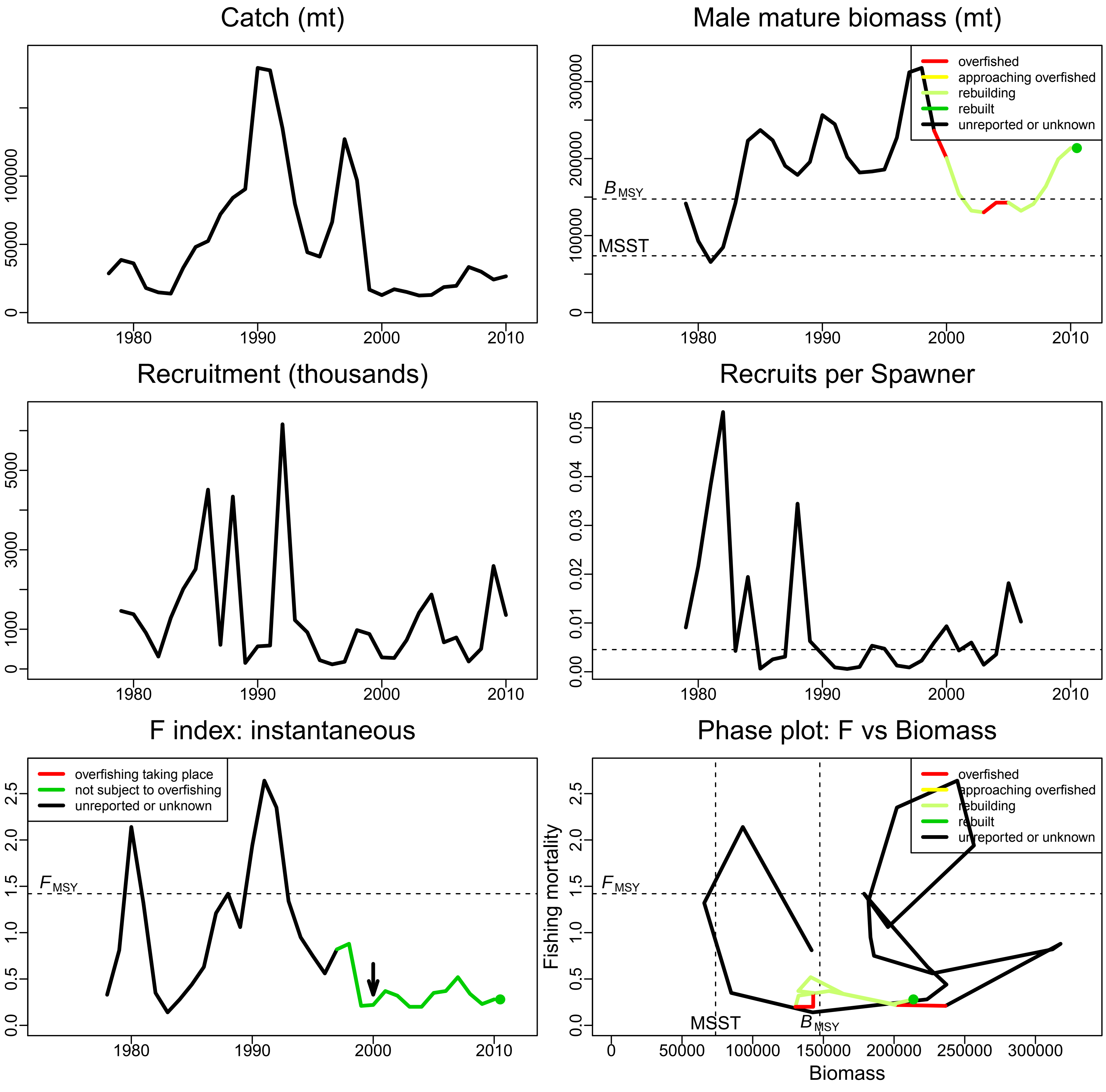


C.44: Sea scallop − Northwestern Atlantic Coast

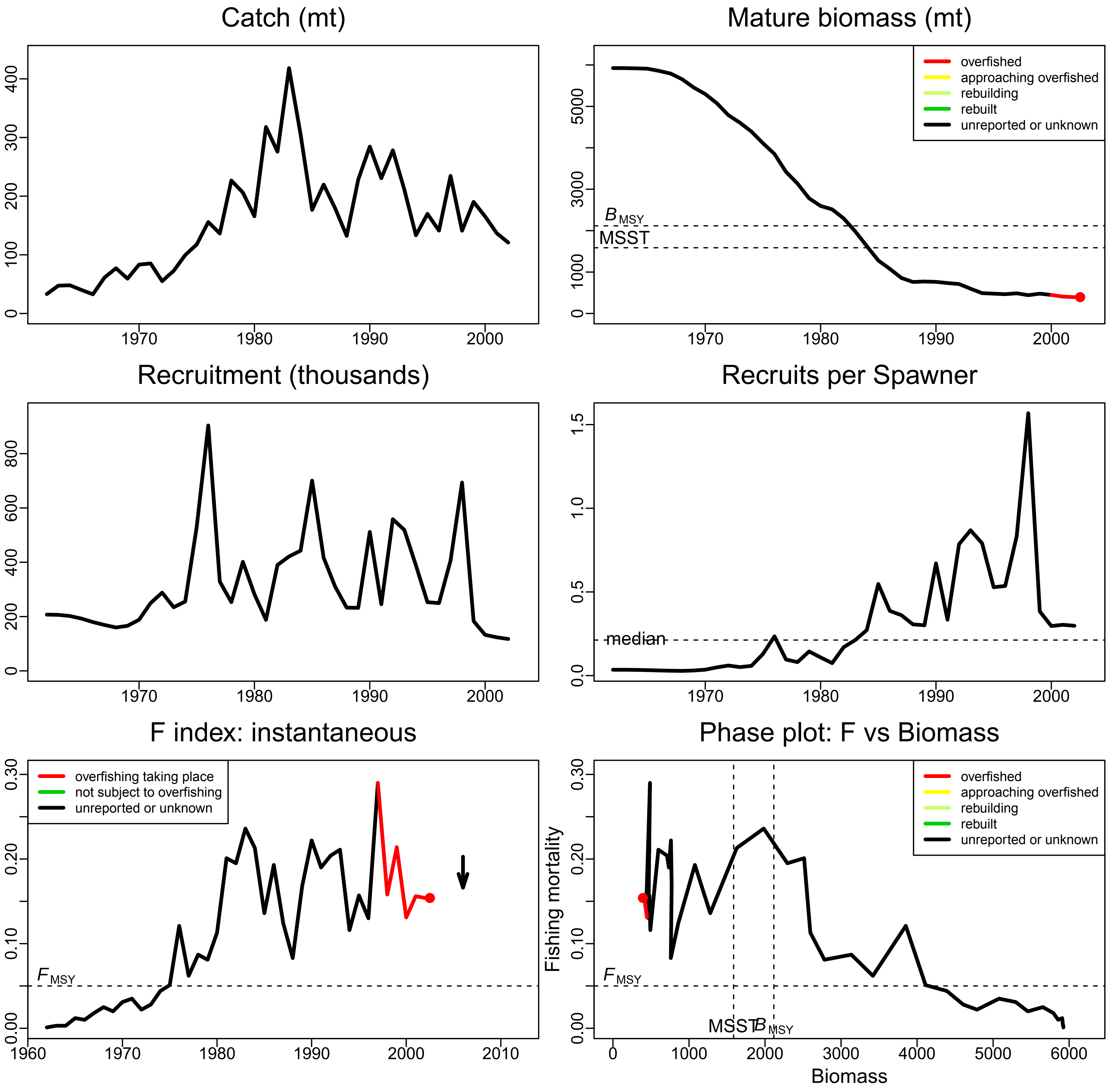




C.45: Snow crab − Bering Sea



C.46: Snowy grouper − Southern Atlantic Coast



Catch (mt)

Mature biomass (mt)

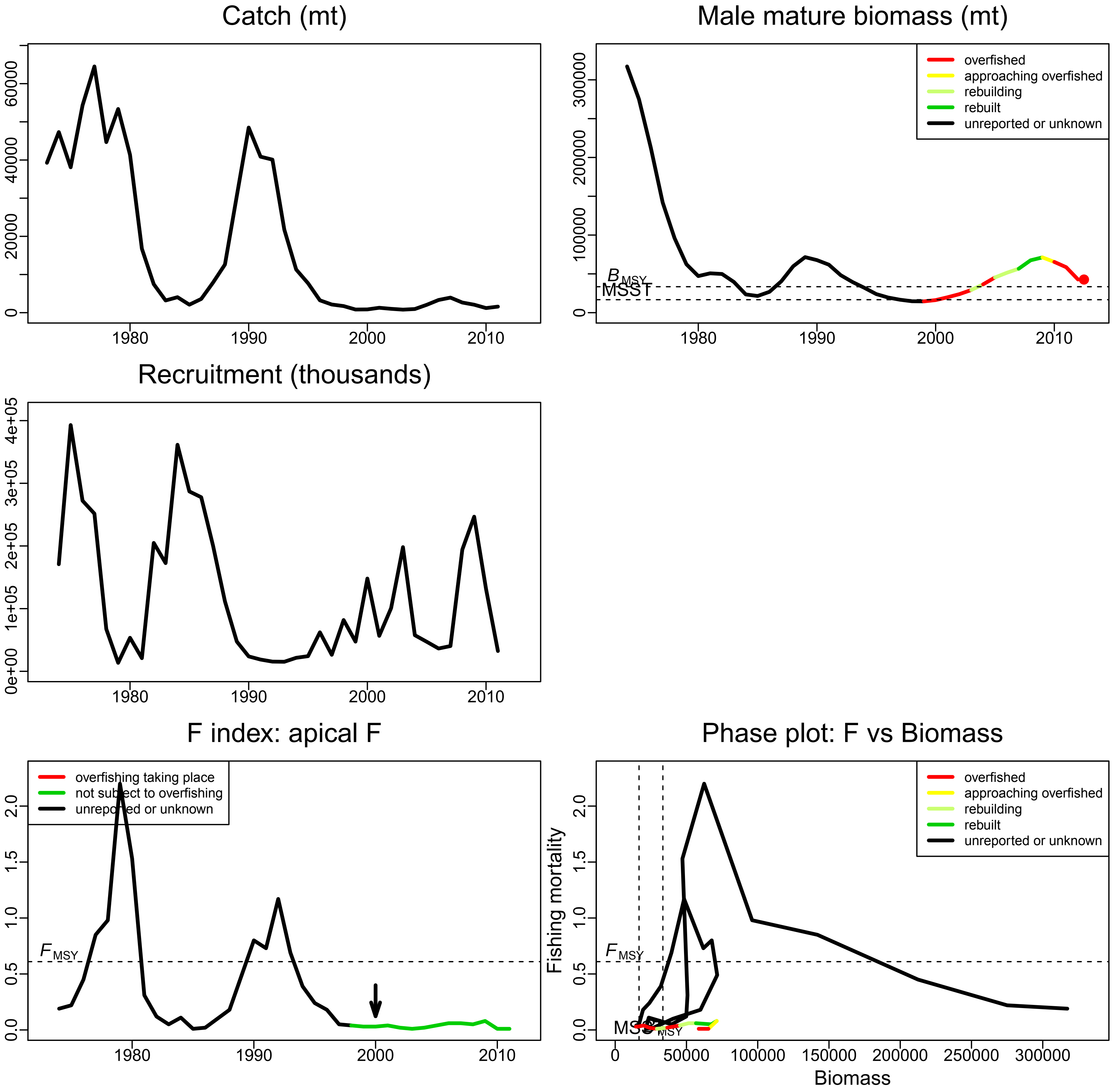
Recruitment (thousands)

Recruits per Spawner

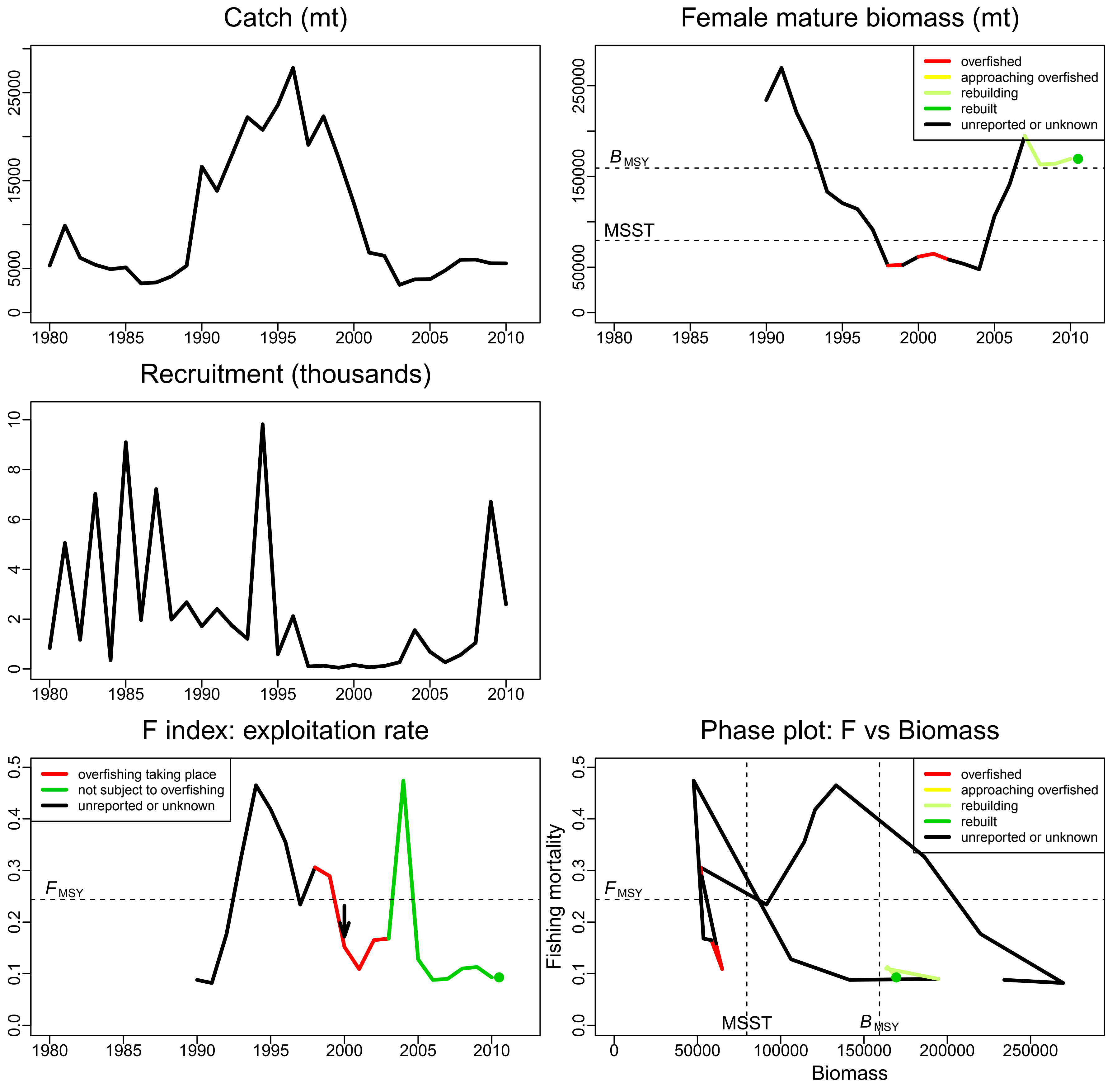
F index: instantaneous

Phase plot: F vs Biomass

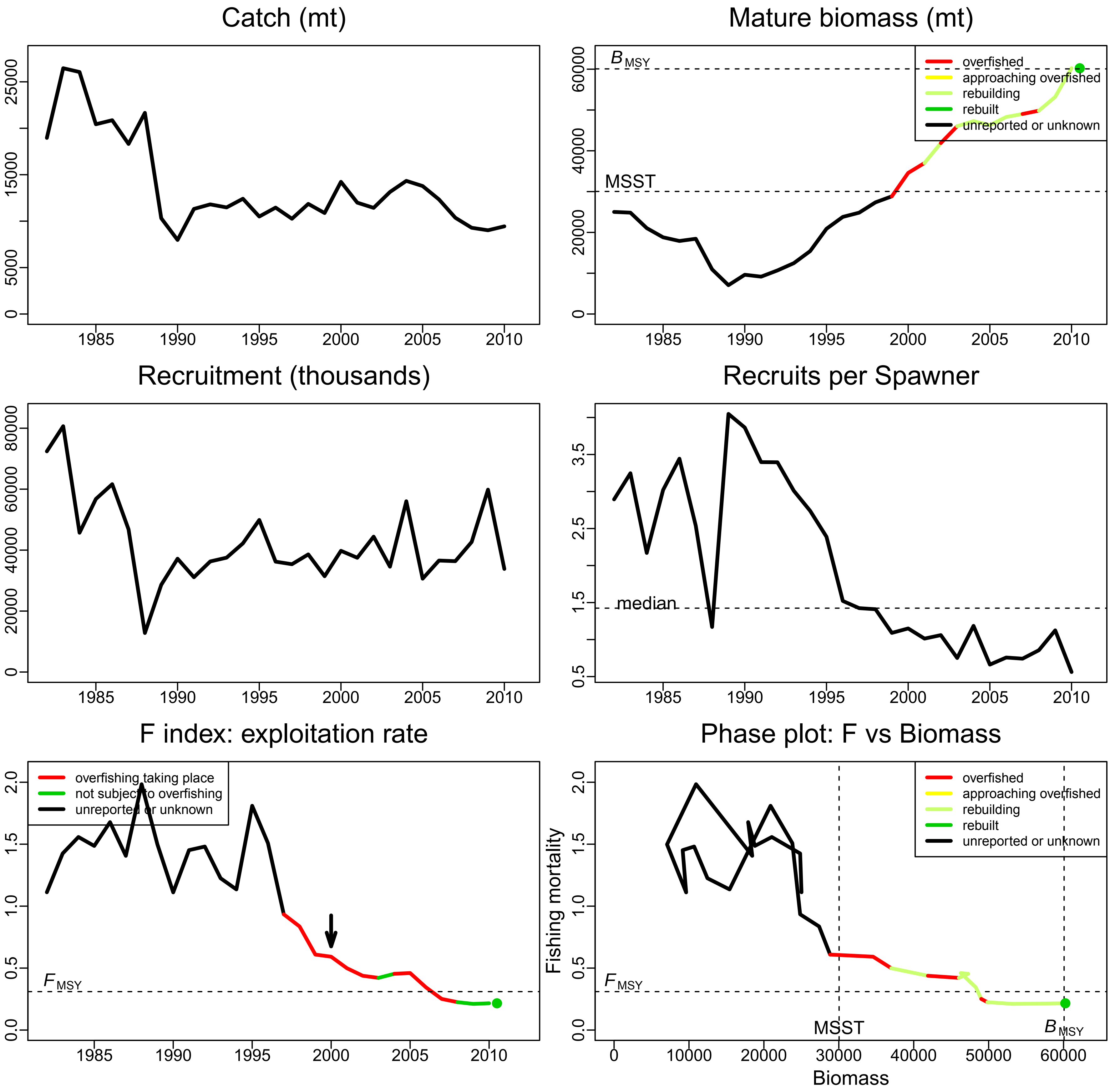
C.47: Southern Tanner crab − Bering Sea



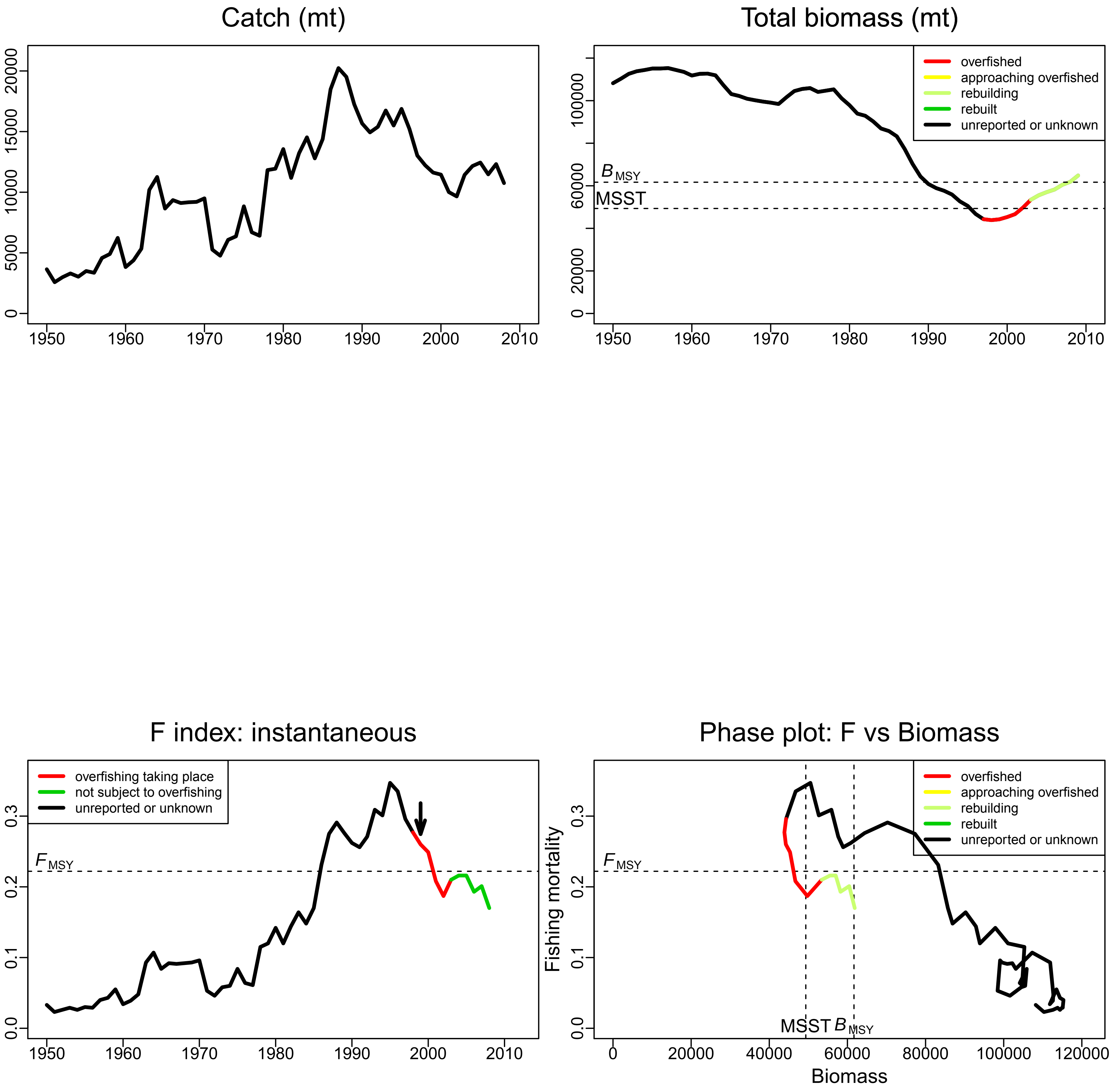
C.48: Spiny dogfish − Atlantic Coast



C.49: Summer flounder − Mid−Atlantic Coast

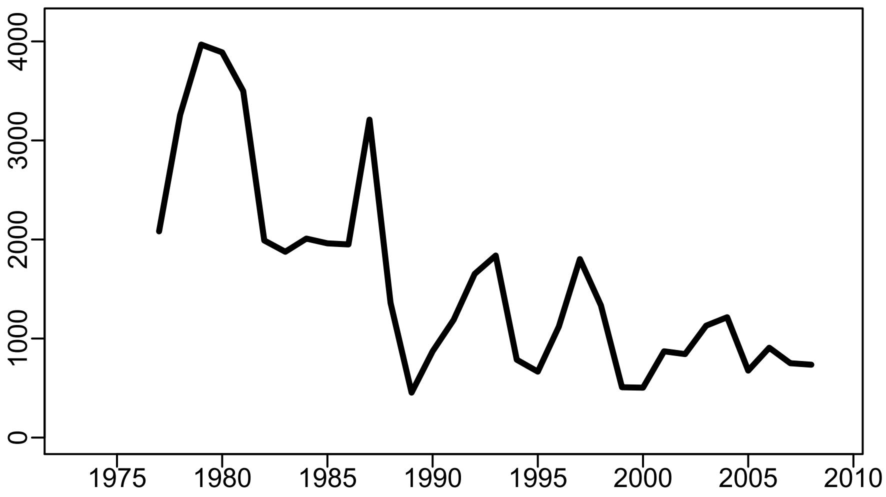


C.50: Swordfish − North Atlantic

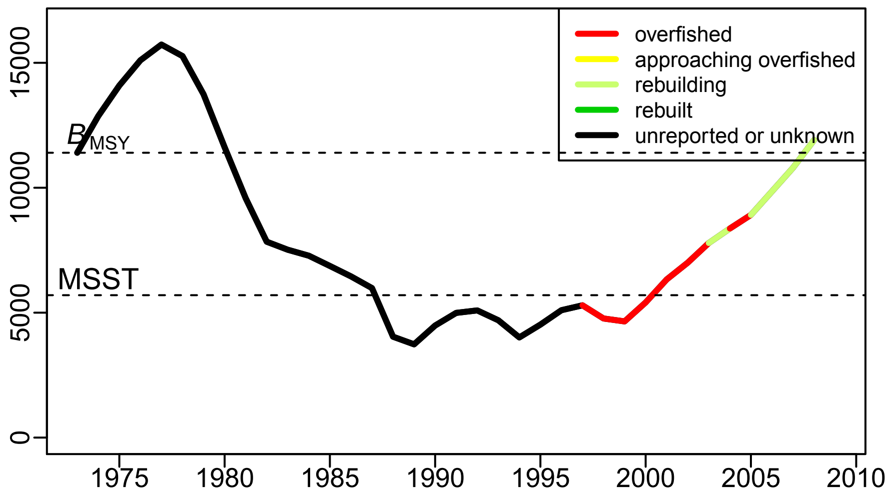


C.51: Tilefish − Mid−Atlantic Coast

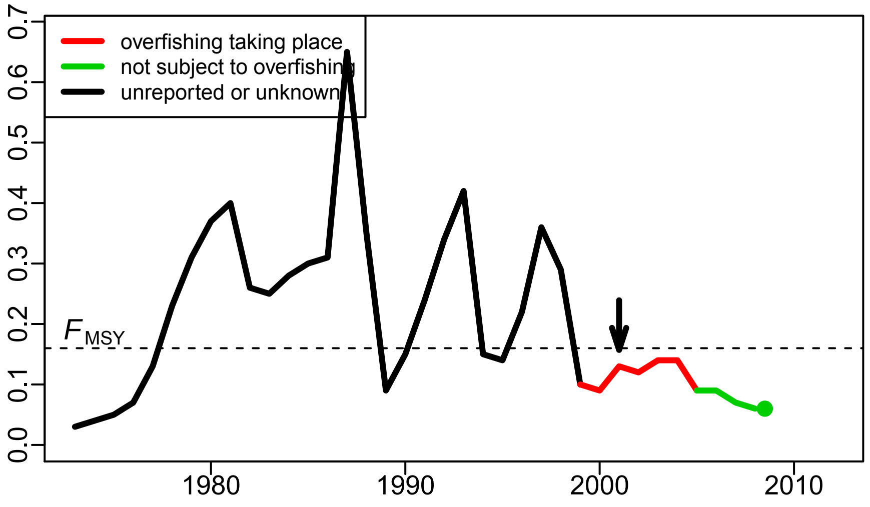
Catch (mt)



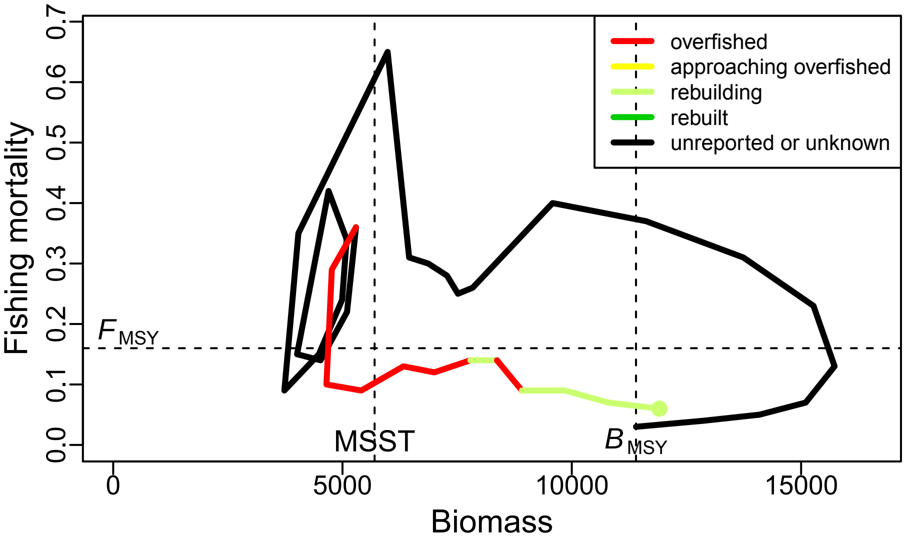
Total biomass (mt)



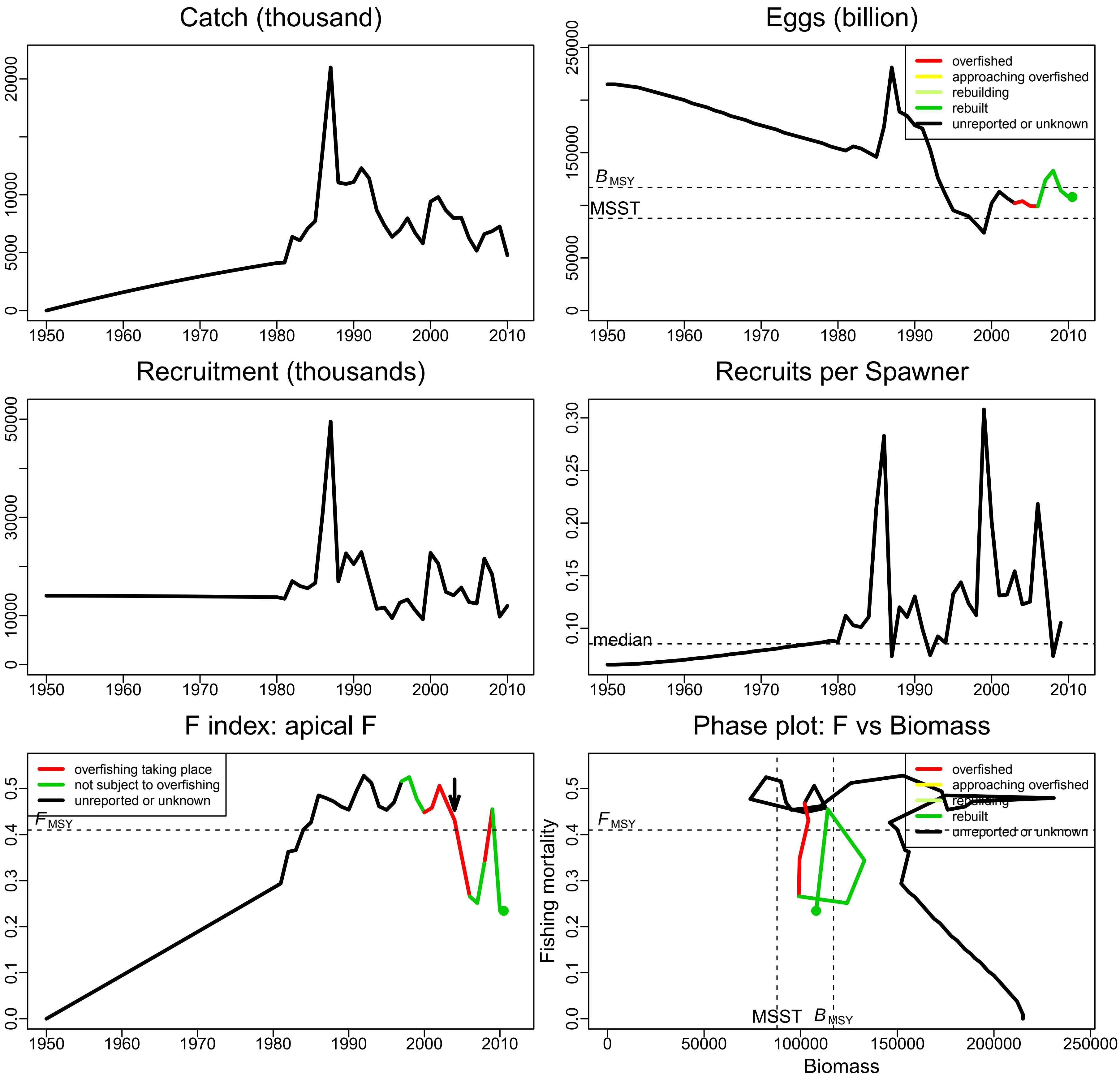
F index: instantaneous



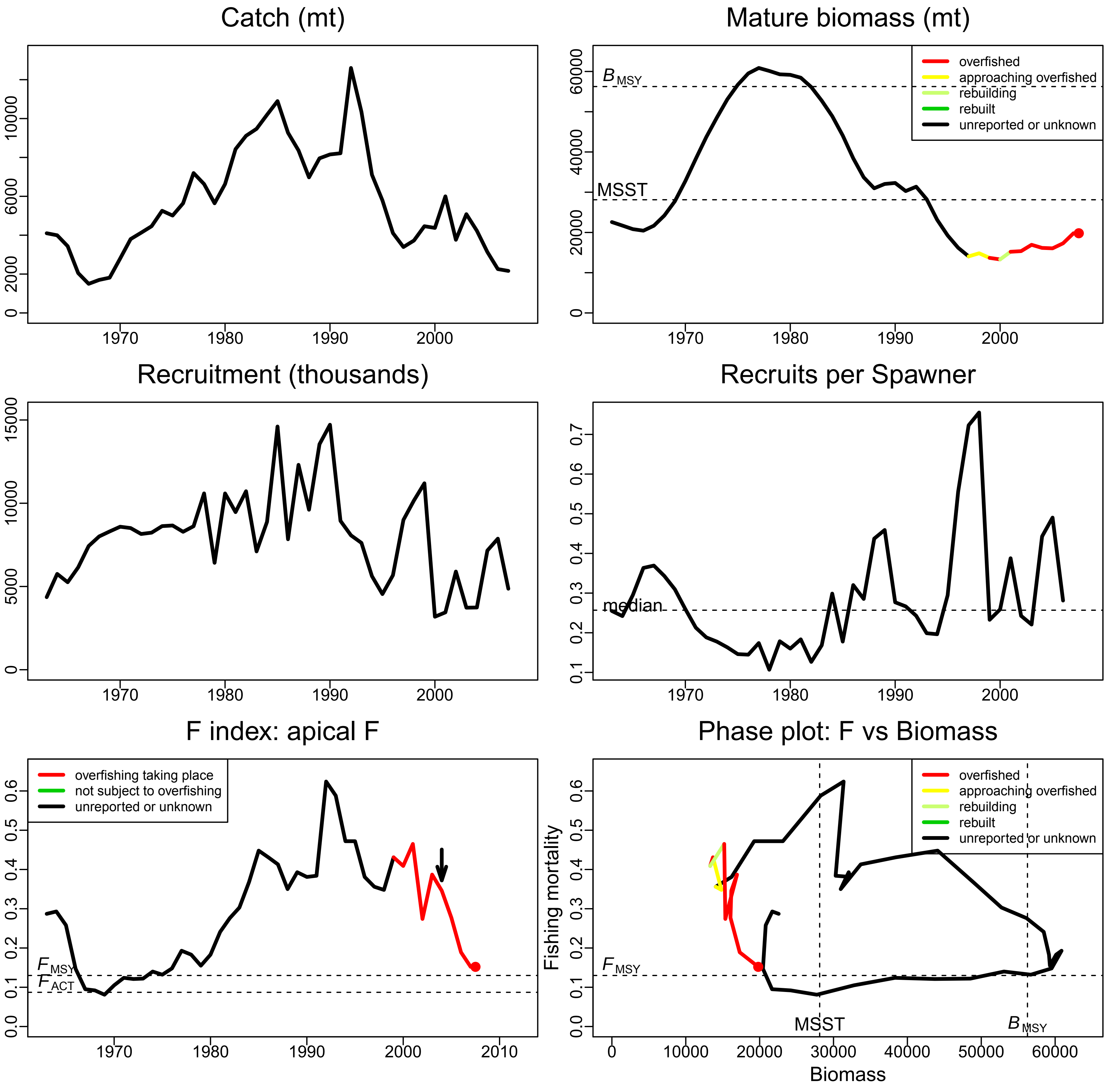
Phase plot: F vs Biomass



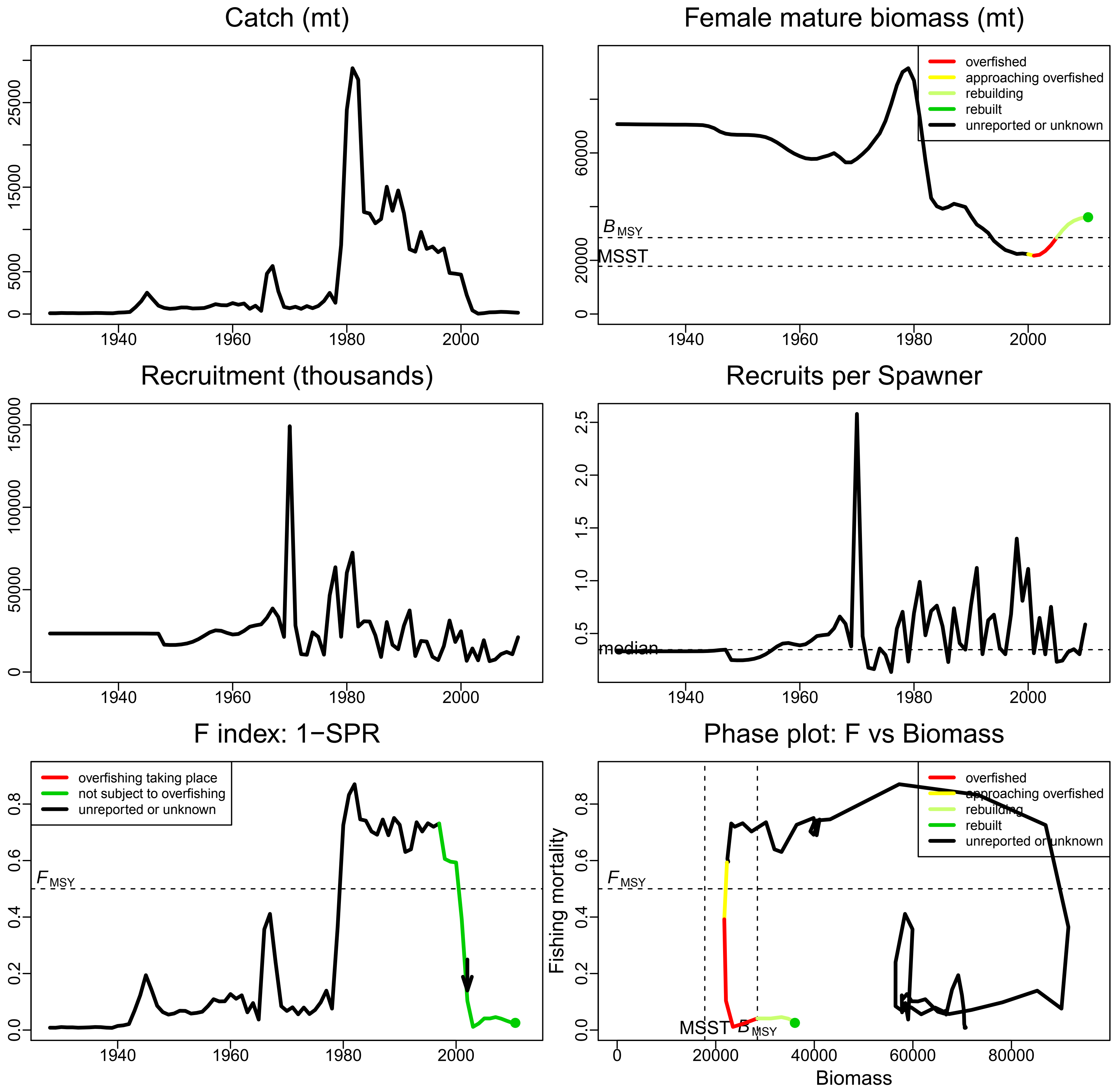
C.52: Vermilion snapper − Gulf of Mexico



C.53: White hake − Gulf of Maine / Georges Bank

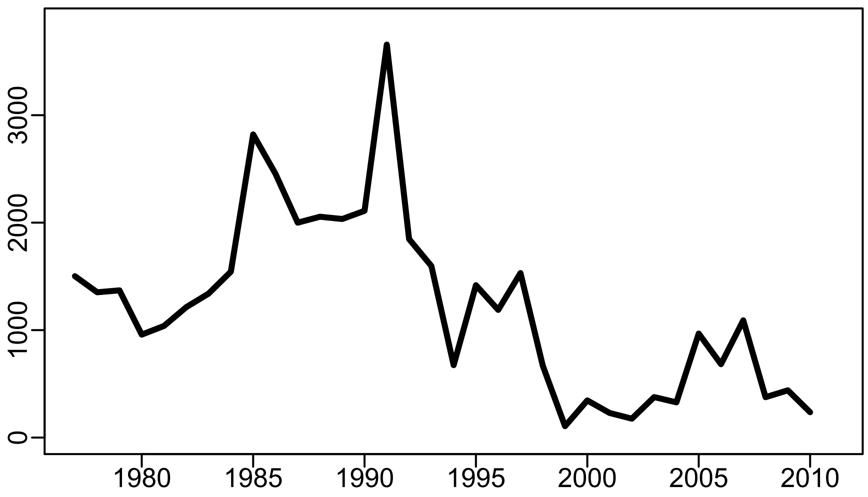


C.54: Widow rockfish − Pacific Coast

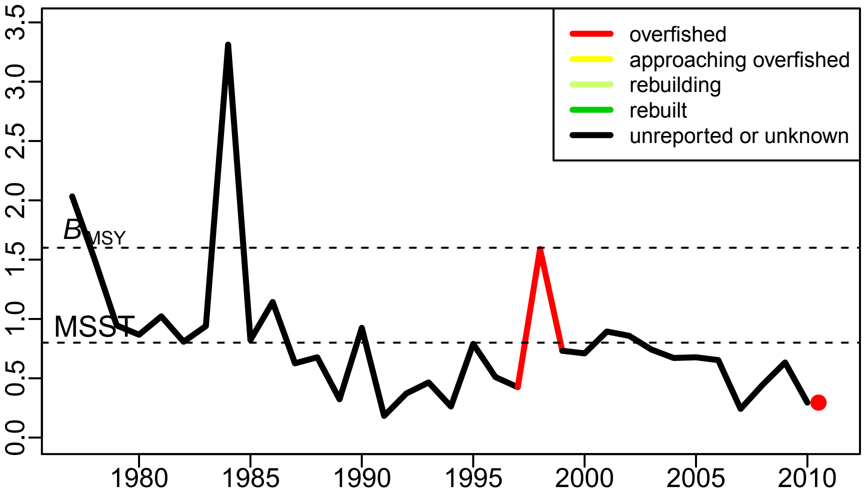


C.55: Windowpane − Gulf of Maine / Georges Bank

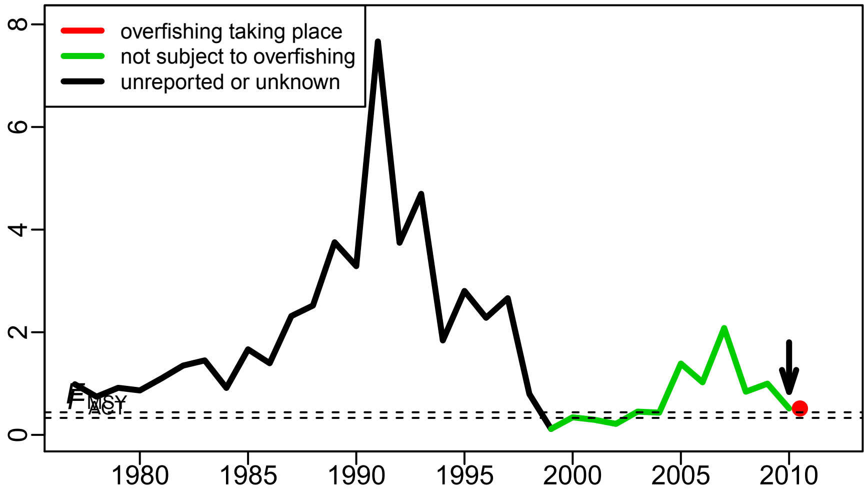
Catch (mt)



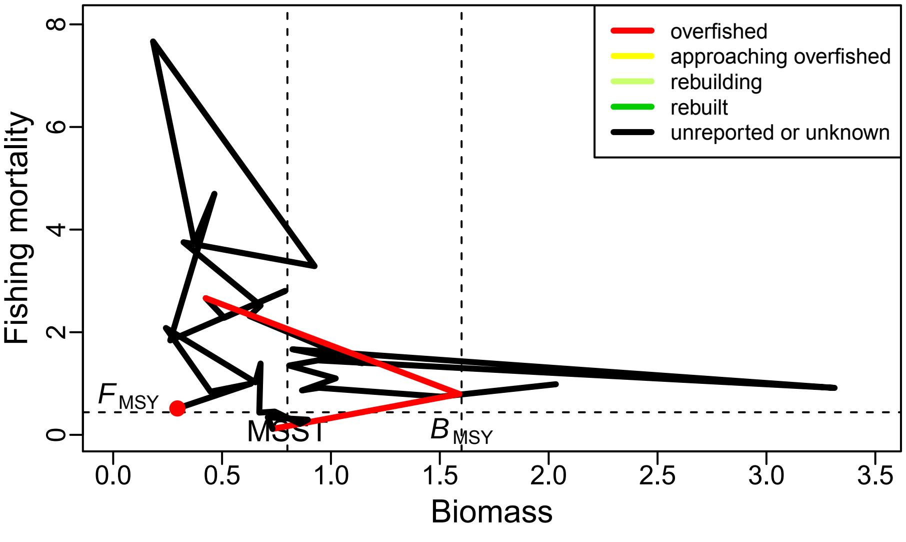
Survey index (kg/tow)



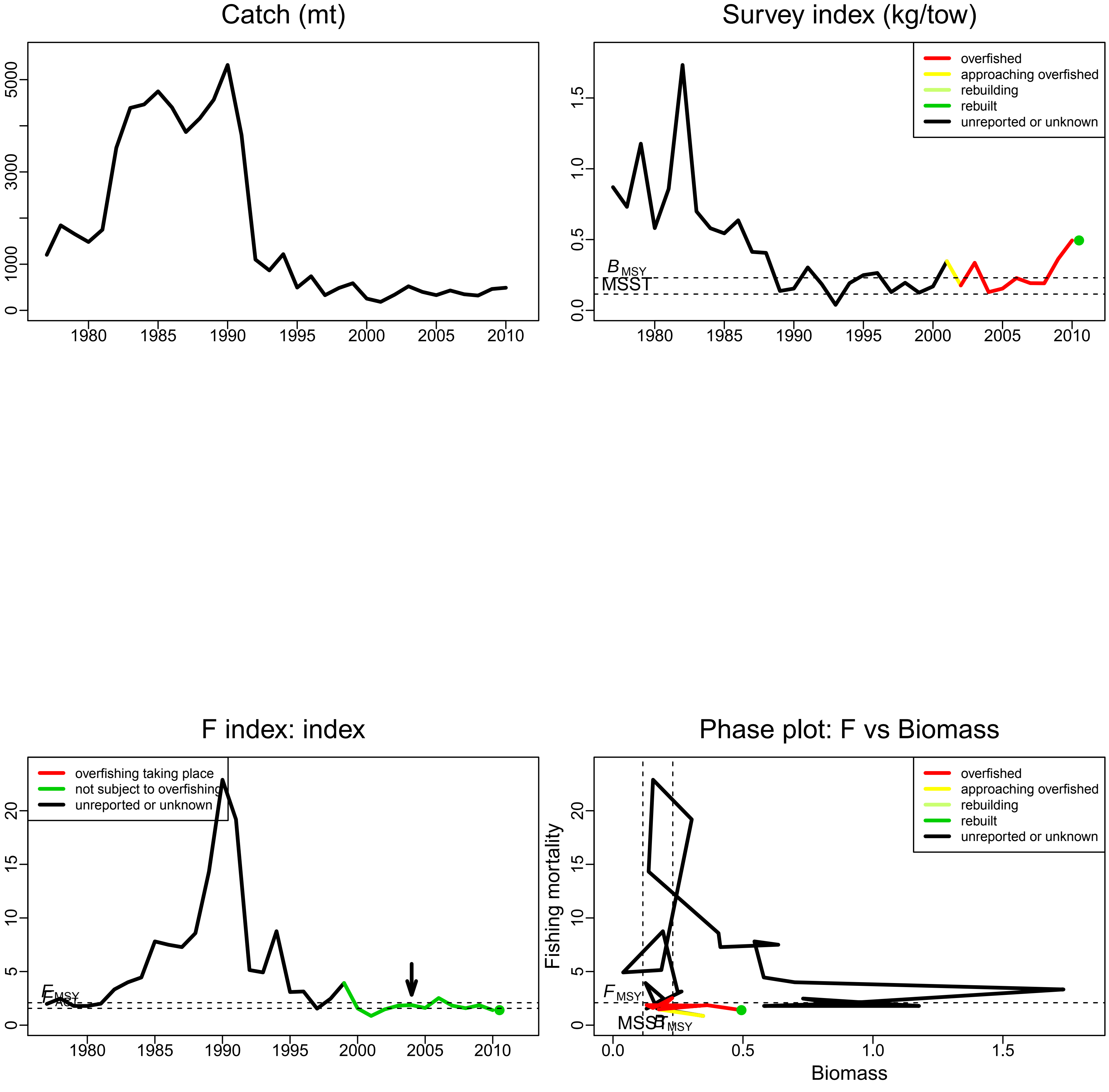
F index: index



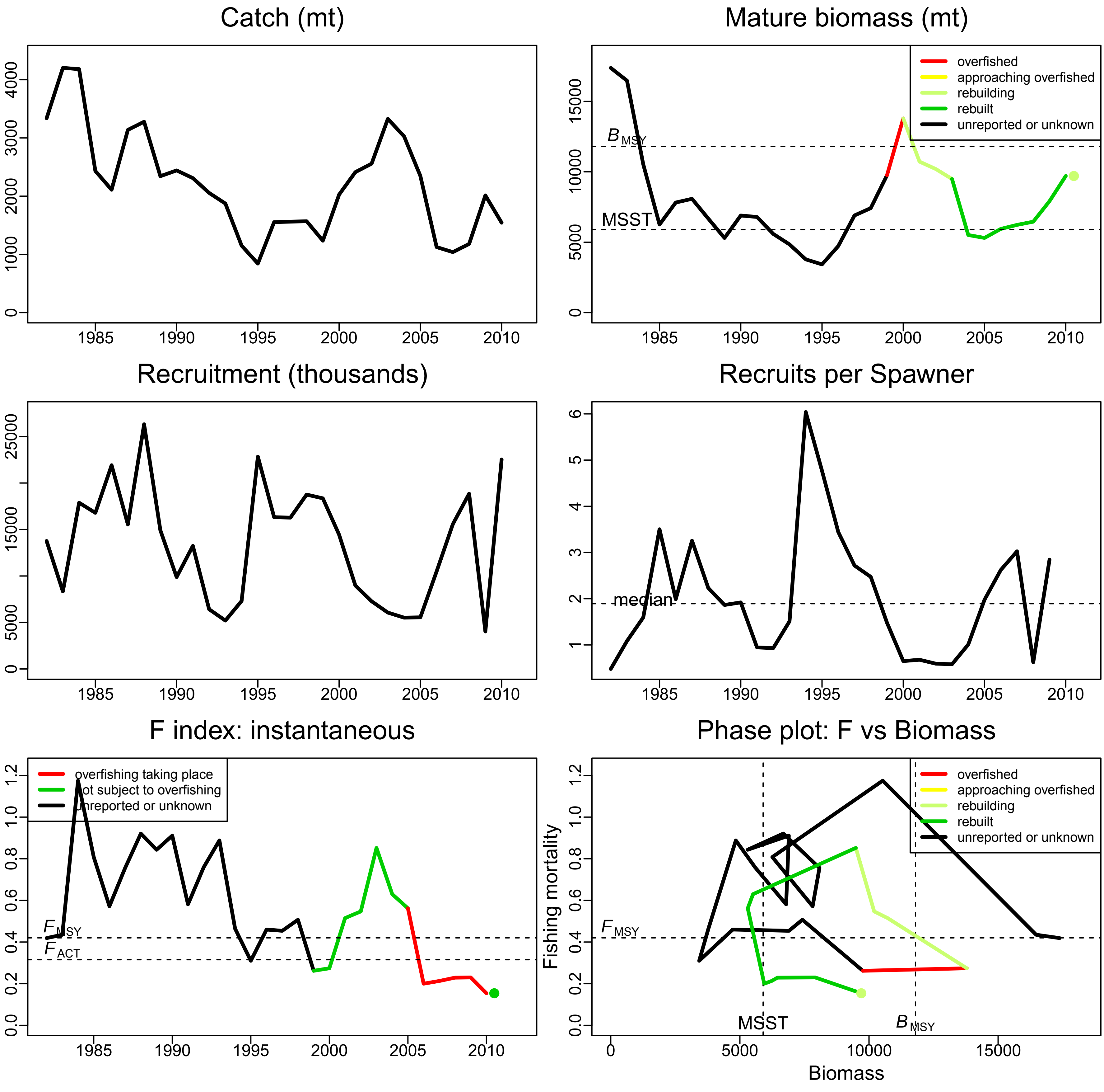
Phase plot: F vs Biomass



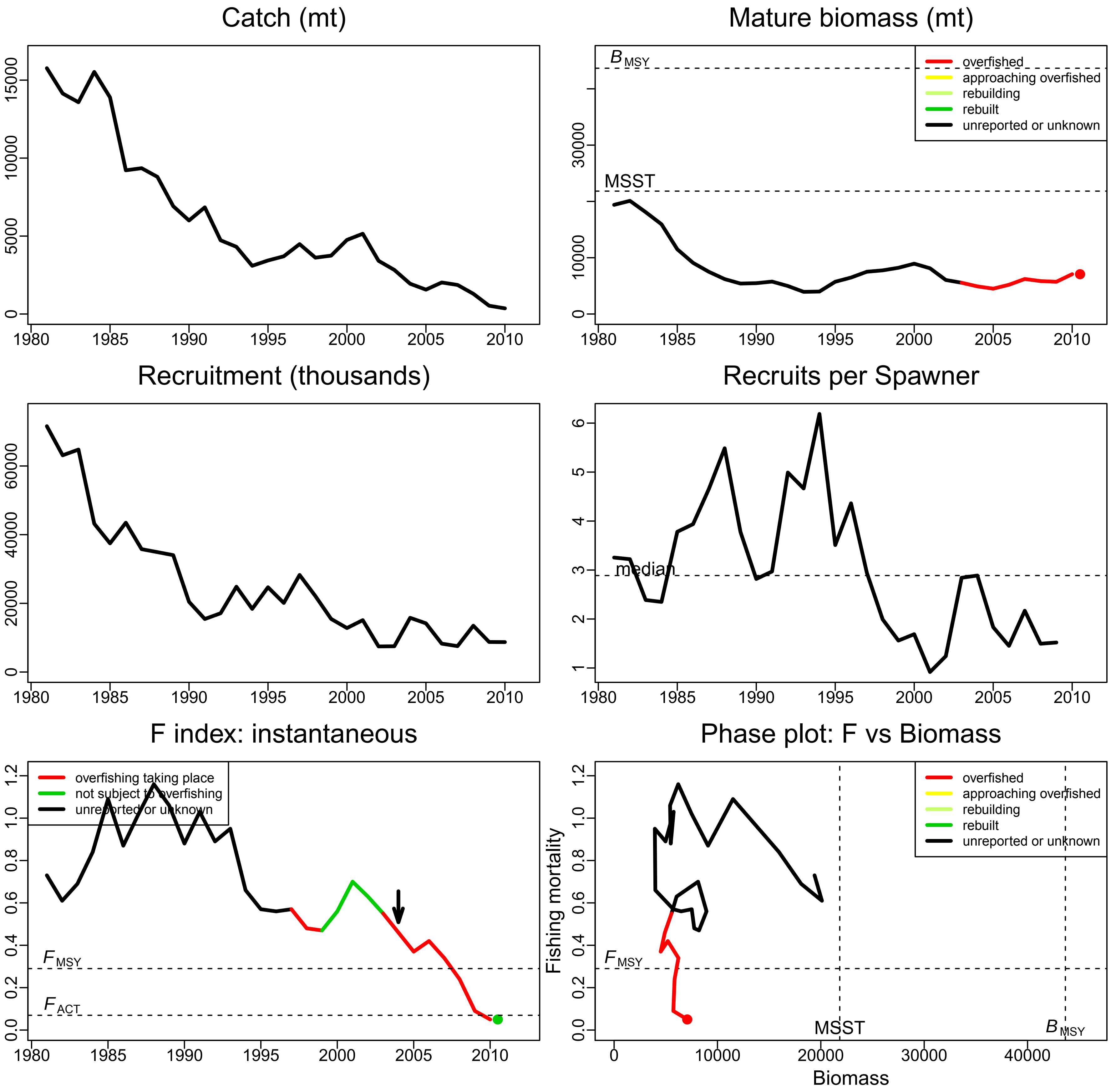
C.56: Windowpane − Southern New England / Mid−Atlantic



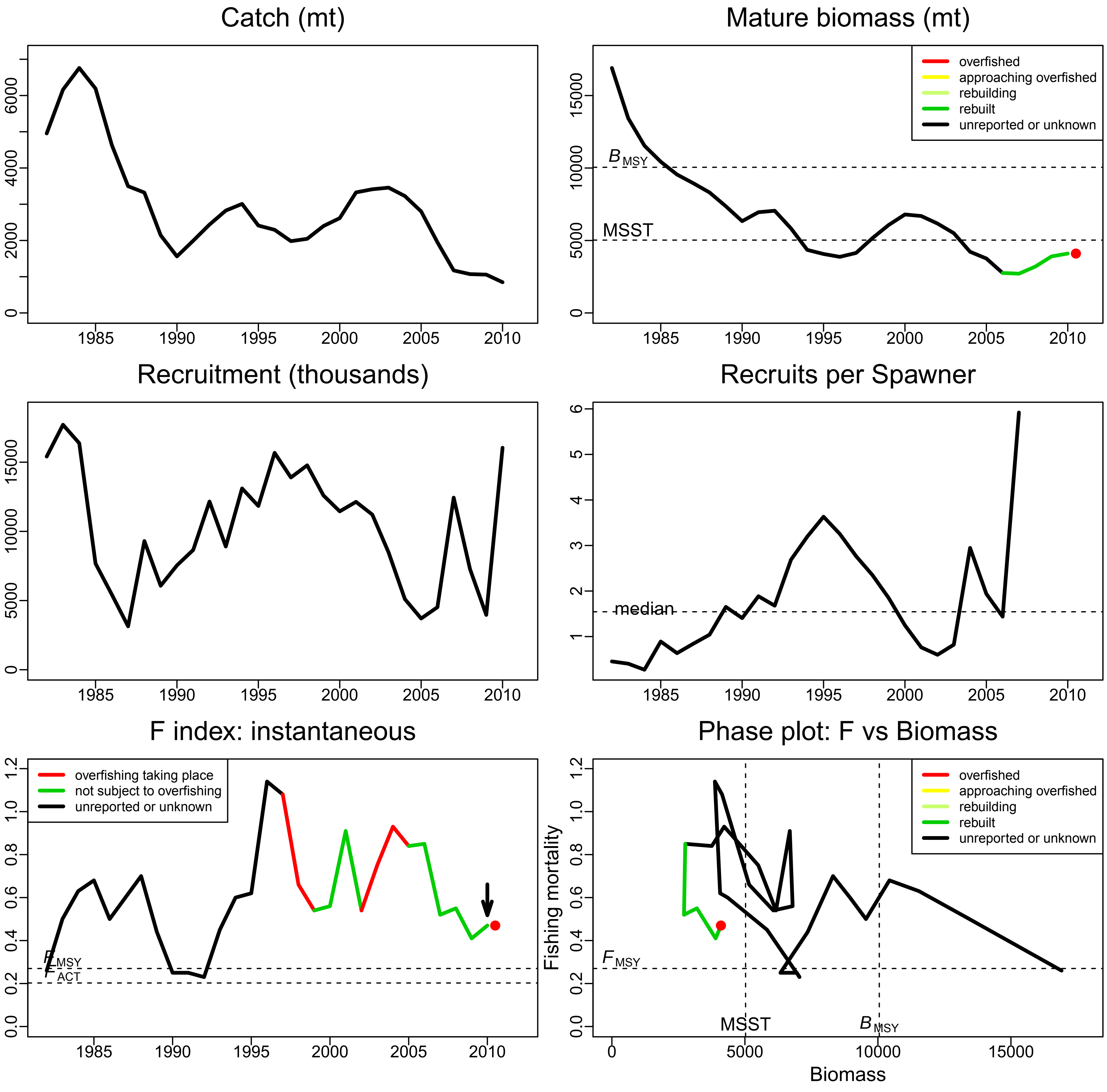
C.57: Winter flounder − Georges Bank



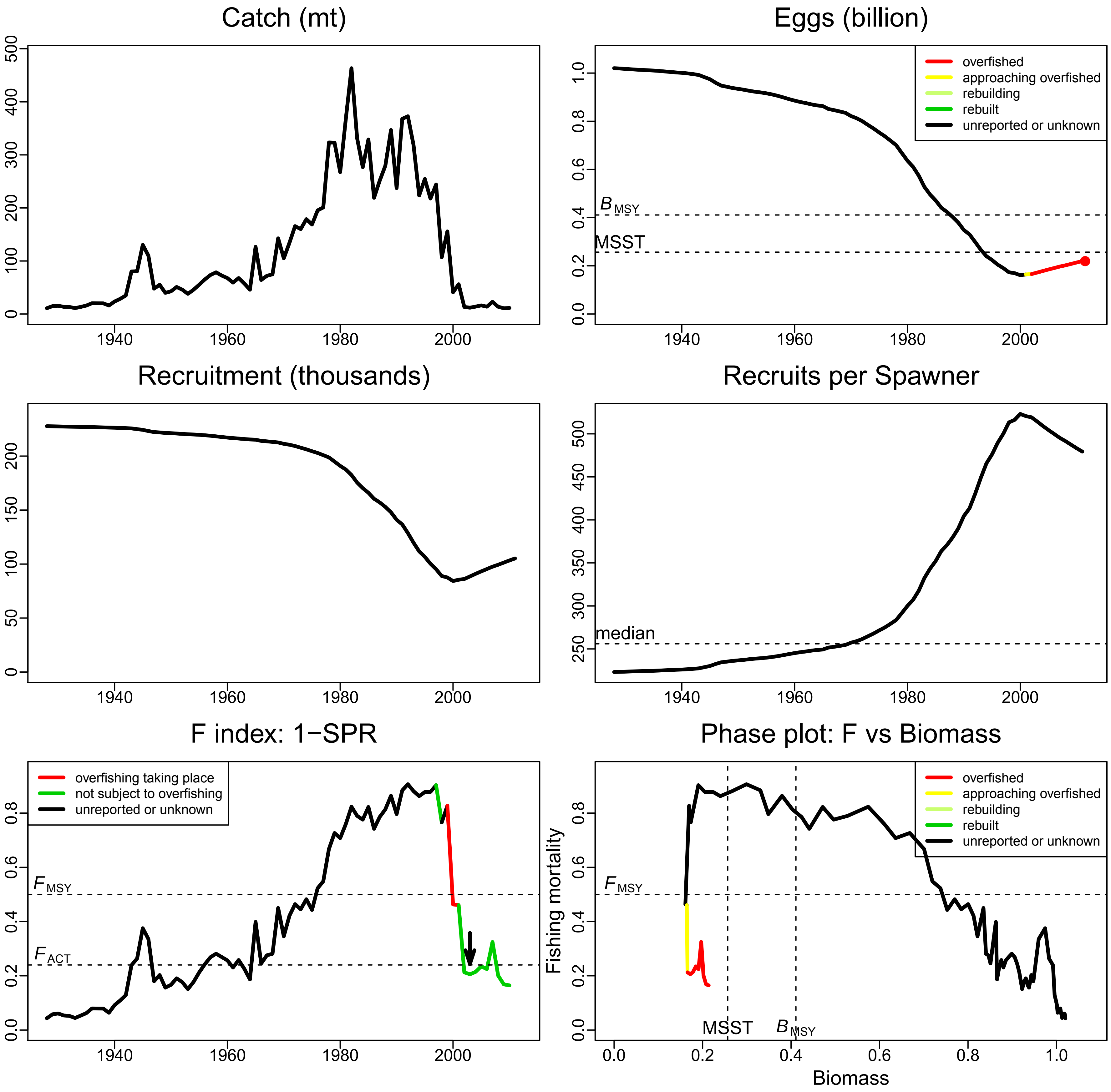
C.58: Winter flounder − Southern New England / Mid−Atlantic



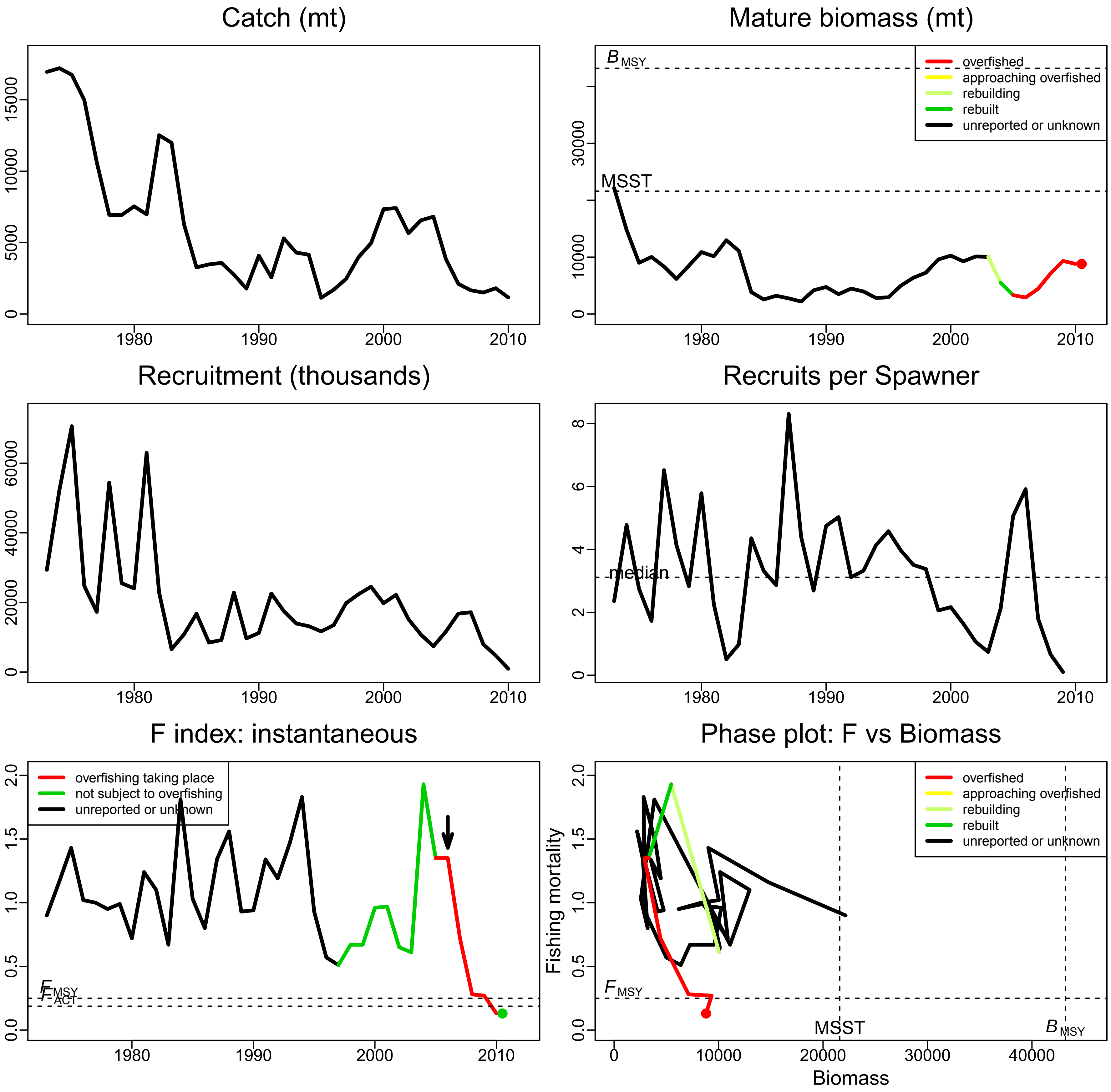
C.59: Witch flounder − Northwestern Atlantic Coast



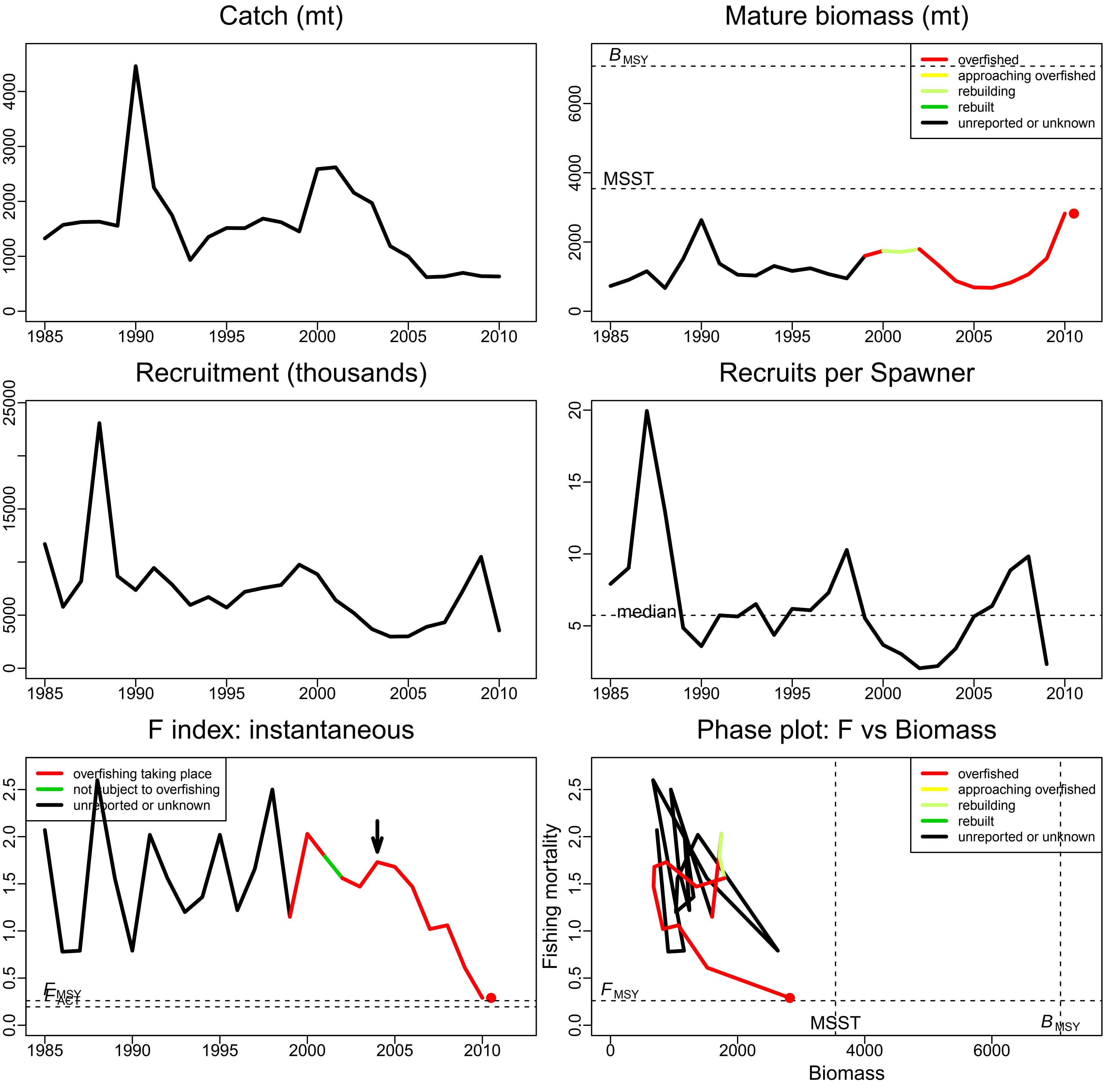
C.60: Yelloweye rockfish − Pacific Coast



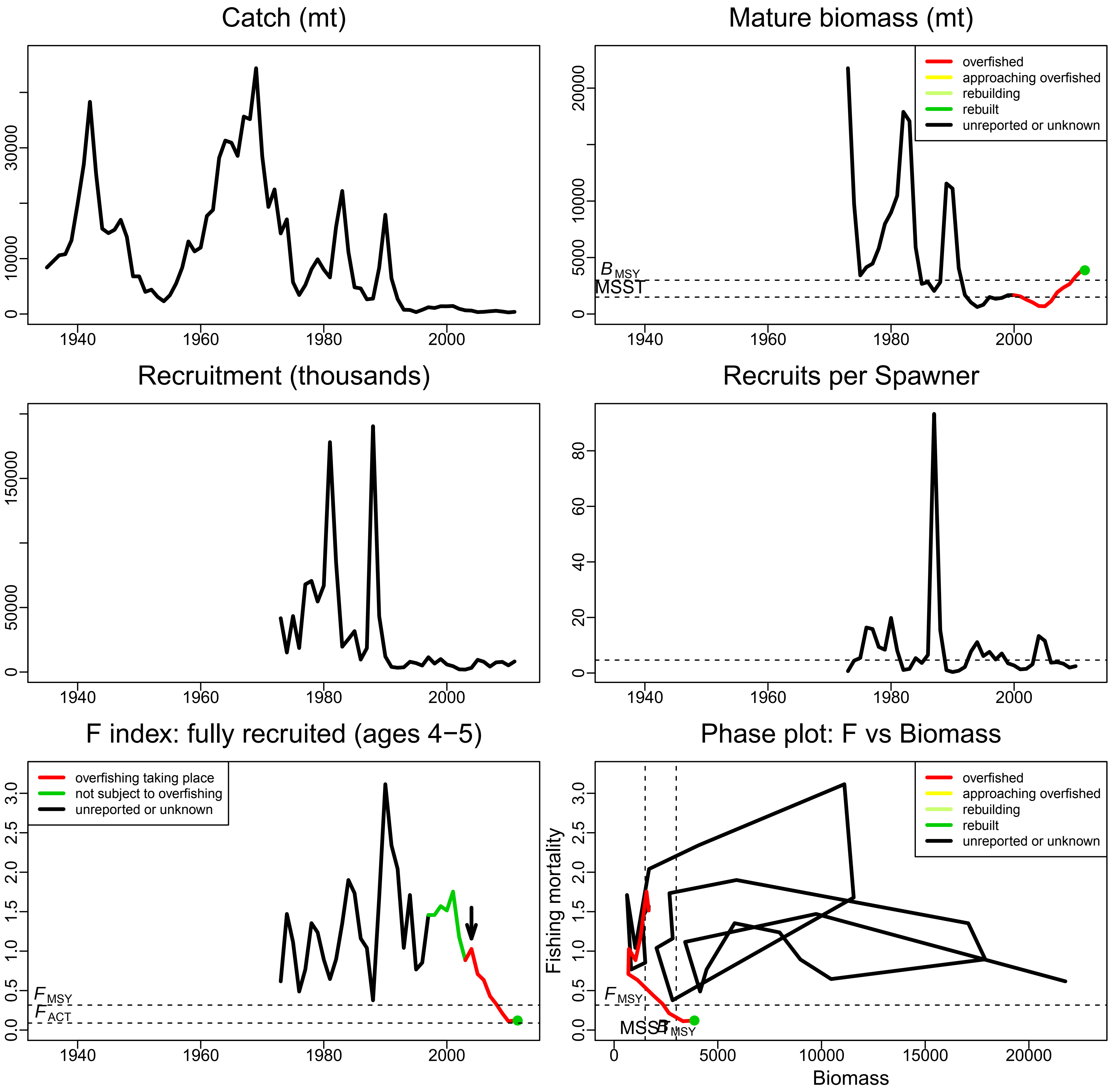
C.61: Yellowtail flounder − Georges Bank



C.62: Yellowtail flounder − Cape Cod / Gulf of Maine



C.63: Yellowtail flounder − Southern New England / Mid−Atlantic



C.64: Yellowtail snapper − Southern Atlantic Coast / Gulf of Mexico