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The Selection and Role of Limit Reference Points for Pacific Herring (*Clupea pallasii*) in British Columbia, Canada

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Foreword

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ABSTRACT

This paper is focussed on the selection of limit reference points for the five major stocks of British Columbia Pacific Herring (*Clupea pallasi*) in partial fulfillment of requirements under the DFO PA Framework and as part of the commitment to renewal of the Pacific Herring management system. The international and Canadian policy basis for "best practice" limit reference points is reviewed in relation to the goal of avoiding "serious harm" to a fish stock. This paper uses an evidence-based approach to evaluate the concept of serious harm and to recommend limit reference points for British Columbia Pacific Herring stocks. Two analytical approaches for diagnosing serious harm are presented. First, production relationships for the five major stocks of Pacific Herring are inspected for persistent periods of low production and low biomass consistent with signs of possible serious harm. Second, theoretical equilibrium reference fishing mortality rates based on the concept of the replacement fishing mortality are investigated, as well as associated proxies based on maximum sustainable yield, spawning potential ratio and yield-per-recruit.

Pacific Herring stocks in the Central Coast, Haida Gwaii, and West Coast Vancouver Island management areas showed recent evidence of persistent low production, low biomass states that began by the mid-2000s and persisted for six to eleven years depending on the stock. These states were preceded by a transition to low production that began as early as the late 1990s from levels of comparatively high spawning biomass. The low spawning stock depletion levels reached during these periods was comparable to the levels estimated during the collapse of all five major stocks in the late 1960s, which was attributed to overharvest rather than loss of production.

The results of this study suggest that a persistent low production and low biomass state can occur at levels below 0.3 of the estimated equilibrium unfished spawning stock biomass for the Central Coast, Haida Gwaii and West Coast Vancouver Island stocks. A limit reference point of 0.3 of the unfished equilibrium spawning biomass is recommended for these stocks. The same limit reference point is recommended for the Prince Rupert District and Strait of Georgia stocks based on common life history and geographic proximity to the other three major stocks. Limit equilibrium fishing mortality rates based on the concept of replacement fishing mortality could not be recommended due to implausible estimates that were attributed in part to non-stationary conditions for natural mortality and size-at-age. It is also recommended that the introduction of limit reference points for Pacific Herring in British Columbia should adopt a management-oriented simulation approach to evaluating the consequences of reference point choices when evaluating alternative management options. It is recommended that management procedures designed to avoid breaching limit reference points and achieving desired targets be phased-in to smooth the transition from existing operational practice.

La sélection et le rôle des points de référence limites pour le hareng du Pacifique (Clupea pallasii) en Colombie-Britannique, Canada

RÉSUMÉ

Ce document est axé sur la sélection des points de référence limites pour les cinq stocks principaux de harengs du Pacifique en Colombie-Britannique (*Clupea pallasi*), afin de répondre partiellement aux besoins du cadre de l'approche de précaution du MPO, et de respecter l'engagement de renouveler le système de gestion du hareng du Pacifique. Le fondement stratégique canadien et international des pratiques exemplaires en matière de points de référence limites fait l'objet d'un examen en fonction de l'objectif d'éviter les « dommages sérieux » à un stock de poissons. Le document utilise une approche fondée sur des données probantes pour évaluer la notion de dommages sérieux et pour recommander des points de références limites pour les stocks de harengs du Pacifique de la Colombie-Britannique. Deux approches analytiques visant à déceler les dommages sérieux sont présentées. D'abord, les liens de production des cinq principaux stocks de harengs du Pacifique sont examinés pour déceler des périodes persistantes de fiable production et de faible biomasse correspondant à des signes de dommages sérieux possibles. Ensuite, on étudie les taux de mortalité théoriques par pêche, tirés des points de référence d'équilibre fondés sur la notion de mortalité par pêche du remplacement, ainsi que les approximations connexes fondées sur le rendement maximal durable, le ratio du potentiel de frai et le rendement par recrue.

Les stocks de harengs du Pacifique des zones de gestion de la côte centrale, de Haida Gwaii et de la côte ouest de l'île de Vancouver ont montré des signes récents de faible production constante et de faible biomasse, qui ont commencé au milieu des années 2000 et qui ont duré de six à onze ans, selon le stock. Ces états étaient précédés par une transition qui a commencé dès la fin des années 1990, où des niveaux relativement élevés de biomasse du stock reproducteur sont passés à une production faible. Les niveaux d'épuisement des stocks reproducteurs atteints pendant ces périodes étaient comparables aux niveaux estimés pendant l'effondrement des cinq stocks principaux à la fin des années 1960, lequel était plus attribuable à la surpêche qu'à la perte de production.

Les résultats de cette étude suggèrent que des états persistants de production et de biomasse faibles peuvent se produire à des niveaux inférieurs à 0,3 de la biomasse d'équilibre non exploitée estimée du stock reproducteur pour les stocks de la côte centrale, de Haida Gwaii et de la côte ouest de l'île de Vancouver. Un point de référence limite de 0,3 de la biomasse d'équilibre non exploitée du stock reproducteur est recommandé pour ces stocks. Le même point de référence limite est recommandé pour les stocks du district de Prince Rupert et du détroit de Georgie, en fonction du cycle biologique commun et de la proximité géographique des trois autres stocks principaux. Des taux limites de mortalité par pêche à l'équilibre fondés sur le concept de la mortalité par pêche pour un remplacement n'ont pas pu être recommandés en raison des estimations peu vraisemblables qui ont été attribuées en partie aux conditions non stationnaires de la mortalité naturelle et de l'âge selon la taille. On recommande également que l'introduction de points de référence limites pour le hareng du Pacifique en Colombie-Britannique adopte une approche de simulation axée sur la gestion pour évaluer les conséquences découlant des choix de points de référence pendant l'évaluation des options de gestion alternatives. On recommande aussi l'intégration progressive des procédures de gestion conçues pour éviter de dépasser les points de référence limites et pour atteindre les cibles souhaitées, afin de faciliter la transition des pratiques opérationnelles actuelles.

1 INTRODUCTION

1.1 BACKGROUND

Biological reference points (BRPs) are commonly used to evaluate the status of fished populations in most management jurisdictions. Reference points are generally categorized as limits and targets. A limit reference point (LRP) can be a minimum level of spawning biomass that should not be breached, or a fishing mortality rate that should not be exceeded, in order to avoid deleterious outcomes for the stock and fisheries. Target reference points (TRPs) indicate desirable states expected to maximize ecosystem, socio-cultural and economic benefits.

The Fisheries and Oceans Canada (DFO) "*Harvest Decision-Making Framework Incorporating the Precautionary Approach*" policy (DFO 2009a), hereafter called the DFO PA Framework, is similar to other fisheries policies that advocate management by reference points (e.g., Restrepo 1998; Sainsbury 2008). In particular, the DFO PA Framework requires the following elements:

PA1. Limit and upper stock status reference points that delineate Critical, Cautious and Healthy zones and a limit fishing mortality rate;

PA2. A harvest strategy and harvest control rules (HCRs, Kronlund et al. 2014a);

PA3. The need to take into account uncertainty and risk when developing reference points and developing and implementing control rules (Kronlund et al. 2014a,b); and

PA4. A requirement to evaluate the performance of the management system against the objectives specified by the harvest strategy (Kronlund et al. 2014b).

Prevailing practice for management by reference points in Canada and elsewhere is to first estimate the historical and current fish population size using some type of stock assessment model. A biological yield optimization approach is typically applied to generate estimates of unfished equilibrium stock size (B_0), or measures of productivity such as maximum sustainable yield (MSY), biomass at MSY (B_{MSY}), or a maximum sustainable fishing mortality (F_{MSY}). Some multiples of these parameters are then chosen for use as BRPs (AFMA 2007; Hilborn 2002; Ministry of Fisheries 2011; Shelton and Rice 2002; UNFSA 1995). For example, the DFO PA Framework cites a provisional LRP of $0.4B_{\mathrm{MSY}}$. Suitable proxies may be substituted for MSY parameters based on yield-per-recruit (Beverton and Holt 1957) or spawner-per-recruit (Clark 1991, 1993, 2002; Gabriel et al. 1989; Mace and Sissenwine 1993). Finally, a recommended catch is calculated for the fishery based on forecast biomass; additional management tactics intended to support achieving that catch may also be applied.

Assessment and management of the five major stocks of Pacific Herring (*Clupea pallasii*) in British Columbia (BC) has followed this practice to a great extent. The major stocks of Pacific Herring in BC are assessed and managed in the Central Coast (CC), Haida Gwaii (HG), Prince Rupert District (PRD), Strait of Georgia (SOG) and West Coast Vancouver Island (WCVI) management areas (DFO 2014; DFO 2016a). There is a long heritage of quantitative stock assessment and management that incorporates comprehensive data collection for stock and fishery monitoring, modern statistical catch-at-age assessment models, and a harvest control rule (e.g., DFO 2016a, Haist and Schweigert 2006; Haist and Stocker 1984; Martell et al. 2012; Stocker 1993). Data collection includes annual fishery-independent surveys of Pacific Herring spawn (egg deposition), catch estimation, and biological sampling for the determination of growth and age characteristics. The assessment models have been modified over time to reflect prevailing hypotheses about stock and fishery dynamics and to improve the statistical fit of models to observed data. However, a single "best" assessment model approach is followed annually where the results of only one assessment model are used as the basis for harvest advice.

Assessment results are input to a harvest control rule (HCR) to calculate a recommended catch. The HCR sets the harvest rate to zero when forecast spawning biomass breaches a threshold biomass (i.e., the "cutoff") and, with some recent exceptions, specifies a target harvest rate of 20% of the forecast spawning biomass when the stock is predicted to be above the threshold biomass. Beginning in 1996, the cutoff was set at 0.25 of the estimate of unfished biomass (B_0) for each major area derived from the 1996 stock assessment model (Schweigert et al. 1997). The cutoff remained fixed at the 1996 estimates until Martell et al. (2012) introduced the practice of estimating the unfished biomass in each annual assessment and setting the cutoff at 0.25 of the updated value. Two management options were provided in subsequent assessments based on a HCR that uses either the 1996 fixed cutoffs or the annual assessment model estimated cutoffs (e.g., DFO 2016a). In keeping with the DFO PA Framework requirement to develop and apply management actions with the involvement of resource users, assessment analysts and managers have worked closely with a diverse array of First Nations, commercial fishery, and other interests to address the challenges posed by the Pacific Herring fisheries.

Although the Pacific Herring management system already meets some requirements of the DFO PA Framework, elements PA1, PA3, and PA4 have not been fully specified. This paper is focussed on the selection of LRPs for the five major stocks of Pacific Herring in partial fulfillment of requirement PA1 and as part of the DFO commitment to renewal of the Pacific Herring management system (DFO 2015a). Identification of upper stock reference points required by element PA1 is not attempted in this paper, nor are TRPs developed. However, a fully specified set of objectives that includes both LRPs and TRPs will be necessary to meet goals for renewal of the Pacific Herring management system and consistency with the *DFO Sustainable Fisheries Framework*, particularly with DFO PA Framework elements PA2-PA4.

The following considerations for LRPs can be drawn from the DFO PA Framework:

1. Biological LRPs should be based on protecting a stock from “serious harm”, which include slowly reversible or irreversible undesirable states;
2. Sustainable fisheries require management to achieve TRPs, therefore there should be a low probability of breaching LRPs; and
3. LRPs need to be considered in the context of the management actions applied if the limit is breached.

When considering BRPs in general, element PA3 of the DFO PA Framework should not be overlooked. Assessment analysts and resource users from the Pacific Herring community have proposed a wide range of structural uncertainties for stock dynamics. For example, questions have been raised about the implications of time-varying natural mortality, observations of declining weight-at-age, stock structure, spatial distribution of spawn, fish movement and ecosystem drivers such as climate change or predator interactions (e.g., Beacham et al. 2008; Benson et al. 2015; DFO 2016a; Fu et al. 2004; Hay et al. 2009; McKechnie 2013; Rose et al. 2008; Schweigert et al. 2010; Small et al. 2005). At least some set of these hypotheses related to Pacific Herring dynamics will be similarly plausible and will not be resolved by available data. Even similarly plausible hypotheses can produce quite different BRPs. Therefore, admitting a range of uncertainties in fulfillment of requirement PA3 means, in effect, that no single set of BRPs can be said with certainty to be correct for a given stock. These challenges mean that the selection and estimation of LRPs is an extremely complex task requiring large amounts of informative data. Both the addition of future data, and the development of models that represent alternative hypotheses for Pacific Herring stock and fishery dynamics, will result in different estimates of LRPs than obtained from the current data and models.

In this paper we begin by reviewing the role of LRPs in Canadian and international fisheries policies with respect to avoiding "serious harm" to a fish stock (e.g., UNFSA 1995). The concept of serious harm is imprecisely defined in policy, but is commonly understood to imply deleterious effects to a stock and fishery. We review definitions of serious harm based on recruitment overfishing and depensatory (Allee) effects. The role of LRPs in defining measurable conservation objectives is discussed, as well as steps necessary to render objectives operational in the context of the ongoing renewal of the BC Pacific Herring management system (DFO 2015a). In particular, we distinguish LRPs defined in measurable objectives from operational control points (OCPs) used in harvest control rules (Cox et al. 2013).

Under a single "best" model approach to assessment and management, such as that followed for BC Pacific Herring, reference points are often selected using simpler "rules of thumb" or policy-based choices founded on meta-analysis of many stocks. In fact, for most cases worldwide management choices are driven by policy (Hilborn and Stokes 2010) and scientific "best practices" (e.g., Sainsbury 2008). We consider the degree to which the BC Pacific Herring management system is aligned with Canadian and international policies, noting that the current assessment and management system was developed well before the establishment of the DFO PA Framework and other policies nested under the umbrella of the Sustainable Fisheries Framework. In fact, the BC Pacific Herring management system introduced in the 1980s anticipated DFO PA Framework elements such as the inclusion of a HCR.

Analyses reported in this paper use the results of two cases of a statistical catch-at-age (SCA) model used to provide harvest advice for the five major stocks of BC Pacific Herring (DFO 2016a). In the main body of the paper we focus on the WCVI management area to illustrate the methods and interpretation of results. Results for stocks in the four other management areas are reported in Appendix A through Appendix D. Current perceptions of status derived from the SCA model cases are illustrated graphically by plotting key model outputs for the 1951 to 2016 period of stock reconstruction. That analysis provides context for the two methods investigated in this paper for diagnosing serious harm.

First, the occurrence of low production, low biomass states over time is evaluated by inspecting the relationship between surplus production and spawning stock biomass. Persistence of such states may be an indicator of serious harm to a fish stock's ability to grow to target levels of spawning biomass and may lead to loss of benefits to resource users. Second, a theoretical equilibrium fishing mortality rate related to the concept of the replacement fishing mortality is investigated, as well as associated proxies based on maximum sustainable yield, spawning potential ratio and yield-per-recruit. Fishing mortality rates that preclude the ability of a fish stock to replace itself are an indicator of possible serious harm. However, since many of the structural uncertainties relevant to Pacific Herring are related to non-stationary processes, it may be difficult to support equilibrium-based BRPs such as those based on MSY, yield-per-recruit or spawning potential ratio (Hilborn 2002; Hilborn and Stokes 2010). Nevertheless, equilibrium fishing mortality rates are calculated to diagnose whether the magnitude of non-stationary effects leads to implausible results that preclude their use.

We provide recommendations on LRPs for each of the five major stocks of Pacific Herring and conclude by outlining the steps needed to identify and evaluate management options expected to avoid LRPs and achieve TRPs. Additional context for LRPs is provided in Appendices E-G. Appendix E describes the goals of the BC Pacific Herring Renewal process and outlines the requirement to maintain an operational stream of short-term harvest advice while developing a long-term strategic stream based on a management-oriented paradigm. The management-oriented paradigm is implemented as an iterative process that emphasizes the establishment of measurable objectives for a stock and fishery. As part of the process, proposed management options are tested using simulation methods to evaluate their relative performance (de la Mare 1998, Kronlund et al. 2014b). This context is needed because LRPs under consideration now, and in the future, will be affected by the Pacific Herring Renewal process. Appendix F contains a summary of management parameters, including LRPs, used for herring stocks worldwide. Simulation studies specific to LRPs for BC Pacific Herring are reviewed in Appendix G. Finally, reference points proposed for forage fish in general are reviewed in Appendix H; these proposed reference points generally reflect considerations for dependent predators and the high variability of population dynamics typically exhibited by forage species.

1.2 SERIOUS HARM AND LIMIT REFERENCE POINTS

A significant challenge to identifying biological LRPs is linking them to the goal of avoiding "serious harm" that underpins international agreements and domestic fisheries policies. For example, the DFO PA Framework states "*… the LRP represents the stock status below which serious harm is occurring to the stock. At this stock status level, there may also be resultant impacts to the ecosystem, associated species and a long-term loss of fishing opportunities.*" This statement establishes three considerations related to serious harm:

1. Serious harm applies not only to the stock of interest, but also to dependent species (e.g., predators) and other ecosystem resources (e.g., habitat);
2. A LRP should be positioned *before* a state of serious harm occurs, rather than *at* the state of serious harm (e.g., at a biomass level above the level where the possibility of serious harm exists or at a fishing mortality rate lower than one expected to produce serious harm); and
3. Long-term loss of benefits to resource users should be avoided.

The DFO PA Framework is partially based on the FAO (1995) definition of the Precautionary Approach that states "*19. Management according to the precautionary approach exercises prudent foresight to avoid unacceptable or undesirable situations, taking into account that changes in fisheries systems are only slowly reversible, difficult to control, not well understood, and subject to change in the environment and human values*". Sainsbury (2008) states that LRPs *"… are set primarily on biological grounds to protect the stock from serious, slowly reversible or irreversible fishing impacts…*". These statements, and consideration (3) above, suggest that states of serious harm need not be restricted to irreversible states, and include states that are only slowly reversible. Failure to prevent states of serious harm could lead to problems such as prolonged loss of harvest opportunities, inability to meet international or domestic policy obligations, stock collapse, loss of genetic diversity, shrinkage of the species' spatial range, or collateral effects on dependent species. The New Zealand harvest strategy (Ministry of Fisheries 2008) states that "*Limits (both 'soft' and 'hard') should be set well above extinction thresholds – rather, they should act as upper bounds on the zone where depensation may occur*".

Although avoiding serious harm is cited as the basis for biologically based LRPs, practical experience shows that it is difficult to uniquely define states of serious harm until they become quite severe (Hilborn and Walters 1992). Paradoxically, this is precisely the situation to be avoided. The difficulty of reliably estimating BRPs based on theoretical considerations is well-established and is due to limitations in our ability to observe complex population processes (Hilborn and Walters 1992). In Canada and other jurisdictions, recruitment overfishing is generally agreed to constitute serious harm (Myers et al 1994; Shelton and Rice 2002). Recruitment overfishing is loosely defined as the state when spawning biomass becomes so small that recruitment declines markedly and, on average, recruitment in a given year is insufficient for the population to replace itself. In practice, identifying the stock size where this occurs is not straightforward, given annual variability in recruitment and noisy or insufficiently informative data. Meta-analyses have been employed in some cases to relate overfishing thresholds with life-history traits and other proxies (e.g., Mace and Sissenwine 1993; Myers et al. 1994; Punt 2000). Growth overfishing occurs when yield per recruit declines below some maximum because high fishing mortality during the rapid growth phase results in loss of the fishery yield that would otherwise be accrued through additional growth. Shelton and Rice (2002) concluded that growth overfishing did not constitute serious or irreversible harm.

Although research has focused on recruitment overfishing as an indicator of serious harm, the emergence of Allee, or depensatory, effects can also be considered to represent serious harm. Allee effects occur when the compensatory response of fish populations to low abundance is compromised (positive density dependence). Sustainable fisheries are based on the assumption that the per capita rate of population increase at low abundance will increase as density-dependent constraints on production are removed (negative density dependence, Nicholson 1933). In contrast, Allee effects arise when the per capita rate of population increase actually decreases as abundance declines (e.g., Courchamp et al 1999).

Until recently, Allee effects have received little attention in the context of LRPs given various meta-analyses that showed little evidence for depensation in fish stock-recruit relationships (e.g., Hilborn et al. 2014; Liermann and Hilborn 1997; Myers et al. 1995). Increased experience with fish populations at low abundance has suggested that Allee effects can arise from both low reproductive success and predation in small populations (Gascoigne and Lipcius 2004; Hutchings 2014, 2015; Hutchings and Rangeley 2011; Keith and Hutchings 2012; Swain and Benoît 2015). In the case of predation, the mortality per predator increases as prey abundance decreases which can produce demographic Allee effects (e.g., due to a type II functional response of predators to prey). So called “emergent” Allee effects can also result when predator mortality that is sustainable at high prey abundance becomes unsustainable with declining prey abundance (Hutchings 2014; Hutchings and Rangeley 2011). Strong evidence of predation-driven Allee effects has been provided for many northwest Atlantic groundfish populations which are expected to decline to extirpation under current conditions even in the absence of fishing (e.g., Swain and Benoît 2015, 2017; Swain and Chouinard 2008; Swain et al. 2016).

The United Nations Fish Stocks Agreement (UNFSA) to which Canada is signatory states *“For stocks which are not overfished, fishery management strategies shall ensure that fishing mortality does not exceed that which corresponds to maximum sustainable yield” (Annex II UNFSA 1995)*. Similarly, the DFO PA Framework states that the reference removal rate from all sources of fishing mortality should not exceed F_{MSY}, i.e., F_{MSY} is a limit fishing mortality rate (Kronlund et al. 2014b; Shelton and Sinclair 2008). This statement is inconsistent since under equilibrium conditions, fishing at F_{MSY} would produce the biomass at maximum sustainable yield (B_{MSY}) which implies B_{MSY} is also a limit biomass level. However, B_{MSY} is commonly regarded as a target level (e.g., UN 2002). In fact, the fishing mortality rate that would produce maximum sustainable yield, F_{MSY}, is, by definition, a valid limit reference point for *growth* overfishing (Mace 2001). Generally, the threshold for recruitment overfishing is understood to be around double the growth overfishing threshold (Goodyear 1993; Mace 1994; 2001; Restrepo et al. 1998; but see Cook et al. 1997; NAFO 2003; Punt 2000 for studies showing that fishing mortality thresholds for growth and recruitment overfishing may be closer together for less productive species). One recommendation is that fish stocks should be managed to avoid growth overfishing, as this should also prevent recruitment overfishing (Gulland 1971).

Regardless of the challenges posed by identifying states of serious harm such as recruitment overfishing or Allee effects, LRPs related to biomass and fishing rates can be categorized in three classes:

1. Model-based LRPs fixed at equilibrium levels (e.g., fractions of B_0, B_{MSY}, F_{MSY}, and yield-per-recruit reference points);
2. Model-based LRPs that dynamically track changes in productivity over time; and
3. Historical LRPs based on previously observed (model-estimated) biomass levels agreed to represent undesirable states.

Several other jurisdictions, notably the USA (Restrepo 1998) and Australia (Sainsbury 2008) have proposed best practices for some of the choices above. We adopt the definition of "best practice" defined by Sainsbury (2008), namely "*The 'best' practice concept is based on the best practice that has been demonstrated through use, and recognizes that views of what is 'best" will continuously improve with experience. Best practice is not an absolute or fixed entity, or a guarantee of adequacy. It is based on experience to date and it is expected to evolve over time.*" The key elements of this definition are that the practice must be *demonstrated through use*, and that best practice is *expected to evolve over time*.

Sainsbury (2008) concluded for target species that the best practice LRP for biomass is the greatest of three quantities (or proxies thereof):

a) B_{LRP}, the biomass below which average recruitment declines or stock dynamics are highly uncertain (i.e., consistent with the concept of recruitment overfishing);

b) the maximum of $0.3B_{\mathrm{unfished}}$, the expected biomass that the stock would return to in the absence of fishing or 0.2 of the median long-term unfished biomass; and

c) the biomass from which rebuilding to the target reference point could be achieved in a period that provides for human intergenerational equity (20-30 years).

The unfished biomass, B_{unfished}, is a dynamic, time-varying estimate provided by model calculations based on the expected stock dynamics in the absence of a fishery. Sainsbury (2080) noted that the equilibrium unfished spawning biomass (B_0) is commonly used as a proxy for B_{unfished} but is vulnerable to violations of equilibrium assumptions (for example if productivity changes). However, we note that for stocks such as Pacific Herring that can exhibit significant fluctuations in productivity (DFO 2016a), the $0.3B_{\mathrm{unfished}}$ level could occur at very low levels of absolute abundance during periods of low productivity and may not serve as a precautionary limit consistent with the DFO PA Framework. For Pacific Herring in Canada, consideration of (c) awaits development of TRPs and policy agreement that intergenerational equity is an acceptable criterion for defending claims of sustainable utilization.

Furthermore, considerable amounts of uncertainty need to be admitted in keeping with DFO PA Framework element PA3. Admitting greater complexity is unlikely to advance improvements to management outcomes under a single best assessment model approach because BRPs derived from over-parameterized stock assessment models are already highly uncertain and difficult to defend. For example, non-stationarity in productivity and carrying capacity (Walters 1986) affects both estimation of BRPs and the management procedures put in place to avoid limits and achieve targets (Haltuch et al. 2008). Pacific Herring observed weight-at-age has declined over the past decade for all five major stocks (DFO 2016a). Assessment model estimates of natural mortality rate vary over time, differ among stock areas, and for at least three of the major stocks have shown an increasing trend over the same period (DFO 2016a). Both these time-dependent processes influence the estimation of population scale and productivity parameters required for calculation of reference points, and can lead to biased estimates. Consequences of non-stationarity to fish population productivity are not well understood generally and there are few evaluations of their effects for Pacific Herring in Canada (e.g., see Cox et al. 2015[1]; Fu et al. 2004; Haist et al. 1993). Possible impacts of non-stationary processes include:

1. Implied changes in BRPs such as B_0 and B_{MSY} that depend on natural mortality, maturity-at-age, growth, or interactions with predatory or competing species;
2. Over- or under-estimation of stock size when time-varying processes are assumed to be stationary in stock assessment models; and
3. Bias in estimates of stock status and fishing rate control points in HCRs that trigger management actions (Haltuch et al. 2008).

The DFO PA Framework acknowledges the challenges of time-varying productivity but states that "*…as a general rule the only circumstances when reference points should be estimated using only information from a period of low productivity is when there is no expectation that the conditions consistent with higher productivity will ever recur naturally or be achievable through management.*" This statement is germane to consideration of Pacific Herring LRPs because of contrary suggestions that it may be desirable to adjust reference points in accordance with changes in productivity attributed to regime shifts or other factors that are not expected to reverse in the *short or medium term*s (e.g., DFO 2013).

The best practice limit reference point for fishing mortality recommended by Sainsbury (2008) is F_{MSY}, the long-term fishing mortality that produces maximum sustainable yield. A suggested acceptable proxy is $F_{50\%}$, the fishing mortality that gives a 50% reduction in the spawning biomass per recruit (SBR) on the basis that for most species, $F_{50\%}$ would provide more than 80% of MSY while depleting spawning biomass to no more than about 30% of the unfished level (Sainsbury 2008).

In the case of important prey species (e.g., forage fish such as Pacific Herring), best practice LRPs may need to reflect requirements for predators that may also be target species. In situations where targeted commercial fishing of forage fish is permitted, best practice reference points would ideally reflect demonstrated requirements to maintain the productivity and ecological role of predators. However, there are very few situations where it is possible to explicitly model these interactions (Hilborn et al. 2017), and none that we are aware of for Pacific Herring. The best practice LRP recommended by Sainsbury (2008) for key prey species is the same as for target species higher in the food chain, but with the additional condition that the probability of breaching 20% of the median unfished biomass should be no more than 10% over some suitable time frame. The DFO PA Framework (Annex 2B) indicates a lower acceptable probability of decline (<5%) as stock states approach a LRP regardless of whether the species is a forage fish.

1.3 THE ROLE OF LIMIT REFERENCE POINTS

The United Nations Fish Stocks Agreement (UNFSA) states that "*Limit reference points set boundaries which are intended to constrain harvesting within safe biological limits within which the stocks can produce maximum sustainable yield* (Annex II UNFSA 1995). The UNSFA also

[1] Cox, S.P., Benson, A.J., Cleary, J.S., and Taylor, N.G. 2015. Candidate limit reference points as a basis for choosing among alternate harvest control rules for Pacific Herring (*Clupea pallasii*) in British Columbia. CSAS Working Paper 2013PEL01 (in prep.)

defines how reference points should be used in concert with harvest control rules by stating *"… management strategies shall ensure that the risk of exceeding limit reference points is very low. If a stock falls below a limit reference point or is at risk of falling below such a reference point, conservation and management action should be initiated to facilitate stock recovery.*" (Annex II UNFSA 1995). Thus, LRPs are thresholds that must be avoided with high probability. Their purpose is to separate management objectives from the operational control points where management actions are triggered (Cox et al. 2013). Limit reference points in isolation are not useful until they are embedded into fully specified measurable objectives. For example, a conservation goal might be to avoid low biomass levels where stock productivity is compromised to the point of serious harm. This goal could be translated to a measurable objective stated as "*avoid spawning biomass levels lower than a threshold biomass* B_{LRP} *with 95% probability over the next 20 years.*" This example illustrates a fully specified measurable objective that has three components:

1. An outcome of interest such as a limit to be avoided or a target to be achieved, e.g., avoid spawning biomass levels less than B_{LRP};
2. A desired probability of achieving the outcome in (1), e.g., 95% probability; and
3. A time-frame over which to measure performance with respect to achieving (1) and (2), e.g., 20 years.

Given a set of measurable objectives related to conservation, economic, and socio-cultural outcomes, actions intended to produce an acceptable tradeoff of management outcomes related to the objectives can be proposed. It is likely that achieving a high probability of avoiding a LRP requires invoking management actions in advance of B_{LRP} being reached, since the LRP is intended as the last barrier to the possibility of serious harm. Establishment of measurable conservation objectives that include LRPs will allow attention to focus on economic and socio-cultural objectives containing TRPs that are needed to articulate the desired states of the stocks and fisheries.

The use of reference points that can be compared to current spawning biomass or fishing mortality is consistent with the DFO PA Framework. However, the DFO PA Framework acknowledges the possibility of other metrics for BRPs that do not require estimation of spawning biomass or fishing rates, such as catch rate indices, size and age profiles, and sex ratios. The policy states "*… other metrics can and should be considered for use in defining serious harm and guiding decision-making in relation to stock condition.*" Additionally, the Sustainable Fisheries Framework "Policy on New Fisheries for Forage Species" states that objectives of this policy include "*maintenance of full reproductive potential of the forage species (including genetic diversity and geographic population structure, whether genetically resolvable or not)*" (DFO 2009b). Pacific Herring migrate inshore to spawn in inter- and sub-tidal waters during late winter and early spring, which suggests the potential for incorporating spatial considerations into objectives consistent with the goals of the Canadian forage fish policy. For example, a goal to maintain a broad spatial distribution of Pacific Herring spawn has been proposed for the West Coast Vancouver Island management area (Nuu-chah-nulth Herring Committee presentation, February 2016). The proposed objective specified the outcome that 70% of the pre-1960s spawn locations should be occupied by year 2025 with 0.75 probability.

There is no impediment to adding fully-specified objectives of this type to more conventional objectives related to spawning biomass or fishing mortality, provided their order of priority is defined. Furthermore, achieving one of the conservation objectives may mean others are also achieved. For example, avoiding spawning biomass levels lower than a specified LRP may mean that the spatial objective for spawn distribution is also met depending on the relationship between biomass and area of spawning occupancy, or vice-versa. Regardless, the limiting constraint cannot be predicted without simulation testing of proposed management options in a reasonable facsimile of the overall management system. This means, however, that quantitative models that can generate reliable performance indicators are needed to be able to evaluate how well spatial objectives are met. Therefore, it may be necessary to delay consideration of spatial objectives until reliable models can be developed or improved data inputs are available.

1.4 HISTORICAL CONSERVATION THRESHOLDS FOR PACIFIC HERRING

In this section we review the design of the BC Pacific Herring harvest control rule in relation to the linkage between LRPs and serious harm described above. We begin by describing the distinction between LRPs and operational control points (OCPs). The role of LRPs is in some sense to act as biological thresholds of last resort intended to prevent the occurrence of undesirable states for fish stocks and fisheries that are irreversible, or only slowly reversible. While LRPs describe biomass or fishing mortality levels to be avoided with high probability, OCPs are components of a HCR that describe when pre-specified management actions are to be taken, such as reduction of the harvest rate or cessation of fishing.

Management actions applied to BC Pacific Herring in the five major management areas over the last 30 years included a HCR that adjusted the intended harvest rate of U=0.2 in response to the forecast spawning stock biomass. Spawning biomass and exploitable biomass were assumed to be equivalent because the fishery was predominately conducted during the spawning period. The HCR can be stated as

(1) $$U_{T+1} = \begin{cases} 0 & B^*_{T+1} \le 0.25B_0 \\ \min\left(\dfrac{B^*_{T+1} - 0.25B_0}{B_{T+1}},\ 0.2 \right) & B^*_{T+1} > 0.25B_0 \end{cases},$$

where T is the terminal year for the stock assessment and B^*_{T+1} is the forecast pre-fishery spawning biomass in year T+1. The output from the harvest control rule is the intended annual harvest rate, U_{T+1}, which is reduced to zero as B^*_{T+1} declines to 0.25B_0 (i.e., the escapement). Alternatively, the HCR can also be stated as

(2) $$U_{T+1} = \begin{cases} 0 & B^*_{T+1} \le 0.25B_0 \\ \dfrac{B^*_{T+1} - 0.25B_0}{B^*_{T+1}} & 0.25B_0 < B^*_{T+1} \le 0.3125B_0 \\ 0.2 & B^*_{T+1} > 0.3125B_0 \end{cases}.$$

Equation (2) makes it clear that there are two biomass-based OCPs at which management actions are triggered. The lower control point at 0.25B_0 is traditionally referred to as the “cutoff”. The 0.31B_0 level is a second biomass-based OCP. This choice of OCPs means that a rapid reduction in harvest rate is invoked in the transition from 0.31B_0 to 0.25B_0. In the design of HCRs, this “ramp-down” of the harvest rate is intended to avoid commercial fishery closures and encourage stock growth. The reduction in equation (2) is so steep however, that there is a very limited range of estimated biomass where reduced harvest rates might arrest stock decline before closure of the commercial fisheries. The effect of this feature of the HCR is the “on or off" behavior seen in the simulation results of Cox et al. (2015[1]). The HCR has a third OCP related to fishing mortality, i.e., the harvest rate of 0.2 implemented in 1983 (Schweigert and Ware 1995[2]).

The technical distinction between LRPs (avoid with high probability) and OCPs (trigger management action) implies nothing about whether the current cutoff values for each of the five major stocks should be considered to be biological LRPs for BC Pacific Herring. In fact, cutoff values have not been used as LRPs in the sense of the DFO PA Framework or worldwide practice because there are no pre-specified management actions taken to avoid breaching them with high probability. However, a distinction between LRPs and OCPs needs to be maintained for two reasons. First, OCPs are quantities that represent stock states estimated from specific data and stock assessment methods. These methods can include population dynamics models or "model-free" methods such as survey trends. Importantly, OCPs are not necessarily directly related to BRPs, as BRPs are usually derived on a theoretical rather than operational basis (Caddy and Mahon 1995). In general, OCPs must be separated from BRPs so that the HCR component of a management procedure can be adjusted to more closely satisfy conservation and yield objectives that include BRPs (Cox et al. 2015[1]).

Second, review of the scientific literature suggests that development of cutoffs for Pacific Herring stocks was originally undertaken with the intent of avoiding serious harm, even though their eventual application was as OCPs. For example, Haist et al. (1986) stated that the cutoffs are biomass levels where commercial fishing should cease and thereby "*… minimize the risk of reaching very low biomass levels where reproduction potential is severely limited*". They additionally commented that the cutoffs should be at levels that "*… allow large population growth for quick rehabilitation.*" Stocker (1993) stated that the cutoffs should "*… minimize the risk of reaching very low biomass where reproductive potential is severely limited and from which the stock requires a long time to rebuild.*" These statements are consistent with an interpretation of the cutoffs that more closely describes LRPs, i.e., biomass levels below which serious harm is thought possible, and the conclusion that cutoffs were intended to prevent slowly reversible states from occurring. Similarly, Schweigert and Ware (1995[2]) state that the "*…Cutoffs for the five major B.C. herring stocks are conservation reference points".*

More recently, Martell et al. (2012) constructed decision tables that showed the consequences of applying a range of fixed catch levels relative to current spawning biomass, B_T, and harvest rate U=0.2. Current spawning biomass was simply a benchmark from which to judge the one-year ahead forecast status of the stock and as such did not represent a limit (or target) reference point. They did not evaluate whether the intended harvest rate of U=0.2 should be considered a limit or target. Regardless, decision-making did not appear to use the decision tables. Instead the HCR was applied with cutoff values based on absolute values of the estimates of $0.25B_0$ derived from the 1996 assessment model (Schweigert et al. 1997) to provide catch recommendations.

However, Martell et al. (2012) did provide the following heuristic argument for a $B_{\mathrm{LRP}}=0.18B_0$ based on application of the DFO PA Framework provisional $B_{\mathrm{LRP}}=0.4B_{\mathrm{MSY}}$. They argued that surplus production for most fish stocks is typically maximized when the stock is depleted to 30-45% of the unfished state. Assuming that Pacific Herring production is maximized at a depletion of 45% means that $B_{\mathrm{MSY}}=0.45B_0$, then $0.4B_{\mathrm{MSY}}=0.4(0.45B_0)=0.18B_0$. If production were

[2] Schweigert, J., and Ware. D. 1995. Review of the biological basis for B.C. herring stock harvest rates and conservation levels. PSARC Working Paper H95:2. 8p.

maximized at $0.3B_0$ then B_{LRP}=$0.12B_0$. However, Martell et al. (2012) did not independently evaluate whether $0.4B_{\mathrm{MSY}}$ was an appropriate LRP for Pacific Herring based on concerns of serious harm. The B_{LRP}=$0.18B_0$ is similar to the quantity $0.2B_0$ in common use as a limit reference point. However, Sainsbury (2008) concluded that while on the basis of empirical evidence $0.2B_0$ avoids recruitment overfishing for productive stocks, it is not regarded as a best practice LRP. The basis for rejection as a best practice LRP was justified on the grounds that $0.2B_0$:

1. Does not avoid recruitment overfishing for low productivity stocks;
2. May not provide adequate protection for other fishing effects likely to be irreversible such as genetic modification, truncated age-structure, impaired spawning success, change in status within food-web dynamics and ease of stock recovery; and
3. May exaggerate errors in estimation or model specification (e.g., less robust to uncertainty leading to higher likelihood that the underlying stock is actually at a lower abundance).

In any case, the results reported below in this paper for the WCVI, HG, and CC management areas indicate that production of Pacific Herring is not maximized at depletion (ratio of spawning biomass to unfished spawning biomass) levels of $0.12\text{-}0.18B_0$ for these stocks and that states consistent with possible serious harm occur at those levels of spawning biomass.

2 METHODOLOGY

2.1 STOCK STATUS

Graphical analysis of the stock and fishery monitoring data and assessment model are presented for the five major management areas of Pacific Herring in BC to provide context for the subsequent production analysis and calculation of equilibrium fishing mortality rates. The graphical analysis shows observed data and model outputs from the 2016 stock assessment for Pacific Herring (DFO 2016a) for two model cases. Following the convention of DFO (2016a), the model cases are denoted AM1 for the case where surface (1951-1987) and dive (1988+) survey catchability parameters are estimated using a prior distribution and AM2 for the case where the surface survey catchability is estimated and the dive survey catchability is fixed at $q_2 = 1$. A multi-plot synopsis of stock status is developed by plotting maximum posterior density (MPD) estimates of key model outputs that include spawning biomass, (log) recruitment deviations, natural mortality rates, and harvest rates. Annual indices of abundance based on surveyed spawn (egg deposition) and observed weight-at-age are also plotted. Results for the WCVI management area are presented below as an example of the analyses; results for other management areas are included in Appendix A.

2.2 PRODUCTION RELATIONSHIPS

We conduct a (surplus) production analysis using model estimates of spawning biomass for the five major stocks of Pacific Herring in BC for both assessment models AM1 and AM2, but with separate analyses for the surface survey and dive survey periods. Hilborn (2001) and Walters et al. (2008) advocated plotting production against biomass to inspect the behaviour of the production function as a function of biomass. Hutchings and Reynolds (2004) and Hilborn and Litzinger (2009) also analyzed production rate as a function of biomass to test for evidence that some stocks recover more slowly than expected from low biomass levels when fishing pressure is removed (i.e., a lower rate of production at low biomass consistent with the lack of a compensatory response). Mohn and Chouinard (2004, their Figure 2) examined the relationships between production and spawning biomass, and production rate and spawning biomass for southern Gulf of St. Lawrence cod to show that the stock had rapidly shifted to a low biomass, low productivity state from near historic high levels of spawning biomass.

A production analysis begins with two assumptions:

1. The index of abundance is proportional to biomass (an assumption shared with the age-structured assessment of BC Pacific Herring); and
2. Catch data record all removals.

Suppose a time series of biomass at the beginning of time t (B_t) and catch biomass during time t (C_t) are available for $t=1,\ldots,T$. Hilborn (2001) defines production (P_t) as

(P1) $P_t = B_{t+1} - B_t + C_t$.

The production is the change in biomass as a function of body growth from year t to t+1, and the catch that was removed in year t. This relationship depends on many fewer assumptions than needed for a statistical catch-at-age model. Let the C_t be determined by

(P2) $C_t = u_t B_t$,

where u_t is the harvest rate in year t. Then, assume an index of abundance (I_t) that is linearly proportional to biomass

(P3) $I_t = qB_t$,

where q is a constant of proportionality or "catchability". This gives the biomass in year t as

(P4) $B_t = \frac{I_t}{q}$.

If catch is assumed known at any time, then given u_t, B_t can be calculated, or given B_t then u_t can be calculated by Eq. P2. Estimates of B_t could be derived from an abundance index with an estimate of q, or obtained from a stock assessment model. Similarly u_t could be calculated given an estimate of B_t. Alternatively, both u_t and B_t could be set at values to represent a hypothesis about the harvest rate or biomass. For BC Pacific Herring, an estimate of B_t can be obtained by assuming a survey catchability value (e.g., q = 1) or by using estimates of B_t obtained from a stock assessment. Hilborn (2001) showed that the Eq. P1 is reasonably robust to the assumed value of q.

If q is determined, then plots of the trajectories of biomass (Eq. 4), harvest rate (via Eq. 2), surplus production (Eq. 1), and phase plots of production against stock biomass can be inspected for patterns that suggest conditions where diminished production occurs and the production relationship may be compromised.

However, unlike most assessment models the BC Pacific Herring model (Martell et al. 2012) estimates end of year spawning biomass rather than beginning of year spawning biomass. In addition, an assumption is made that spawning biomass is observed after the catch is taken. Therefore, an adjustment to Eq. P1 is required that considers when spawning biomass is observed relative to the catch that arose from spawning biomass in year t given by

(P5) $P_t = B_{t+1} - B_t + C_{t+1}$.

Pacific Herring stock assessments in BC have focused on complex age-structured analyses that attempt to use all available data to estimate key parameters in stock-recruitment relationships, recruitment deviations, gear selectivity and natural mortality. Various structural assumptions to allow time-varying natural mortality and/or observed weight at age have been added in efforts to improve statistical model fit (Fu et al. 2004; Haist et al. 1993; Martell et al. 2012). The perceived stock productivity and status are therefore conditioned by model assumptions, which produce differences in outputs used for management decision-making. Although we could have used the observed spawn index values and an assumed value of q to calculate biomass, we chose to use model estimates of spawning biomass for both models AM1 and AM2 for two reasons. First, the values of q for models AM1 and AM2 allow direct comparison to the graphical analysis of stock status described in the previous section and the equilibrium analysis of reference points described in the next section. Second, it is important to separate temporal trends in production estimates from less important interannual variation that results from observation errors. To this end, the stock assessment model imparts some degree of smoothing to the signals in the data series. Hilborn (2001) smoothed the production estimates with a moving average filter, while Peterman et al. (2003) applied a Kalman filter to achieve similar smoothing to avoid chasing noise rather than the underlying signal.

The production analysis implies a third assumption: since the new recruitment in time t depends on recruitments from earlier year classes, and somatic growth also is a function of earlier year classes, plots of surplus production against biomass invoke equilibrium conditions. Evidence of changes in size-at-age for Pacific Herring and of non-stationary natural mortality (DFO 2016a) call this assumption into question. Direct use of the spawn index values as proxies for spawning biomass eliminates dependency on the assumptions of the age-structured stock assessment, but also implies equilibrium conditions and does nothing to separate interannual variation from observation errors. However, the estimates of spawning biomass are obtained from the assessment models already incorporate the effects of time-varying natural mortality and observed annual weight-at-age, so some adjustment for non-stationary effects is contained in the estimates of spawning biomass as well as the benefit of smoothing noted previously.

For Pacific Herring in BC there remains a complication as a result of the transition from surface surveys of spawn (1951-1987) to dive surveys (1988+). The two spawn index time series should not be linked as a single series because survey catchability differs between the two methods, thus assuming a constant q for the entire 1951-2016 time series may be unreasonable. Furthermore, the conditions that led to historical low spawning biomass levels during the surface survey period are not the same as conditions leading to historical low levels for the CC, HG and WCVI stocks during the dive survey period. Therefore, we separate the production analysis into the two survey periods. Nevertheless, adopting the spawning biomass estimates from the assessment models allows comparison of the range of spawning biomass and production to be contrasted between the two periods. This is because estimates are scaled by the survey catchability parameters and results can be compared in terms of the estimated ratio of spawning biomass to unfished spawning biomass (i.e., spawning biomass depletion).

2.3 EQUILIBRIUM REFERENCE POINTS

Avoiding “serious harm” is often framed in terms of avoiding recruitment overfishing (Myers et al 1994; Shelton and Rice 2002). In contrast to growth overfishing there is no precise definition of recruitment overfishing. Conceptually, recruitment overfishing can be defined as the state when fishing has sufficiently reduced the size of the spawning stock so that recruitment is compromised. Recruitment overfishing can be thought of in terms of the stock-recruit relationship, when the spawning biomass is small enough that the recruitment rate is an approximately linear function of spawning biomass (i.e., there is no density-dependence) (Hilborn and Walters 1992). Many stock-recruit datasets show fairly constant recruitment over a wide range of stock sizes. The theory of stock-recruit relationships assumes this characteristic is due to compensatory, or density-dependent, survival of pre-recruits (juveniles), where the survival rate is predicted to increase as the stock is reduced from unfished levels (e.g., Beverton and Holt 1957; Ricker 1954). The assumption of compensatory survival of juveniles is fundamental to the theory of MSY and biologically sustainable fishing (Goodyear 1993; Myers et al., 1999). Mechanisms for compensatory juvenile survival include reduced competition at lower stock sizes, and reduced per-capita predation (Walters and Korman 1999). At low stock sizes, however, most stock-recruit relationships predict a near-linear relationship between spawning biomass and recruits, as compensatory processes break down. In theory, stocks that have been fished down to this level may be considered recruitment overfished as there is no compensatory buffer in juvenile survival.

Perhaps a more intuitive definition of recruitment overfishing is that, on average, recruitment in a given year is insufficient for the population to replace itself. This means that, over its lifetime, each cohort must produce sufficient surviving offspring to produce at least the average number of recruits (R) per unit of spawning biomass (B) for the population (Sissenwine and Shepherd 1987; Mace and Sissenwine 1993). Sissenwine and Shepherd (1987) suggested using the replacement fishing mortality rate (F_{rep}) as a threshold for recruitment overfishing, defined as the fishing mortality rate that would result in the median juvenile survival rate (R / B) observed in the stock recruitment data. Similarly, based on the outcome of a meta-analysis, Myers et al (1994) recommended a threshold based on the spawning biomass at which average recruitment is 50% of R_0, as predicted from the stock-recruit relationship. An important outcome of these early studies is the direct link they showed between equilibrium fishing mortality and the stock-recruit function, via the implied juvenile survival rate or, its inverse B / R, the lifetime average biomass per recruit, under different levels of constant fishing mortality.

In this study, we only consider the Beverton-Holt stock-recruit relationship (Beverton and Holt 1957), as this is the form currently assumed for the BC Pacific Herring catch-at-age assessment model (DFO 2016a). Alternative stock-recruit relationships have been applied to BC Pacific Herring, with Ware and Schweigert (2001, 2002) suggesting that the Ricker stock-recruit function was appropriate under cold environmental conditions for most stocks and a Hockey Stick parameterization under warm environmental conditions. However, the Hockey Stick parameterization resulted in some models failing to converge for some stocks, and it was concluded that the Beverton-Holt parameterization should provide comparable results under most conditions (Ware and Schweigert 2001, 2002). Schweigert et al. (2007) applied both the Beverton-Holt and Ricker stock-recruit parameterizations and noted some convergence failures for the latter depending on the stock and other model structural assumptions.

The Beverton-Holt relationship is a two-parameter model, given by

(E1) $$R = \frac{\alpha B}{1 + \beta B} \quad ,$$

where α is the maximum juvenile survival rate and β controls the degree of density-dependence (if β is zero, the function simplifies to a linear function of biomass with slope α). For a given species with known growth, mortality, maturity and selectivity schedules, and a constant equilibrium fishing mortality rate (F), the corresponding equilibrium B_F / R_F can be calculated as

(E2) $$B_F / R_F = \sum_a l_a w_a m_a \quad ,$$

where w_a is weight-at-age a, m_a is maturity-at-age, and l_a is equilibrium survivorship-at-age (Botsford 1981). Equilibrium survivorship-at-age is a function of selectivity-at-age (s_a), equilibrium natural mortality (M) and equilibrium fishing mortality given by

(E3) $$l_a = \begin{cases} 1 & a = a_{rec} \\ l_{a-1}e^{-(M+s_{a-1}F)} & a_{rec} < a < A \\ \dfrac{l_{a-1}e^{-(M+s_{a-1}F)}}{1-e^{-(M+s_A F)}} & a = A \end{cases},$$

where a_{rec} is age at first recruitment to the fishery and A is the maximum age. Note that Equation E3 assumes a constant equilibrium level of fishing and natural mortality.

Equation E2 provides an explicit linkage between equilibrium fishing mortality and lifetime average production per recruit (B / R), via the survivorship function in Eq. E3. Also, because B / R is the inverse of the juvenile survival rate (R / B), there is a direct link between fishing mortality and replacement, i.e., the inverse of Eq. E2 at any given F is the slope of a straight line, passing through the origin of the stock-recruitment plot (Sissenwine and Shepherd 1987). Three such replacement lines corresponding to different values of F are illustrated in Figure 1.

The extreme case of a fishing mortality rate that should be avoided is that which would cause eventual extinction of the stock (F_{ext}). For the Beverton-Holt and Ricker stock-recruitment relationships, this occurs, by definition, when the slope of the replacement line is equal to the maximum survival rate of the stock-recruit relationship (i.e., as B approaches zero, the slope R / B approaches α; Eq. E1; red dashed line in Figure 1). If an estimate of α is available from a stock assessment, Eq. E2 can be solved for F_{ext} by numerically searching for the value of F that results in $B/R = 1/\alpha$ (Shepherd 1982). Obviously, a recruitment overfishing threshold (LRP) that would result in extinction is not precautionary because it occurs at or past the point of irreversible serious harm.

The equilibrium spawning biomass associated with F_{rep} is B_{rep}, which could be considered as a recruitment overfishing threshold (Sissenwine and Shepherd 1987; black dotted line Figure 1). The slope of the F_{rep} replacement line is defined as the median R / B from the estimates of spawning biomass and recruits. However, given the sensitivity of the scale of estimated spawning biomass to stock assessment assumptions, especially for Pacific Herring (DFO 2016a), we do not recommend selecting an absolute value for the LRP. One option to avoid the scaling problem is to express B_{rep} as a ratio relative to B_0. Best practice LRPs are commonly expressed relative to B_0 (Sainsbury 2008). However, setting a status-based LRP on the spawning biomass (scaled by B_0) implied by equilibrium fishing mortality rates may be problematic for stocks such as Pacific Herring, which have high recruitment variability (see above discussion of Sainsbury (2008) recommendations).

The consequences of high recruitment variability are illustrated in Figure 1, which shows the F_{rep} replacement line and the B_0 replacement line for the WCVI stock. For this stock, B_0 is estimated to be considerably lower than the maximum estimated spawning biomass with approximately half of the estimates of biomass larger than B_0 (Figure 1). This is a result of high estimated recruitment variability, producing much larger than average spawning biomasses in many years. Therefore, in the case illustrated in Figure 1, B_{rep} occurs at approximately 75% of B_0.

It follows from Equation 2 that any equilibrium fishing mortality rate has an associated replacement line related to the stock-recruit curve (Figure 1). A proxy reference point that is commonly used in US fisheries is the replacement Spawning Potential Ratio (SPR), defined as the ratio of B / R at F_{rep} to unfished *B/R* (Clark 1991; Mace and Sissenwine 1993), i.e.,

$$\text{(E4)} \quad \mathrm{SPR} = \frac{B_{F=F_x}/R_{F=F_x}}{B_{F=0}/R_{F=0}} \quad ,$$

where F_x is a proxy for F_{rep} that produces a value of $\mathrm{SPR}=x$. An advantage of the SPR is that it does not depend upon estimates of recruitment parameters, although it does depend on estimates of M and other life history parameters (Equations E2-E3). It can therefore be applied when recruitment estimates are absent, or uncertain, as for Pacific Herring.

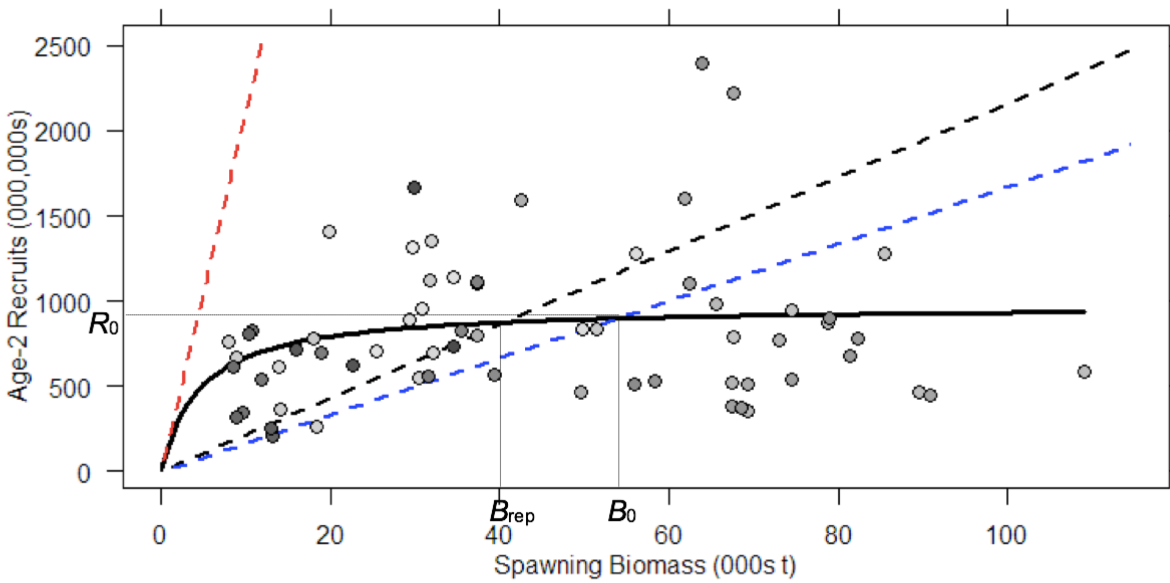

Figure 1. Maximum posterior density (MPD) estimates of annual age-2 recruits (grey circles) for the WCVI Pacific Herring population from model AM1. The solid black line shows the Beverton-Holt stock-recruit relationship based on MPD estimates of recruitment parameters from model AM1. The red dashed line represents the replacement line with slope equal to the maximum juvenile survival rate, α. It is associated with the fishing mortality rate F_{ext}. The blue dashed line represents the replacement line with slope equal to the unfished equilibrium juvenile survival rate (i.e., at B_0). By definition, the equilibrium recruitment at this stock size is R_0 and the fishing mortality rate is F_0. Note that B_0 (and the corresponding R_0) are considerable smaller than the maximum estimated values (see text). The black dashed line represents the replacement line with slope equal to the median juvenile survival rate obtained from the MPD estimates of spawning biomass and recruits. The fishing mortality that would theoretically produce this rate at equilibrium is F_{rep}. Lighter circles represent older recruitment estimates. Darker circles represent more recent estimates.

A number of studies have been done to relate SPR to fishery objectives. Most commonly, these studies have been meta-analyses using data-rich species in order to draw conclusions about the relationship of SPR to fishery reference points, most often F_{MSY}. As a general conclusion, an SPR in the range 0.35 - 0.5 could be considered a reasonable proxy for F_{MSY} for most stocks (Clark 1991; 1993; 2002; Mace 1994; Dorn 2002). SPR has also been considered as a candidate proxy for F_{rep}, specifically in the context of recruitment overfishing (Sissenwine and Sheperd 1987). Mace and Sissenwine (1993) performed a meta-analysis, based on stock-recruit data for 91 Atlantic stocks, and suggested that "replacement" SPR averaged 0.187 but ranged from 0.02 to 0.65 for the species they considered. Importantly for the present study, they noted that small pelagic stocks showed the lowest resilience to fishing and had correspondingly higher than average estimates of replacement SPR, with Baltic Herring and other Clupeids having an estimated average replacement SPR of 0.375. This is in agreement with the findings of Goodwin et al. (2006) who found that Atlantic herring stocks could be fished down to low levels at relatively low harvest rates. Walters and Martell (2004) warned that values of SPR less than 0.3 could substantially increase the risk of recruitment overfishing and that values less than 0.1 have led to recommendations of fishery closures. Walters and Kitchell (2001) suggested that values of SPR less than 0.5 could lead to long-term changes in community structure and invite the possibility of depensatory effects through changes in predator-prey relationships.

In recent years there has been renewed interest in using F_{MSY} as a limit reference point rather than a target in both single species and ecosystem-based management contexts (Mace 2001; Punt and Smith 2001). Meta-analytical studies have suggested that F_{MSY} represents a precautionary limit to fishing mortality for preventing both growth and recruitment overfishing (Cook et al. 1997; Mace 1994; NAFO 2003; Punt 2000).

One more potential candidate for a LRP is the equilibrium spawning biomass associated with $F_{0.1}$, which is defined as the fishing mortality rate that corresponds to a point on the yield per recruit function with a slope of 10% of the slope at the origin (Gulland and Boerema 1973; Figure 2). $F_{0.1}$ is intended as a precautionary reference point, although 10% was an arbitrary choice. Mace and Sissenwine (1993) found that the relationship between $F_{0.1}$ and F_{rep} was inconsistent among their 91 stocks with $F_{0.1}$ sometimes greater, and sometimes smaller, than F_{rep}. Under the assumption that F_{rep} is an appropriate measure of recruitment overfishing, they therefore concluded that adoption of an $F_{0.1}$–based LRP was not guaranteed to safeguard against recruitment overfishing. They did, however, find that $F_{0.1}$ was less than F_{rep} for 13 out of 17 of the Clupeid stocks they examined; implying that in most cases it was more precautionary.

Using MPD estimates of key model parameters from model AM1 (DFO 2016a), we calculate equilibrium fishing mortality rates F_{ext}, F_{rep}, F_{SPR30}, F_{SPR40}, and F_{MSY} and $F_{0.1}$ and the relative equilibrium spawning biomass B associated with each one, for the five major BC Pacific Herring stocks. Note that we used the long-term average value for M for each stock, and our results are conditional on this assumption. The replacement fishing mortality F_{rep}, was obtained by calculating the median slope B / R from the MPD stock-recruit estimates, then numerically solving Equation 2 for the value of F that resulted in that value of B / R. The fishing mortality rates F_{SPR30} and F_{SPR40} were obtained by numerically solving Equation E4 for the value of F_{x} that produced SPR=0.3 or SPR=0.4, respectively.

Fishing mortality at maximum sustainable yield, F_{MSY}, was calculated by using a Newton-Raphson algorithm to numerically solve for the value of equilibrium fishing mortality F that maximized the equilibrium yield Y function

(E5) $$Y_e = F_e \varphi_q R_e \quad ,$$

where Y_e is equilibrium yield, F_e is the equilibrium fishing mortality rate, φ_q is equilibrium vulnerable biomass per recruit (Martell et al. 2008), and R_e is equilibrium recruitment. The latter parameter is given by

(E6) $$R_e = \frac{\alpha\varphi_B - 1}{\beta\varphi_\beta} \quad ,$$

where φ_β is equilibrium spawning biomass per recruit (Equation E2) and φ_q is a function of F, M and selectivity-at-age s_a given by

(E7) $\varphi_q = \sum_a \frac{l_a w_a s_a}{M + s_a F}\left(1 - e^{-(M+s_a F)}\right)$,

where survivorship-at-age, l_a , is evaluated at F (Martell et al. 2008).

Finally, the fishing mortality rate $F_{0.1}$ was calculated by numerically solving for the value of F that resulted in the point on the yield per recruit function with a slope of 10% of the slope at the origin, illustrated in Figure 2 using

(E8) $Y/R = F_e \varphi_q$.

We discuss consistency of the equilibrium reference point results with those from the production analysis and the literature review. We also consider general limitations of equilibrium analyses for these stocks.

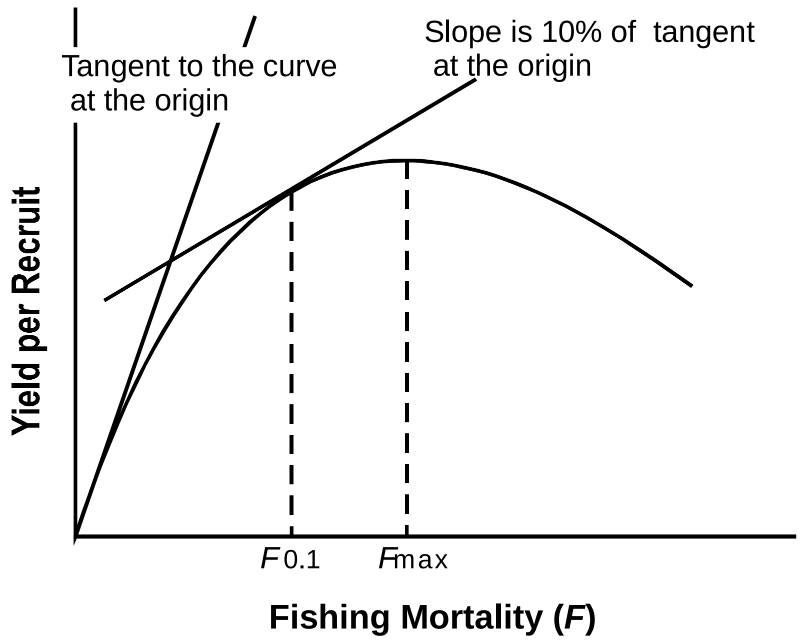

Figure 2. Location of F0.1 and Fmax on yield per recruit curve (Equation E8) (redrawn from Gulland and Boerema 1973).

3 RESULTS

3.1 STOCK STATUS

We use the Pacific Herring stock in the WCVI management area to illustrate the results in detail. Figures for management areas other than WCVI are provided in Appendix A. There are currently two assessment model cases (AM1 and AM2) for each of the five major management areas described by DFO (2016a). Perceptions of WCVI stock status based on outputs from the catch-at-age assessment model are summarized in Figure 3 for AM1 and in Figure 4 for AM2. Each figure consists of six panels (a-f):

a) time series of total catch and estimated spawning biomass with reference lines at the model estimates of $0.1B_0$, $0.25B_0$, $0.3B_0$, the 1996 fixed cutoff (Schweigert et al. 1997) used with AM2 in DFO (2016a) and the model estimate of B_0;

b) observed and fitted time series of surface (1951-1987) and dive (1988-2016) survey index values scaled to spawning biomass by division by their respective estimates of catchability parameters, q_1 and q_2; a 3-year trailing moving average smoother is overlaid on each index series to summarize trend;

c) time series of (log) deviations from the estimated Beverton-Holt recruitment function overlaid with a 3-year trailing moving average smoother;

d) time series of natural mortality (M) estimates;

e) time series of the average of the observed weight-at-age for age-classes 2-6; and

f) time series of estimated harvest rates with reference lines at the intended harvest rate (U=0.2, or 0.1 and 0.07 for the CC area in 2015 and 2016 respectively) and at the average harvest rate from 1983 to the first year of commercial fishery closure or 2016.

The model reconstruction of spawning biomass for AM1 (Figure 3a) suggests that the estimated spawning biomass increased from historic lows to levels above the estimate of unfished biomass 3 years after cessation of the commercial reduction fishery in 1968, and to a historic high level in 8 years (1975). Above-average levels of spawning biomass were sustained until about the late 1990s, and then the biomass declined to below the estimated $0.25B_0$ level over the first half of the 2000s to levels near those estimated during the collapse of the late 1960s by 2010. The last three years of the reconstruction show an increase in spawning biomass to above the $0.3B_0$ level following 8-10 years of relatively low estimated spawning biomass. The rate of increase from low spawning biomass levels to historic high levels in the late 1960s/ early 1970s occurred more rapidly than the more modest increase from similar lows in the mid-2000s to mid-2010s to near $0.3B_0$. This outcome occurred despite the cessation of commercial roe fishing in 2006 and commercial spawn-on-kelp fishing in 2007.

Figure 3b shows that the spawning biomass reconstruction closely follows the trends in the surface and dive survey spawn index values as expected. Recruitment deviations (Figure 3c) show a generally negative trend on average, particularly so after about 2000 until 2012. Estimated natural mortality (Figure 3d) has an increasing trend since 1990 with the highest estimates occurring in the late 2000s. Over this same period, there has been a generally declining trend in weight at age (Figure 3e). Finally, Figure 3f shows that the estimated harvest rate has been near or below the intended target harvest rate of U=0.2 since 1983 with an average of 0.12 from 1983 to 2005; prior to the 1968 closure harvest rates are estimated to be 30 to 80 percent of the spawning biomass in most years.

The patterns of spawning biomass, recruitment deviations, and estimated natural mortality are similar for AM2 (Figure 4). The primary difference between the AM1 and AM2 reconstructions is the decrease in biomass scale that results from fixing the dive survey catchability to $q_2 = 1$. The average harvest rate from 1983 to 2005 correspondingly increases to 0.17.

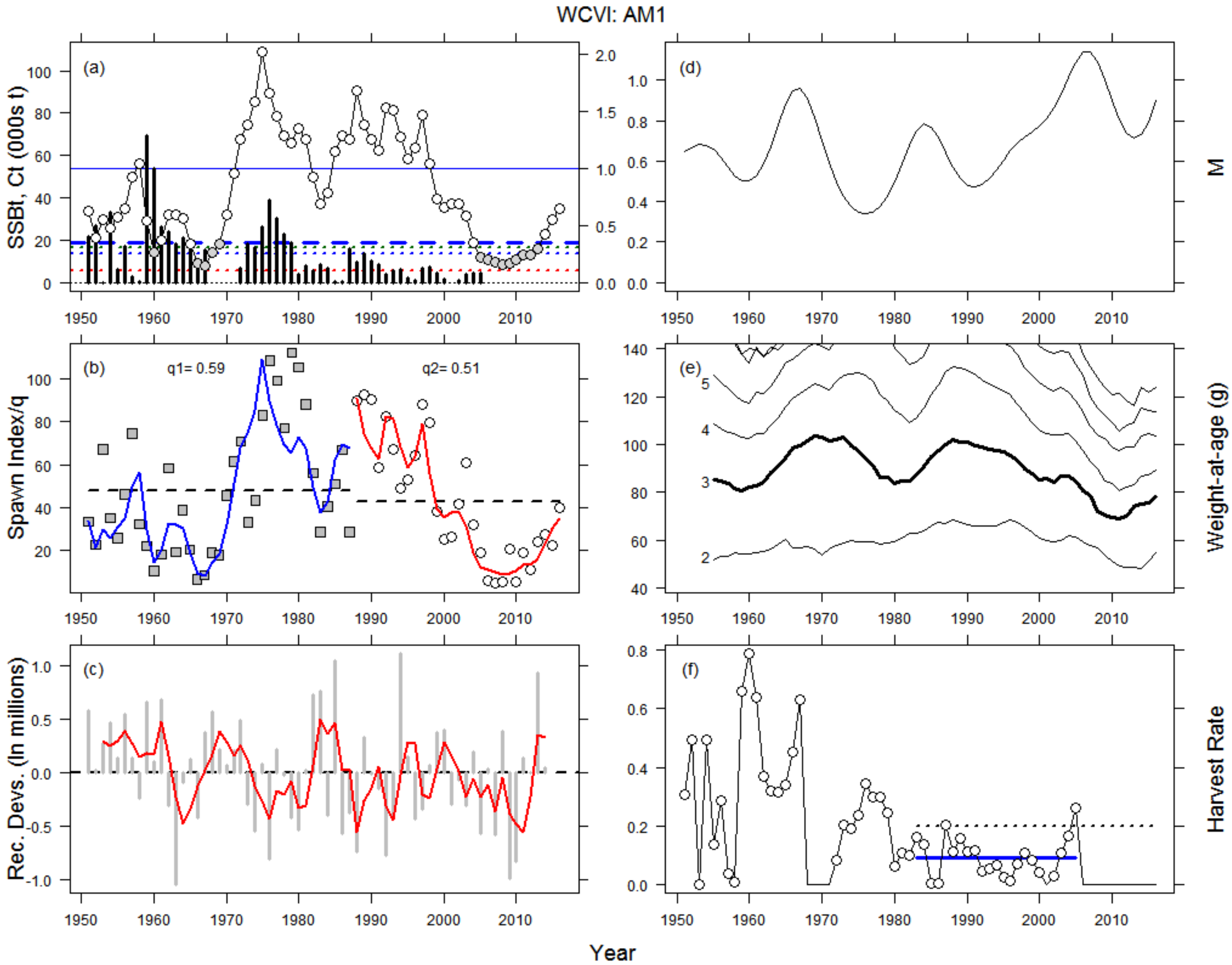

Figure 3. Assessment model AM1 stock reconstruction for WCVI Pacific Herring. Panel (a) shows the 1951-2016 time series of estimated spawning biomass (circles) and catch (bars). The lower 20% of spawning biomass estimates are shaded grey. Reference lines are shown at estimates of 0.1B_0, (red dashed line) 0.25B_0 (blue dashed line), 0.3B_0 (green dashed line), the 1996 fixed cutoff value (thick blue long dashed line), and unfished spawning biomass (solid blue line). Panel (b) shows observed surface (grey squares) and dive (open circles) survey indices scaled to spawning stock biomass. A trailing 3-year moving average smoother indicates trend for the surface (solid blue line) and dive (solid red line) survey indices. The horizontal dashed line is positioned at the mean observed value for each survey series. Estimated q_1 (surface survey) and q_2 (dive survey) are shown in the panel. Recruitment deviations (grey bars) from a Beverton-Holt stock-recruitment function are shown in panel (c). A 3-year trailing moving average smoother (red line) shows the trend in deviations. Estimated natural mortality is shown in panel (d). Panel (e) shows a 3-year trailing moving average of observed weight-at-age for age classes 2-6. Estimated harvest rates are shown in panel (f). Reference lines are shown at the intended harvest rate of 0.2 (horizontal dotted line) and at the average harvest rate from 1983 to the first year of commercial fishery closure or 2016. All model estimates are the maximum likelihood estimates (DFO 2016a).

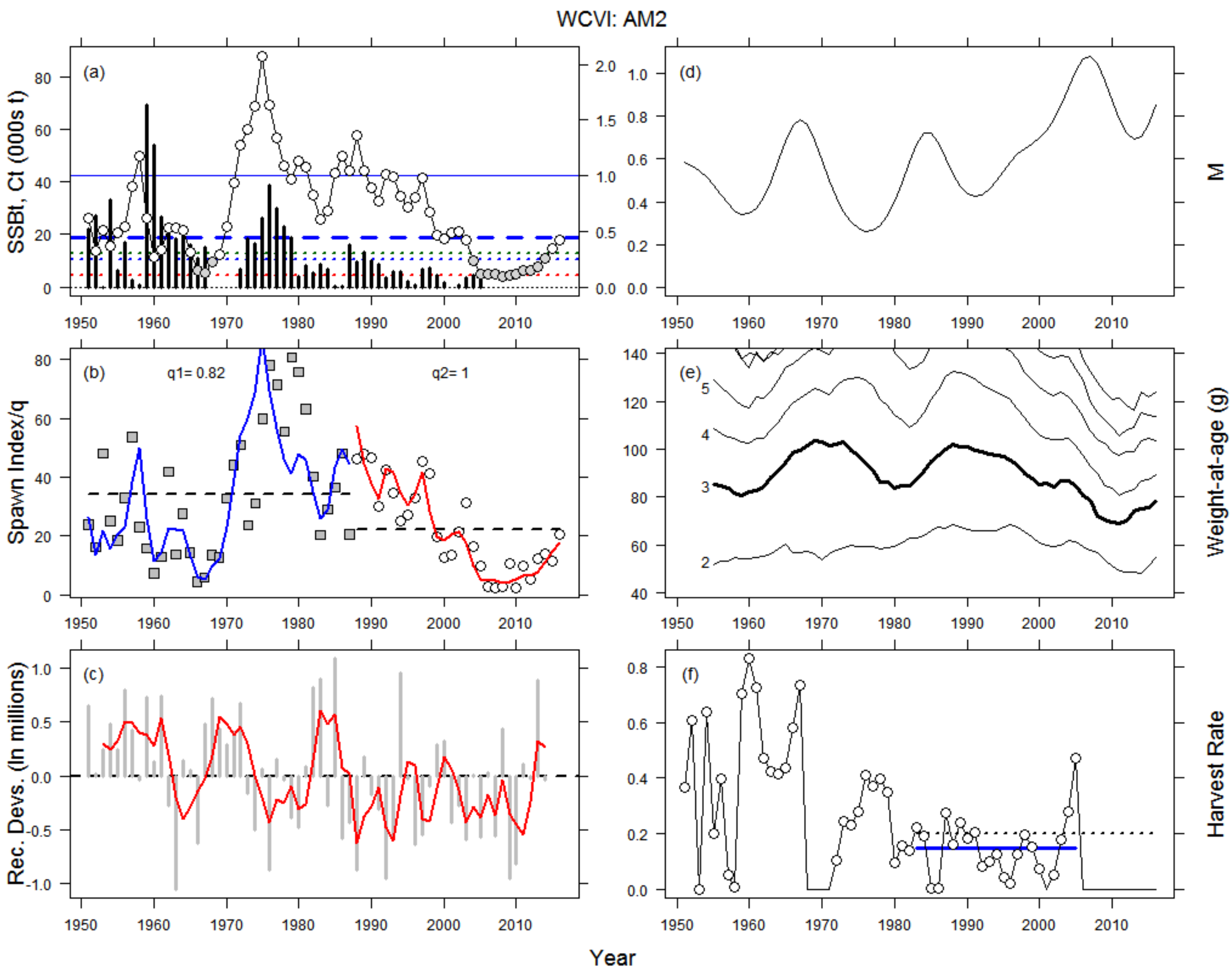

Figure 4. Assessment model AM2 stock reconstruction for WCVI Pacific Herring. Description as for Figure 3.

3.2 PRODUCTION RELATIONSHIPS

Results from the production analysis are described in detail here for the WCVI management area. Figures for management areas other than WCVI are provided in Appendices B (time-series of production estimates) and C (phase plots of production estimates against spawning biomass). Time-series plots shown in Figure 5 for model AM1 and Figure 6 for model AM2 are based on production calculated from estimated annual spawning biomass (MPDs) and catches as determined by Eq. P5 (Table 1 and Table 2 for model AM1 and AM2, respectively). The three panels (a-c) of each figure show:

a) Time-series of observed spawn index values scaled to spawning biomass by dividing by the estimates of catchability parameters (q_1, q_2) for the surface and dive survey series, respectively;

b) Time-series of spawning biomass production estimates with a 3-year moving average smoother overlaid to indicate trend; and

c) Time-series of spawning biomass *production rate* with a 3-year trailing moving average smoother overlaid to indicate trend.

Results for both models AM1 and AM2 show that average production after 1987 is substantially lower than the 1951-1987 period, with zero or negative production in many years. The positive range of production is much smaller during the (latter) dive survey period, with only production in 1991 exceeding the 1951-1987 average production. Furthermore, the production rate is on average lower, meaning that less surplus biomass is produced per ton of spawning biomass after 1987. A decline in production during the early 1980s was followed by a period of declining spawning biomass by the mid-1980s. Production increased briefly in the late 1980s which correspondingly increased spawning biomass before a generally declining trend in production to negative levels (on average) commenced until the late 2000s when production remained near 0 until about 2013. In comparison, production and production rate were positive and much higher during the stock collapse of the late 1960s, which supported the rapid rebuilding of the stock to historically high levels by the mid-1970s (Figure 5 and Figure 6).

Phase plots of production and production rate against spawning biomass are shown in Figure 7 (AM1) and Figure 8 (AM2). Plotted points are shaded such that darker greys represent progressively more recent years. The surface survey and dive survey periods are separated into the left and right columns of the figures, respectively. During the dive survey period, the phase plot shows a large negative production value in 1997 at a spawning biomass level well above the estimate of B_0 for model AM1 (Figure 7). Subsequently there is a rapid reduction in spawning biomass during a period of near zero production from 1999-2001, followed by 3 years of negative production before the stock settles into a low production, low biomass (LP-LB) state with values that are typically in the lower 30% of spawning biomass and production values (Table 1). A cluster of values at low biomass and low production sustained over a period of years was used to diagnose a LP-LB state, which was considered to end when a sustained increase in spawning biomass for at least 2-3 years occurred. A temporary (1-2 year) increase of biomass and production was not considered to be sufficient as a sign of recovery from a LP-LB state. Beginning in 2013, increased production resulted in increased spawning biomass approaching the level estimated in 2000. The commercial fishery has remained closed in this area since 2006, a period of 11 years. Spawning stock depletion levels from 2004 to 2012 ranged between 0.16 and 0.24.

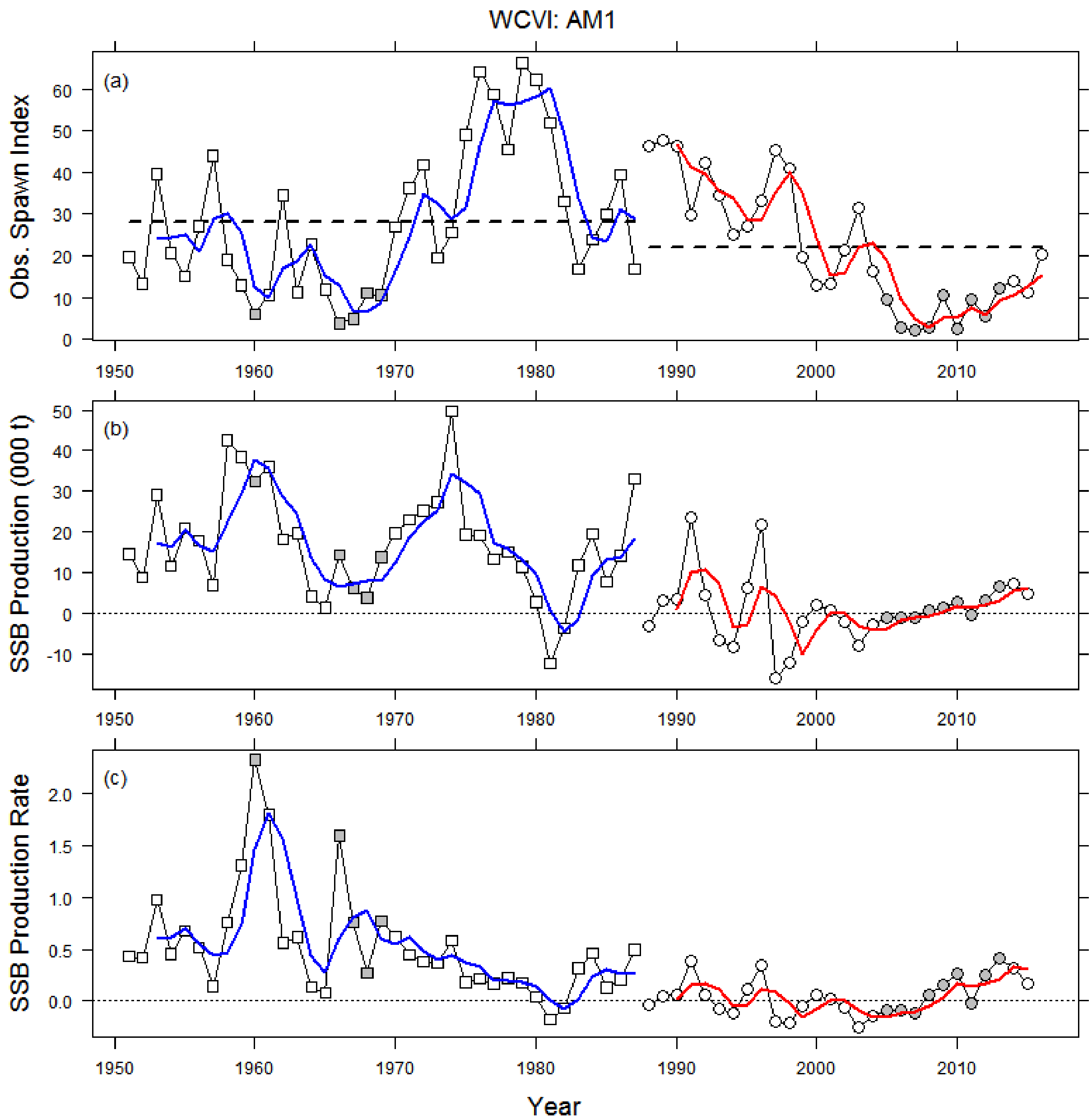

Figure 5. Time series of observed spawn index scaled by catchability parameters (panel a), spawning stock biomass (SSB) production (panel b), and SSB production rate (panel c) for model AM1. The 1951-2015 time series is broken into two periods corresponding to the surface survey (1951-1987, open squares) and dive survey (1988-2015, open circles). Years with SSB values in the lower 20% of the 1951-2015 series are shaded grey. Blue and red lines indicate a 3-year trailing moving average smoother applied to the plotted values. Horizontal dashed lines in panel (a) indicate the mean survey index value for each survey period. Horizontal dotted lines are positioned at zero production and production rate in the two lower panels.

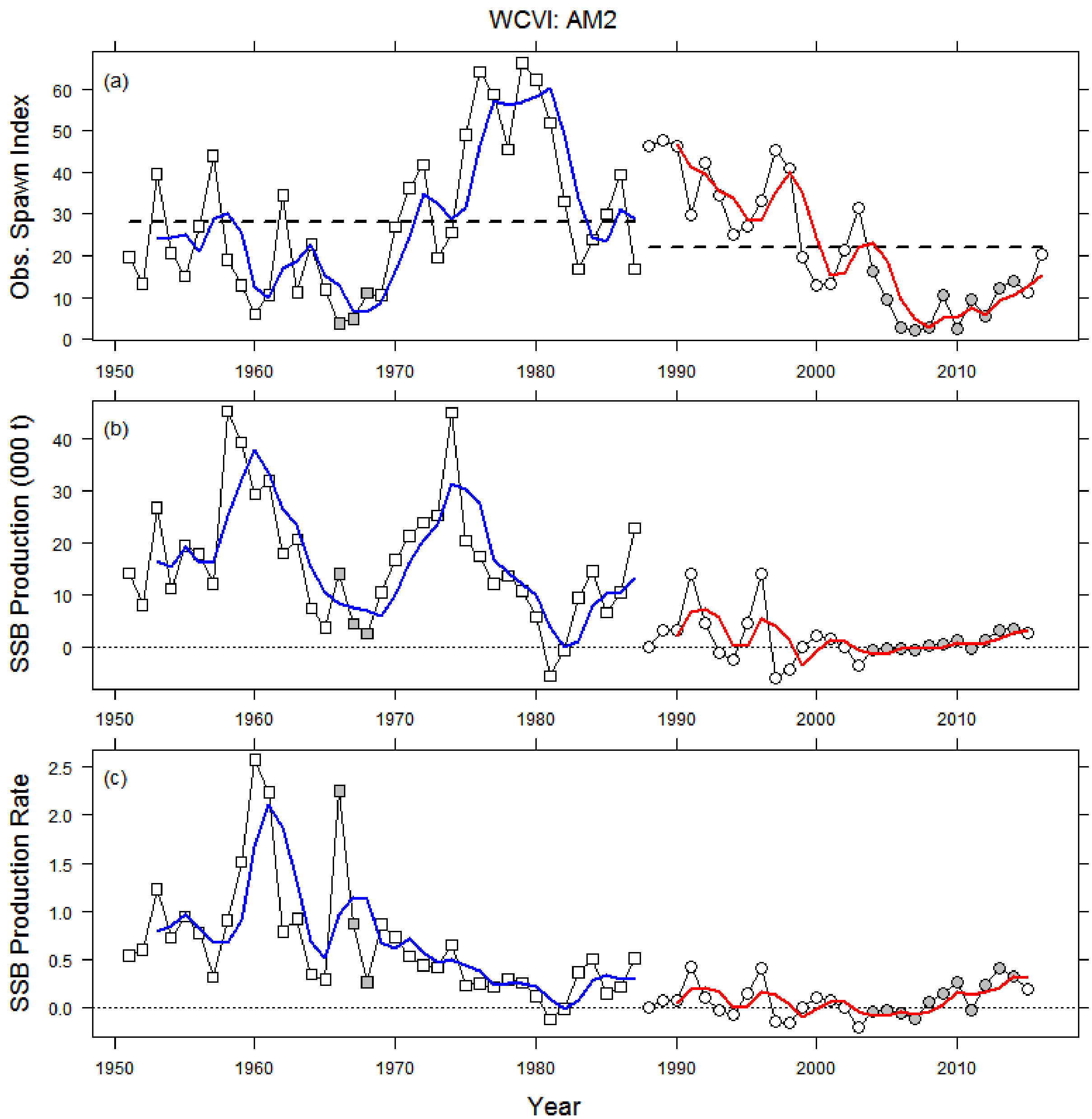

Figure 6. Time series of observed spawn index (panel a), spawning stock biomass (SSB) production (panel b), and SSB production rate (panel c) for model AM2. Description as for Figure 5.

Table 1. Annual estimates of spawning biomass (B, 000s t), spawning biomass depletion (D), spawn index (I), spawn index scaled by catchability (I/q), catch (C, 000s t), harvest rate (U), surplus production (P), and production rate (P / B) for the WCVI management area for model AM1. Years during the stock collapse from 1966-1969 are shaded light grey and years with a recent persistent LP-LB state from 2005-2012 are shaded dark grey.

Year	Period	B	D	I	I/q	C	U	P	P/B
1951	Surface	33.54	0.62	19.60	33.18	21.82	0.31	14.40	0.43
1952	Surface	20.93	0.39	13.31	22.53	27.01	0.49	8.74	0.42
1953	Surface	29.65	0.55	39.57	66.99	0.02	0.00	28.97	0.98
1954	Surface	25.41	0.47	20.65	34.96	33.21	0.49	11.48	0.45
1955	Surface	30.77	0.57	15.11	25.58	6.12	0.13	20.90	0.68
1956	Surface	34.57	0.64	27.18	46.02	17.10	0.29	17.78	0.51
1957	Surface	49.73	0.92	44.11	74.68	2.61	0.04	6.91	0.14
1958	Surface	56.09	1.04	18.99	32.14	0.56	0.01	42.50	0.76
1959	Surface	29.36	0.55	12.98	21.97	69.22	0.66	38.46	1.31
1960	Surface	13.91	0.26	6.02	10.18	53.91	0.79	32.42	2.33
1961	Surface	19.90	0.37	10.56	17.87	26.44	0.64	35.91	1.80
1962	Surface	32.12	0.60	34.47	58.36	23.68	0.37	18.08	0.56
1963	Surface	32.00	0.59	11.25	19.04	18.21	0.32	19.66	0.61
1964	Surface	30.39	0.56	22.76	38.53	21.27	0.32	4.03	0.13
1965	Surface	18.38	0.34	11.89	20.13	16.05	0.34	1.38	0.08
1966	Surface	8.92	0.17	3.72	6.30	10.84	0.45	14.27	1.60
1967	Surface	8.04	0.15	4.81	8.15	15.15	0.63	6.09	0.76
1968	Surface	14.13	0.26	11.03	18.67	0.00	0.00	3.81	0.27
1969	Surface	17.94	0.33	10.47	17.72	0.00	0.00	13.90	0.78
1970	Surface	31.84	0.59	26.91	45.56	0.00	0.00	19.63	0.62
1971	Surface	51.47	0.96	36.21	61.30	0.00	0.00	22.96	0.45
1972	Surface	67.53	1.25	41.86	70.86	6.89	0.08	25.34	0.38
1973	Surface	74.57	1.38	19.48	32.98	18.30	0.20	27.26	0.37
1974	Surface	85.50	1.59	25.54	43.24	16.33	0.19	49.69	0.58
1975	Surface	109.09	2.03	49.15	83.21	26.11	0.24	19.38	0.18
1976	Surface	89.65	1.66	64.20	108.69	38.83	0.34	19.12	0.21
1977	Surface	78.73	1.46	58.68	99.34	30.04	0.30	13.29	0.17
1978	Surface	69.27	1.29	45.61	77.21	22.75	0.30	15.03	0.22
1979	Surface	65.61	1.22	66.40	112.41	18.69	0.25	11.34	0.17
1980	Surface	72.96	1.35	62.31	105.49	3.98	0.06	2.62	0.04
1981	Surface	67.50	1.25	52.01	88.06	8.09	0.11	-12.39	-0.18
1982	Surface	49.62	0.92	33.05	55.95	5.49	0.10	-3.68	-0.07
1983	Surface	37.37	0.69	16.77	28.39	8.58	0.16	11.70	0.31
1984	Surface	42.49	0.79	23.87	40.41	6.58	0.14	19.47	0.46
1985	Surface	61.79	1.15	30.01	50.81	0.18	0.00	7.73	0.13
1986	Surface	69.31	1.29	39.51	66.90	0.20	0.00	14.17	0.20
1987	Surface	67.55	1.25	16.86	28.54	15.93	0.20	33.06	0.49
1988	Dive	90.89	1.69	46.24	89.94	9.72	0.11	-3.14	-0.03
1989	Dive	74.45	1.38	47.72	92.81	13.29	0.16	2.89	0.04

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1990	Dive	67.49	1.25	46.46	90.37	9.85	0.11	3.57	0.05
1991	Dive	62.43	1.16	30.00	58.34	8.64	0.12	23.61	0.38
1992	Dive	82.32	1.53	42.37	82.40	3.71	0.05	4.61	0.06
1993	Dive	81.32	1.51	34.41	66.92	5.61	0.05	-6.74	-0.08
1994	Dive	68.53	1.27	25.25	49.11	6.04	0.07	-8.25	-0.12
1995	Dive	58.34	1.08	27.13	52.76	1.95	0.02	6.28	0.11
1996	Dive	63.82	1.19	33.12	64.42	0.79	0.01	21.75	0.34
1997	Dive	78.91	1.47	45.36	88.23	6.66	0.07	-16.10	-0.20
1998	Dive	55.83	1.04	41.01	79.77	6.98	0.11	-12.03	-0.22
1999	Dive	39.42	0.73	19.73	38.38	4.37	0.08	-2.29	-0.06
2000	Dive	35.50	0.66	12.80	24.89	1.63	0.04	1.86	0.05
2001	Dive	37.37	0.69	13.41	26.09	0.00	0.00	0.74	0.02
2002	Dive	37.28	0.69	21.24	41.32	0.82	0.03	-2.28	-0.06
2003	Dive	31.48	0.58	31.40	61.07	3.52	0.11	-8.02	-0.25
2004	Dive	19.00	0.35	16.43	31.96	4.45	0.16	-2.85	-0.15
2005	Dive	11.88	0.22	9.66	18.80	4.27	0.26	-1.12	-0.09
2006	Dive	10.76	0.20	2.88	5.59	0.00	0.00	-1.05	-0.10
2007	Dive	9.71	0.18	2.25	4.37	0.00	0.00	-1.22	-0.13
2008	Dive	8.50	0.16	2.74	5.33	0.00	0.00	0.49	0.06
2009	Dive	8.98	0.17	10.61	20.63	0.00	0.00	1.41	0.16
2010	Dive	10.39	0.19	2.46	4.79	0.00	0.00	2.77	0.27
2011	Dive	13.16	0.24	9.66	18.79	0.00	0.00	-0.26	-0.02
2012	Dive	12.90	0.24	5.41	10.52	0.00	0.00	3.14	0.24
2013	Dive	16.04	0.30	12.34	24.00	0.00	0.00	6.53	0.41
2014	Dive	22.57	0.42	13.94	27.11	0.00	0.00	7.27	0.32
2015	Dive	29.84	0.55	11.32	22.02	0.00	0.00	4.77	0.16
2016	Dive	34.61	0.64	20.53	39.93	0.00	0.00	*NA*	*NA*

Table 2. Annual estimates of spawning biomass (B, 000s t), spawning biomass depletion (D), spawn index (I), spawn index scaled by catchability (I/q), catch (C, 000s t), harvest rate (U), surplus production (P), and production rate (P/B) for the WCVI management area for model AM2. Years during the stock collapse from 1966-1969 are shaded light grey and years with a recent persistent LP-LB state from 2004-2014 are shaded dark grey.

Year	Period	B	D	I	I/q	C	U	P	P/B
1951	Surface	26.32	0.62	19.60	23.84	21.82	0.37	14.27	0.54
1952	Surface	13.58	0.32	13.31	16.19	27.01	0.61	8.21	0.61
1953	Surface	21.77	0.51	39.57	48.14	0.02	0.00	26.85	1.23
1954	Surface	15.41	0.36	20.65	25.12	33.21	0.64	11.25	0.73
1955	Surface	20.53	0.48	15.11	18.39	6.12	0.20	19.44	0.95
1956	Surface	22.87	0.54	27.18	33.07	17.10	0.40	17.91	0.78
1957	Surface	38.17	0.90	44.11	53.67	2.61	0.05	12.25	0.32
1958	Surface	49.87	1.18	18.99	23.10	0.56	0.01	45.30	0.91
1959	Surface	25.95	0.61	12.98	15.79	69.22	0.70	39.35	1.52
1960	Surface	11.39	0.27	6.02	7.32	53.91	0.83	29.33	2.57
1961	Surface	14.29	0.34	10.56	12.84	26.44	0.73	31.96	2.24
1962	Surface	22.57	0.53	34.47	41.94	23.68	0.47	17.98	0.80
1963	Surface	22.34	0.53	11.25	13.68	18.21	0.42	20.66	0.92
1964	Surface	21.74	0.51	22.76	27.69	21.27	0.41	7.52	0.35
1965	Surface	13.21	0.31	11.89	14.47	16.05	0.44	3.87	0.29
1966	Surface	6.24	0.15	3.72	4.53	10.84	0.58	14.07	2.26
1967	Surface	5.16	0.12	4.81	5.86	15.15	0.73	4.51	0.88
1968	Surface	9.67	0.23	11.03	13.42	0.00	0.00	2.57	0.27
1969	Surface	12.24	0.29	10.47	12.73	0.00	0.00	10.65	0.87
1970	Surface	22.89	0.54	26.91	32.74	0.00	0.00	16.80	0.73
1971	Surface	39.69	0.94	36.21	44.05	0.00	0.00	21.31	0.54
1972	Surface	54.11	1.28	41.86	50.92	6.89	0.10	23.98	0.44
1973	Surface	59.79	1.41	19.48	23.70	18.30	0.24	25.32	0.42
1974	Surface	68.78	1.62	25.54	31.07	16.33	0.23	45.09	0.66
1975	Surface	87.76	2.07	49.15	59.79	26.11	0.28	20.39	0.23
1976	Surface	69.32	1.64	64.20	78.11	38.83	0.41	17.42	0.25
1977	Surface	56.69	1.34	58.68	71.39	30.04	0.37	12.23	0.22
1978	Surface	46.18	1.09	45.61	55.49	22.75	0.40	13.71	0.30
1979	Surface	41.20	0.97	66.40	80.78	18.69	0.35	10.81	0.26
1980	Surface	48.03	1.13	62.31	75.80	3.98	0.09	5.89	0.12
1981	Surface	45.82	1.08	52.01	63.28	8.09	0.16	-5.47	-0.12
1982	Surface	34.87	0.82	33.05	40.20	5.49	0.14	-0.45	-0.01
1983	Surface	25.84	0.61	16.77	20.40	8.58	0.22	9.56	0.37
1984	Surface	28.83	0.68	23.87	29.04	6.58	0.19	14.64	0.51
1985	Surface	43.29	1.02	30.01	36.51	0.18	0.00	6.65	0.15
1986	Surface	49.73	1.17	39.51	48.07	0.20	0.00	10.64	0.21
1987	Surface	44.43	1.05	16.86	20.51	15.93	0.27	22.85	0.51
1988	Dive	57.55	1.36	46.24	46.30	9.72	0.16	0.12	0.00
1989	Dive	44.39	1.05	47.72	47.78	13.29	0.24	3.41	0.08

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1990	Dive	37.95	0.90	46.46	46.52	9.85	0.18	3.17	0.08
1991	Dive	32.48	0.77	30.00	30.03	8.64	0.20	14.02	0.43
1992	Dive	42.79	1.01	42.37	42.42	3.71	0.08	4.60	0.11
1993	Dive	41.78	0.99	34.41	34.45	5.61	0.10	-1.15	-0.03
1994	Dive	34.59	0.82	25.25	25.28	6.04	0.13	-2.29	-0.07
1995	Dive	30.35	0.72	27.13	27.16	1.95	0.05	4.53	0.15
1996	Dive	34.08	0.80	33.12	33.16	0.79	0.02	14.11	0.41
1997	Dive	41.54	0.98	45.36	45.42	6.66	0.13	-5.97	-0.14
1998	Dive	28.58	0.68	41.01	41.06	6.98	0.20	-4.37	-0.15
1999	Dive	19.84	0.47	19.73	19.76	4.37	0.15	0.16	0.01
2000	Dive	18.37	0.43	12.80	12.82	1.63	0.08	2.08	0.11
2001	Dive	20.45	0.48	13.41	13.43	0.00	0.00	1.55	0.08
2002	Dive	21.18	0.50	21.24	21.27	0.82	0.05	0.09	0.00
2003	Dive	17.75	0.42	31.40	31.44	3.52	0.18	-3.52	-0.20
2004	Dive	9.78	0.23	16.43	16.45	4.45	0.28	-0.41	-0.04
2005	Dive	5.10	0.12	9.66	9.68	4.27	0.47	-0.16	-0.03
2006	Dive	4.95	0.12	2.88	2.88	0.00	0.00	-0.28	-0.06
2007	Dive	4.66	0.11	2.25	2.25	0.00	0.00	-0.51	-0.11
2008	Dive	4.15	0.10	2.74	2.74	0.00	0.00	0.27	0.06
2009	Dive	4.42	0.10	10.61	10.62	0.00	0.00	0.69	0.16
2010	Dive	5.11	0.12	2.46	2.47	0.00	0.00	1.37	0.27
2011	Dive	6.49	0.15	9.66	9.68	0.00	0.00	-0.11	-0.02
2012	Dive	6.37	0.15	5.41	5.41	0.00	0.00	1.49	0.23
2013	Dive	7.87	0.19	12.34	12.36	0.00	0.00	3.24	0.41
2014	Dive	11.11	0.26	13.94	13.95	0.00	0.00	3.65	0.33
2015	Dive	14.75	0.35	11.32	11.34	0.00	0.00	2.87	0.19
2016	Dive	17.62	0.42	20.53	20.55	0.00	0.00	*NA*	*NA*

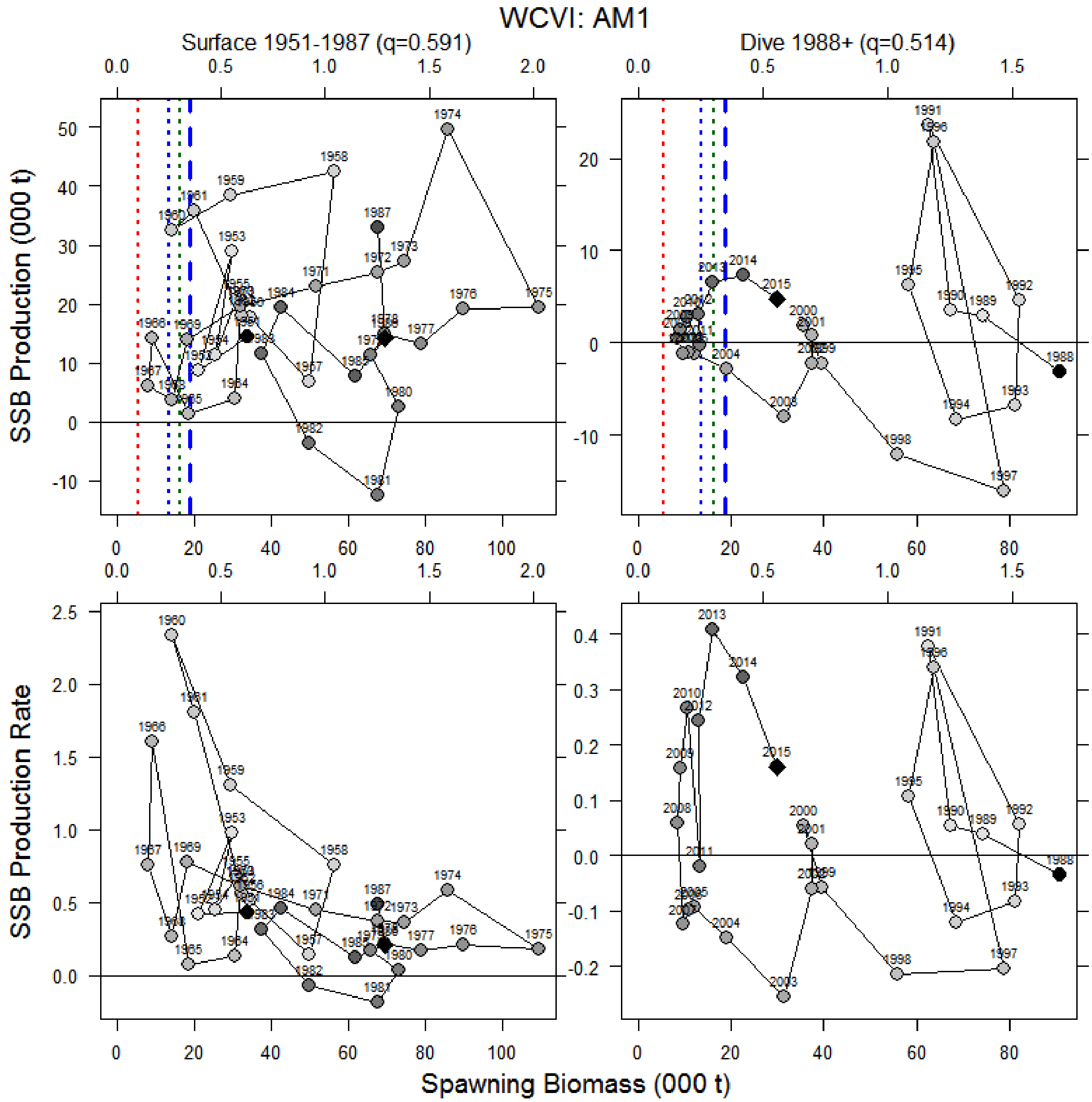

Figure 7. Phase plots of spawning stock biomass (SSB) production (upper panels) and SSB production rate (lower panels) against SSB for the WCVI management area based on model AM1. The time series is broken into two periods corresponding to surface (1951-1987, left panels) and dive (1988-2015, right panels) survey periods. The start and end of each time series is indicated by a black circle and black diamond, respectively. Grey shading of the circles becomes darker from in chronological order. Calendar years are indicated above each symbol. The axis scales at the top of each panel are in units of spawning biomass depletion, i.e., SSB divided by the estimated unfished spawning biomass (B_0) from the assessment model. Estimated values of surface and dive survey catchability parameters are reported above the left and right panels, respectively. Vertical reference lines are positioned at estimates of 0.1B_0, (red dashed line) 0.25B_0 (blue dashed line), 0.3B_0 (green dashed line), and the 1996 fixed cutoff value (thick blue long dashed line). Note that the axes are scaled independently to avoid visually compressing the phase pattern for the dive survey period.

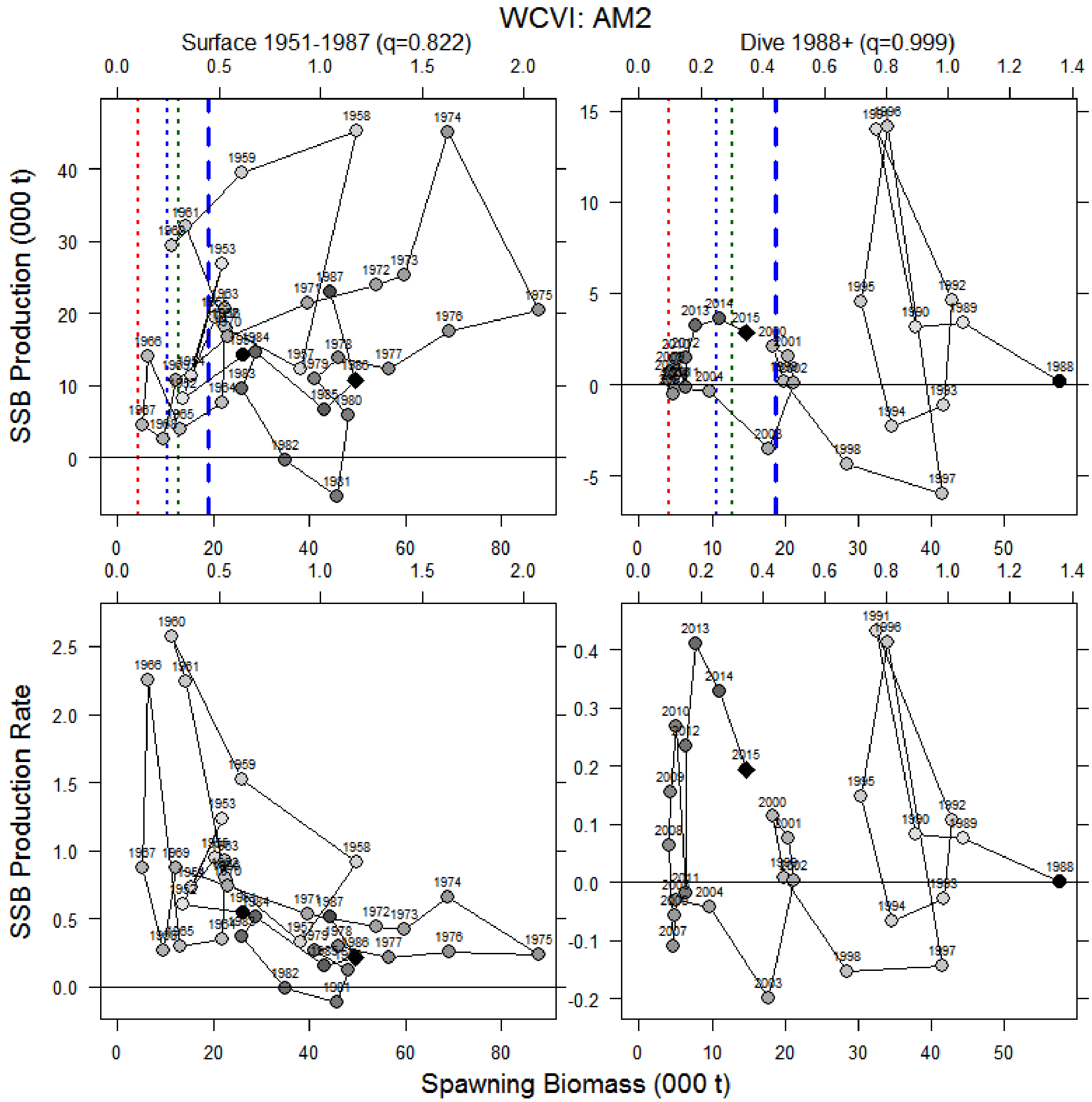

Figure 8. Phase plots of spawning stock biomass (SSB) production (upper panels) and SSB production rate (lower panels) against SSB for the WCVI management area based on model AM2. Description as for Figure 7.

Similar conclusions can be drawn from production phase plots based on model AM2, except that the depletion levels of years in the scale of the LP-LB state is lower, and extends from 2004 to 2014; spawning stock depletion levels from 2004 to 2014 range from 0.1 to 0.23 (Figure 8, Figure 4, Table 2).

Regardless of the model, the striking feature of the 50% decline from high spawning biomass in 1997, to a period of near zero production from 1999-2001, followed by a transition to a persistent low productivity, low biomass (LP-LB) state occurred rapidly, with each step occurring in 3 years. Fishery removals were comparatively modest during this period (Figure 3a) and the HCR cutoff fixed at the value established by the 1996 assessment (Schweigert et al. 1997) is now calculated at a spawning stock depletion level of 0.44 for model AM2. However, the decline in production that preceded the LP-LB state diagnosed for the mid-2000s to early 2010s was initiated as early as the late 1990s.

Figure 9 shows the production phase plots for models AM1 and AM2 scaled to emphasize the pattern at the lower range of biomass relative to model estimates of $0.1B_0$, $0.25B_0$, $0.3B_0$ and the fixed cutoff value estimated in 1996. Although similar spawning biomass depletion levels were reached during the stock collapse of the late 1960s, the production estimates are much higher, a large component of which was fishery removals. Nevertheless, in contrast to the recent period, the stock did not remain at low biomass for more than a few years after 1965 before transiting to higher biomass following substantial reductions in annual harvest rates and sustained positive production.

During the recent LP-LB period, the age-structure of the stock shows proportionally fewer age 6+ fish than at high biomass levels (Figure 10), however the number of fish that were aged decreased at the same time so there may be a confounding effect due to sample size which warrants future attention.

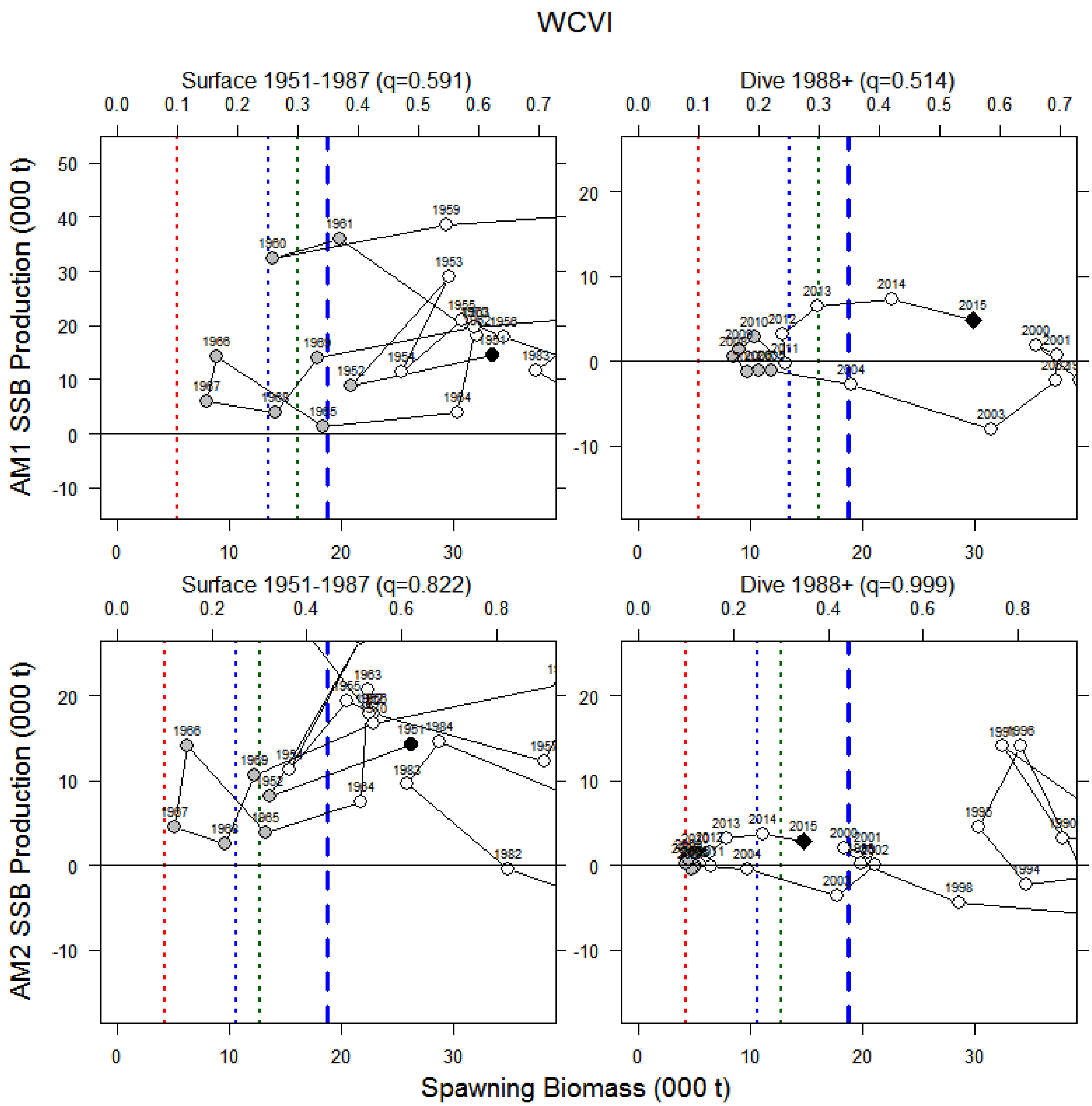

Figure 9. Phase plots of spawning stock biomass (SSB) production against SSB for model AM1 (upper panels) and model AM2 (lower panels) for the WCVI management area. Panels have been scaled to emphasize the relative positions of points in the lowest 20th percentile of SSB values (grey circles). The first (black circle) and last year (black diamond) of the surface survey (1951-1987) and dive survey periods (1988-2015) are indicated. Vertical reference lines are positioned at the model estimates of 0.1B_0 (red dotted), 0.25B_0 (blue dotted), 0.3B_0 (green dotted) and the fixed cutoff estimated in 1996 (thick blue dashed).

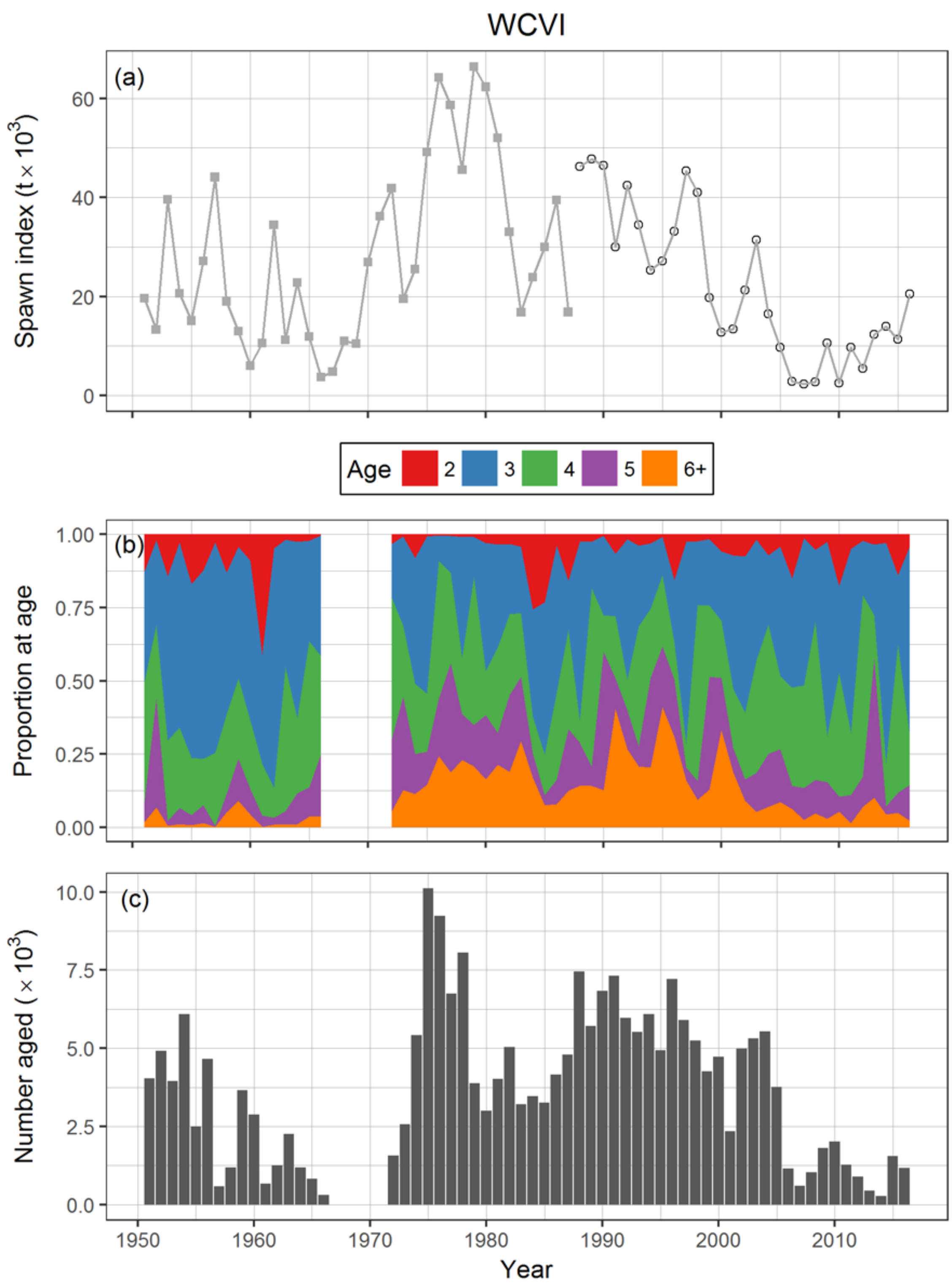

Figure 10. Time series of annual spawn index values for the surface (1951-1987, grey squares) and dive (1988-2016, open circles) survey periods (panel a) for the WCVI management area. Observed annual proportions at age for ages 2-5, and a plus group of ages 6 and older are shown in panel (b). Numbers of fish aged in each year are shown in panel (c). Panels (b) and (c) include biological samples from the seine roe and seine test fisheries only.

3.3 SUMMARY OF PRODUCTION ANALYSIS FOR ALL STOCKS

We visually inspected phase plots developed from models AM1 and AM2 for each major stock to diagnose periods of low biomass (LB) during the early period of the time series (approximately the late 1960s) when Pacific Herring stocks were considered to be collapsed (Hourston 1980). Similarly, we diagnosed low production and low biomass (LP-LB) states that coincided with recent declines of stocks in the CC, HG and WCVI management areas. The start and end years were determined by the first year of entry, and last year before a persistent exit from the LP-LB state. We interpreted the spawning biomass frontier (maximum spawning biomass depletion) of LP-LB states as a threshold for possible serious harm. Stocks in the PRD and SOG management areas did not show a recent LP-LB state. The phase plots are shown for the WCVI stock in the previous section and in Appendix C for other stocks.

Key results for stocks in all five major management areas are summarized in Table 3 for model AM1 and Table 4 for model AM2. For the early LB state, the maximum spawning stock depletion levels for all five major stocks ranged from 0.19 (HG) to 0.33 (WCVI) based on model AM1 and from 0.218 (PRD) to 0.289 (WCVI) for model AM2. For the CC, HG, and WCVI stocks the frontiers of the LP-LB states were estimated to be at spawning depletion levels of 0.244 (WCVI) to 0.328 (HG) for model AM1 and at 0.174 (CC) to 0.284 (HG) for model AM2. These levels are comparable to maximum depletion levels estimated for the early LB period. The LP-LB states persisted from about one (CC) to two Pacific Herring generations (HG, WCVI) where generation time was estimated at about five years by Cleary et al. (2010).

The transition into the LP-LB state was rapid, usually occurring within 3 years from relatively large spawning biomass levels and coincident with negative production values. Based on results from model AM1, the CC stock was estimated to be at a spawning depletion of 0.47 in 2003 when production became negative and entered the LP-LB state by 2006. Similarly, the HG stock declined from an estimated spawning depletion level of 0.78 in 1998 into the LP-LB state by 2000 (model AM1 results). Finally, the WCVI stock declined into the LP-LB state by 2005 from an estimated depletion level of 0.69 in 2002. For CC, HG and WCVI stocks, the transition was coincident with negative production values.

Table 3. Summary of key results for stocks in all five major management areas for model AM1. Visual inspection of phase plots was used to interpret persistent clusters of early low biomass (LB) or recent low production and low biomass (LP-LB) states. The year of entry and sustained exit from the state determined the year ranges. For stocks in the PRD and SOG management areas, a LP-LB state was not diagnosed. The number of years (n) and minimum, average, and maximum spawning biomass depletion (D) values are reported for LB and LP-LB states. The column C=0 indicates the number of years of 0 catch following entry to the recent LP-LB state for stocks in the CC, HG, and WCVI management areas. Depletion levels are reported for spawning biomass (000s t, min, avg, max) and depletion corresponding to 25% of the unfished biomass. The average estimated harvest rate (DFO 2016a) in years beginning in 1983 with positive catch is reported as U_{avg}.

	Early					Recent								
Stock	LB Range	n	$D_{\min}$	D_{avg}	$D_{\max}$	LP-LB Range	n	C=0	$D_{\min}$	D_{avg}	$D_{\max}$	$B_{0.25}$	$D_{0.25}$	U_{avg}
CC	1964-1969	6	0.126	0.194	0.260	2006-2011	6	6	0.195	0.245	0.282	14.348	0.250	0.12
HG	1965-1969	5	0.078	0.140	0.188	2000-2008	9	13	0.239	0.279	0.328	9.244	0.250	0.07
WCVI	1966-1969	4	0.149	0.228	0.333	2005-2012	8	10	0.158	0.200	0.244	13.462	0.250	0.09
PRD	1967-1972	6	0.169	0.208	0.238	na	-	-	-	-	-	13.400	0.250	0.18
SOG	1966-1969	4	0.119	0.172	0.227	na	-	-	-	-	-	36.600	0.250	0.13

Table 4. Summary of key results for stocks in all five major management areas for model AM2. Description as for Table 3. The fixed 1996 cutoff value and associated spawning stock depletion levels are reported as B_{Cutoff} and D_{Cutoff}.

	Early					Recent								
Stock	LB Range	n	$D_{\min}$	D_{avg}	$D_{\max}$	LP-LB Range	n	C=0	$D_{\min}$	D_{avg}	$D_{\max}$	B_{Cutoff}	D_{Cutoff}	U_{avg}
CC	1964-1969	6	0.126	0.184	0.256	2006-2011	6	6	0.126	0.159	0.174	17.600	0.345	0.17
HG	1965-1969	5	0.087	0.168	0.225	2000-2010	11	13	0.179	0.222	0.284	10.700	0.447	0.11
WCVI	1966-1969	4	0.121	0.197	0.289	2004-2014	11	10	0.098	0.150	0.262	18.800	0.444	0.15
PRD	1967-1972	6	0.154	0.191	0.218	na	-	-	-	-	-	12.100	0.227	0.20
SOG	1965-1970	6	0.077	0.167	0.252	na	-	-	-	-	-	21.200	0.192	0.22

3.4 EQUILIBRIUM LIMIT REFERENCE POINTS

Using MPD estimates of key model parameters (steepness, M, B_0, selectivity) from the respective AM1 stock assessments, and the associated growth and maturity schedules, we calculated the following equilibrium reference points for the five major BC Pacific Herring stocks: F_{ext}, F_{MSY}, F_{SPR40}, F_{SPR30} and $F_{0.1}$. We also calculated F_{rep} from the annual MPD estimates of spawning biomass (B) and recruits (R), by calculating the median value of R / B for each stock and numerically solving the inverse of equation E2 for the matching value of F. For each stock, we then calculated equilibrium spawning biomass (B) and equilibrium B / B_0 associated with each reference point. Therefore, all results are conditional on the model AM1 inputs and structural assumptions.

Resulting values of B / B_0 for each reference point are provided in Table 5 for each of the five major stocks (DFO 2016a). MPD estimates of spawning biomass and recruits, the stock-recruit curve and seven replacement lines are shown in Figure 11.

Recall that the equilibrium reference points are intended here as proxies for F_{rep}, i.e., to identify candidate LRPs (Mace and Sissenwine 1993). F_{rep} is shown as a black dashed line on each plot in Figure 10, indicating the median replacement line through the estimated stock-recruit estimates. Following this logic, one could reject candidate reference points that result in replacement lines far to the left of F_{rep}, i.e., replacement lines approaching F_{ext}`. For example, for most stocks, F_{SPR30} resulted in a replacement line with few recruitment estimates occurring to the left of the replacement line (Figure 11, purple lines). F_{SPR40} produced replacement lines (orange lines) closer to F_{rep} but still far from producing median replacement. For all stocks, the closest proxy for F_{rep} was F_{MSY} (green lines).

Table 5 shows the values of equilibrium B / B_0 associated with F_{rep}, F_{MSY}, F_{SPR40}, F_{SPR30} and $F_{0.1}$. Note that relative B_{rep} is high (> 0.6) in all stocks. This is mainly due to high variability in estimated recruitment, which led to many years of estimates of spawning biomass that were greater than B_0 (Figure 10), coupled with the assumption in the stock assessment that the stock was not at unfished equilibrium in the first modeled year (DFO 2016a). High relative biomasses at F_{rep} (and F_{MSY}), make interpretation of these reference points problematic because they suggest the stocks need to be maintained close to B_0 to maintain viability, where B_0 is, for some stocks, well below the maximum observed biomass.

Many of the high biomass, low recruitment observations occurred in the earlier part of the time series when the average biomass was greater than presently (Figure 3; Appendix A; Figure 11, lighter grey circles). This is possibly indicative of high compensatory density-dependence in juvenile or egg survival in these areas, evidenced by low recruitment productivity at high biomass and also some high observed recruitment events for stocks that experienced low biomass (notably CC, HG and WCVI; Figure 11). In general, strong density-dependence in recruitment tends to indicate more resilient stocks, as compensatory juvenile survival at lower stock size buffers against spawning stock decline. However, as noted in the previous section, the productivity regime for BC Pacific Herring appears to have shifted in recent years, with none of the CC, HG or WCVI stocks showing similar rates of recovery from low stock levels similar to those estimated after the collapses of the 1960s. This could be due to the increasing adult natural mortality seen in recent years (Figure 3; Appendix A; DFO 2016a), in which case the change in productivity would be driven by changes in the adult component of the population rather than recruitment. In this case, a LRP based on F_{rep} may not protect against persistent LP-LB states.

Another reason the equilibrium results are difficult to interpret is the high values of F associated with some reference points (Table 5). Estimates of F_{MSY} reported here and in Cox et al. (2015[1]) were among the highest produced for herring species world-wide (Table 4), partly due to the high value of M used in the analysis (long-term average of the time series of M), and partly due to the juxtaposition of the maturity and selectivity-at-age schedules (Figure 12). For all stocks, 50% maturity is estimated to occur at a much younger age than 50% selectivity, which essentially guarantees a large component of the Pacific Herring population can spawn at least once before becoming vulnerable to the fishery and allows for much higher sustainable harvest rates. This is markedly so for the HG stock (Figure 12), evidenced by the highest estimate of F_{MSY} among all stocks (Table 6). This is a counter-intuitive result, since the HG stock has not recovered and does not seem resilient to any fishing pressure currently. We recommend review of the maturity-at-age schedule, which has not been updated for many years, especially given the reductions in weight-at-age that have occurred since 1990 (Figure 3 and Appendix A). We also note that model estimates of selectivity are conditional on other structural assumptions, notably the representation of time-varying natural mortality.

Finally, we have sufficient concerns about the impacts of non-stationary growth and mortality on interpretation of equilibrium reference points based on F_{rep} or proxies that we do not recommend using them for BC Pacific Herring without simulation testing. In our analyses, we used the average MPD values of M_t and weight-at-age averaged across the whole time series. To our knowledge, there are no widely-accepted best-practice recommendations for how to select a period for averaging time-varying model parameters to use in reference point calculations, when model parameters are known to vary over time. Selecting a more recent period for averaging is commonly done (e.g., DFO 2016a). However, in the case of BC Pacific Herring stocks, the impacts of including very high recent estimates of M into equilibrium calculations when other related parameters (steepness, B_0, selectivity) are not allowed to vary over time are unclear at best and may make the stock seem even more productive. We suggest simulation testing is the only reasonable approach for evaluating the consequences of equilibrium reference points based on F_{rep} or proxies for BC Pacific Herring stocks. Extension of the work presented by Cox et al. (2015[1]) is recommended.

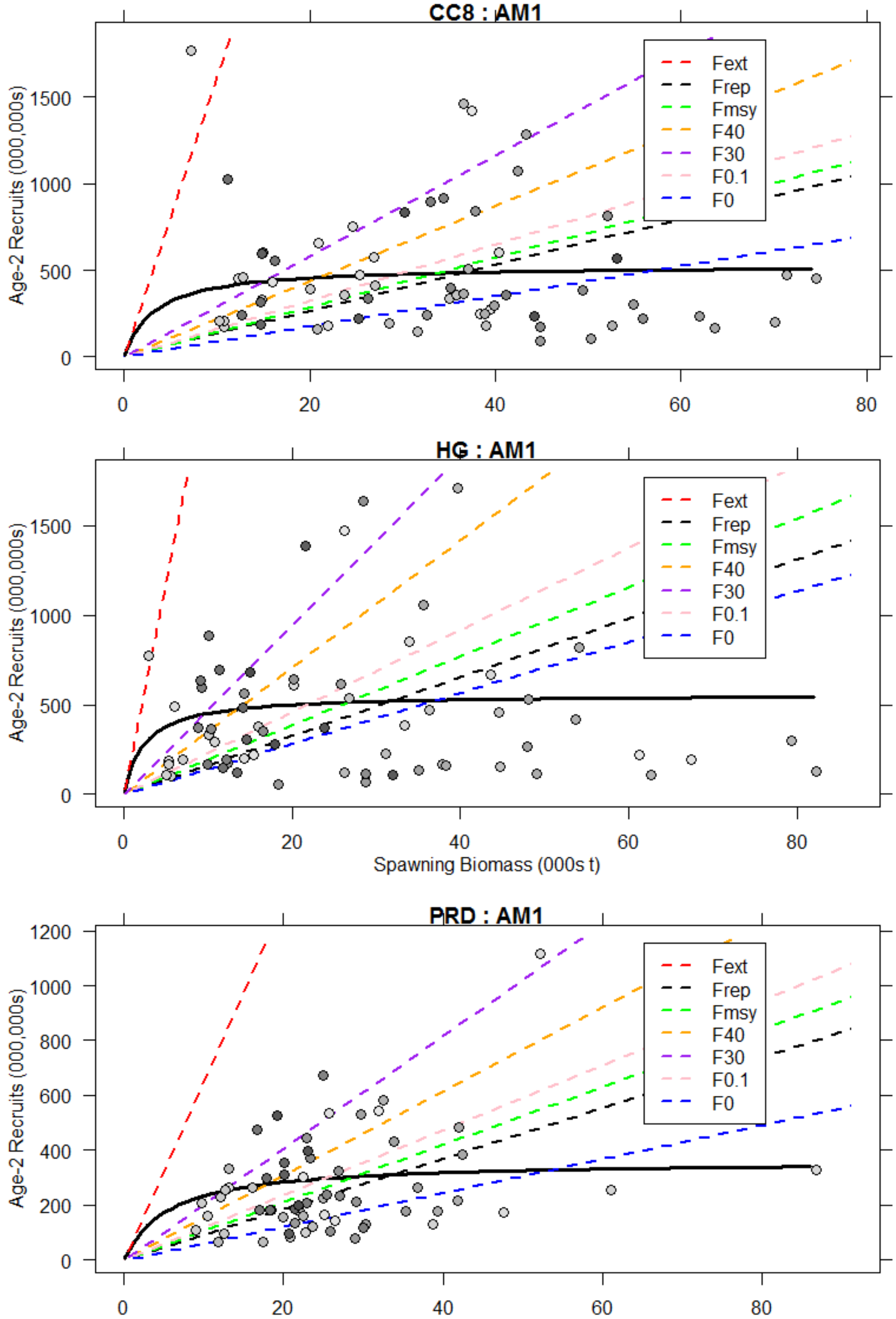

Figure 11. Stock-recruit curves, MPD estimates and replacement lines associated with F_{ext}, F_{rep}, F_{MSY}, F_{SPR40}, F_{SPR30}, $F_{0.1}$ and F_0 for the five major stocks (see text and Figure 1 caption).

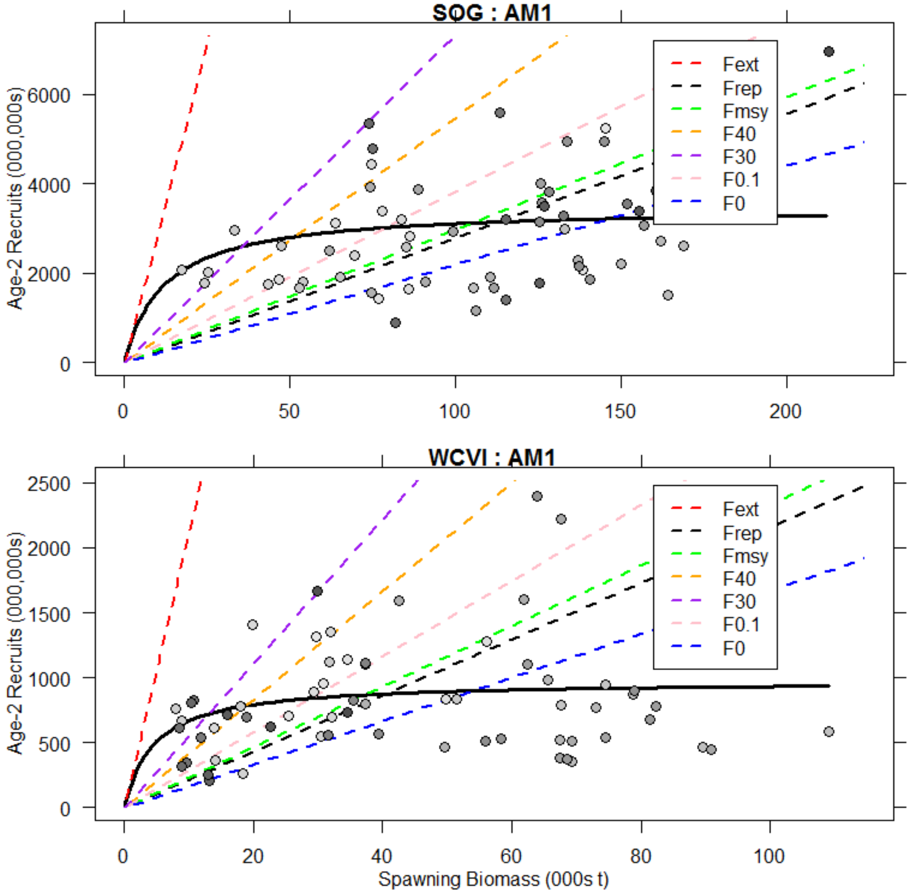

Figure 11 (cont.). Stock-recruit curves, MPD estimates and replacement lines associated with F_{ext}, F_{rep}, F_{MSY}, F_{SPR40}, F_{SPR30}, $F_{0.1}$ and F_0 for the five major stocks (see text and Figure 1 caption).

Table 5. Summary of key B / B_0 results corresponding to equilibrium fishing mortality rates for all management areas (see text). Equilibrium B / B_0 for F_{ext} is by definition zero for all stocks and is not shown. Similarly equilibrium B / B_0 for F=0 is by definition 1 for all stocks and is not shown. Subscripts indicate the equilibrium F rates (i.e., $B_{0.1}$ = equilibrium biomass when $F = F_{0.1}$).

	B_{rep}	B_{MSY}	B_{SPR40}	B_{SPR30}	$B_{0.1}$
CC	0.639	0.586	0.366	0.260	0.509
HG	0.860	0.720	0.362	0.255	0.597
PRD	0.630	0.540	0.338	0.227	0.470
SOG	0.771	0.719	0.350	0.242	0.540
WCVI	0.751	0.692	0.349	0.240	0.535

Table 6. Summary of key equilibrium fishing mortality rate results for all management areas (see text). $F > 4$ essentially implies U approaching 1, where harvest rate U and instantaneous fishing mortality rate F are related by $U = 1 - e^{-F}$. This implies that virtually all of the vulnerable biomass can be harvested because all fish have had a chance to spawn at least once before being vulnerable to harvest. This is a possible artefact of the structural assumptions of the model, affecting estimates of selectivity, or of the method to estimate maturity at age.

	F_{rep}	F_{MSY}	F_{SPR40}	F_{SPR30}	$F_{0.1}$
CC	0.41	0.54	2.02	> 4	0.82
HG	0.26	0.78	> 4	> 4	1.74
PRD	0.30	0.45	1.33	2.89	0.64
SOG	0.39	0.55	> 4	> 4	1.62
WCVI	0.39	0.56	> 4	> 4	1.42

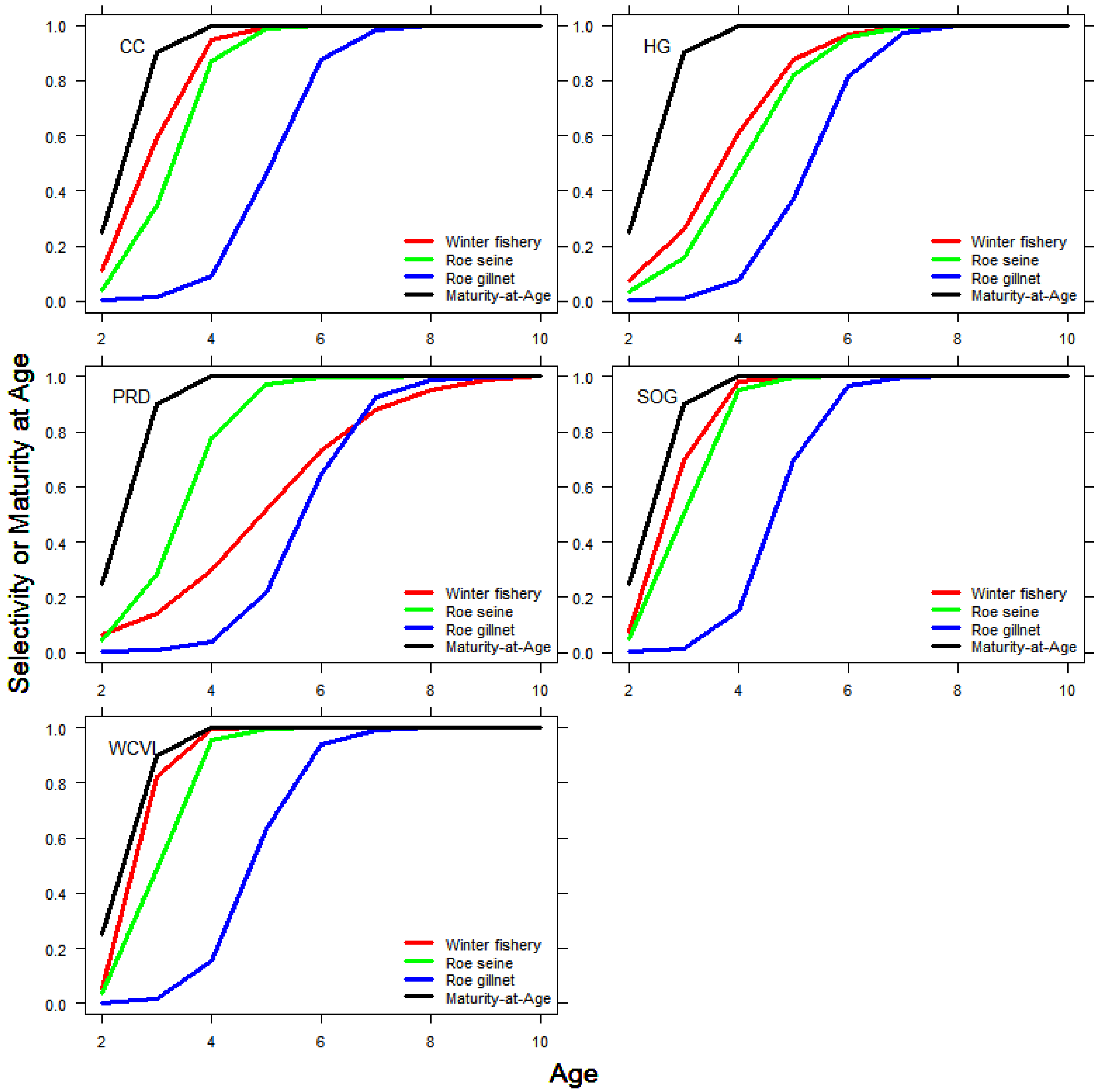

Figure 12. Maturity and MPD selectivity schedules for the five major stocks of BC Pacific Herring from AM1 (DFO 2016a). For each stock the maturity ogive (black) is far to the left of the roe seine selectivity ogive (green), the selectivity used in this study and Cox et al. 2015[1]. This implies that all fish have had a chance to spawn at least once before becoming vulnerable to the fishery.

4 DISCUSSION

4.1 LRP CHOICES FOR BC PACIFIC HERRING

Claims of fisheries sustainability in many management jurisdictions including Canada require the identification of limit and target reference points. Limit reference points are thresholds intended to prevent states of possible serious harm which is generally interpreted in a single-stock context to mean recruitment overfishing or the occurrence of depensatory effects. Although there is no precise definition of these states, we conducted a production analysis to evaluate whether stocks showed persistent states of low productivity and low spawning biomass that resulted in loss of benefits to resource users. We also estimated a suite of equilibrium reference fishing mortality rates based on the concept of replacement fishing mortality, F_{rep}, and spawning potential ratio proxies for F_{rep} that could potentially serve as limit fishing rates and imply limit spawning biomass thresholds.

The major stocks of BC Pacific Herring simultaneously declined to historically low spawning biomass levels in the late 1960s and the CC, HG, and WCVI stocks declined to similar low biomass levels by the mid-2000s. Our analyses show that the characteristics of the stock dynamics differed between these periods. For example, the PRD and SOG stocks also declined by the late 2000s, but not to levels estimated for the late 1960s. Furthermore, the SOG stock has since increased to an estimated historic high level of spawning biomass. Key conclusions from the production analysis include:

1. Surplus production estimates trended to negative average levels (WCVI) or near zero levels (CC, HG) in advance of the decline in spawning biomass of the mid-2000s;
2. The loss of production occurred at relatively high (above average) levels of spawning biomass;
3. The decline in spawning biomass was preceded by declines in observed weight at age and increasing estimates of natural mortality that began about 1990;
4. The transition to a LP-LB state for the CC, HG and WCVI stocks was rapid, occurring in 3 years or less than one Pacific Herring generation;
5. The low biomass state of the late 1960s was not associated with persistent low productivity;
6. The LP-LB state persisted for 6 years (CC, AM1) to 9 years (HG; AM1), and 6 (CC, AM2) to 11 (HG and WCVI, AM2) years despite large reduction or cessation of commercial catches;
7. The estimated harvest rates are on average, less than the target harvest rate of 20% (model AM1, CC, HG and WCVI for model AM2) or about 0.2 (PRD and SOG, model AM2) and are much less than estimated harvest rates during the 1960s;
8. Estimated spawning biomass depletion levels averaged about $0.25B_0$ during the LP-LB period for model AM1 and were less than $0.25B_0$ for model AM2 for the CC, HG and WCVI stocks;
9. The PRD stock showed a modest decline in the mid-2000s to about $0.3B_0$ but does not show a persistent LP-LB state; and
10. The SOG stock declined by more than 50% from 2000 to 2008-2010 to 0.4 or $0.5B_0$ (model AM1 and AM2, respectively) but has since increased to a historical high level of estimated spawning biomass and does not show a persistent LP-LB state.

The Pacific Herring fishery in British Columbia is noteworthy among Canadian fisheries because of the introduction of a target harvest rate of 20% of the forecast pre-fishery spawning biomass in 1983 and the addition of the cutoff at the estimate of $0.25B_0$ in 1986. These measures resulted in one of the earliest Canadian examples of a feedback HCR that identified a level at which harvest would be curtailed as spawning biomass declined. Based on Stocker et al. (1983), Haist et al. (1986) and Stocker (1993) the intended goals of these changes were to:

1. Establish a conservative level of harvest that would also encourage catch stability;
2. Minimize the occurrence of very low biomass levels where reproductive potential may be severely limited; and
3. Allow for rapid stock increases by preserving a productive level of spawning biomass.

These goals were reasonable at the time the HCR was introduced, particularly given the recovery of Pacific Herring stocks in BC following the collapse of the late 1960s that was largely attributable to fishing mortality (Hourston 1980). Commercial catches at that time were minor (SOG) or zero for up to four years (WCVI) depending on the stock. However, stocks in most of the major management areas recovered to historical highs by the early 1970s and commercial fishing was resumed, albeit at much reduced harvest rates in comparison to levels prior to the collapse. This outcome suggested that BC Pacific Herring stocks can be expected to recover rapidly if sufficient spawning biomass is preserved, e.g., 25% of the estimated unfished spawning biomass. Simulation studies conducted in the late 1980s and early 1990s indicated HCR performance consistent with the goals (1-3) above (Haist et al. 1986, Haist et al. 1993; Hall et al. 1988; Zheng et al. 1993). However, the decline of the CC, HG and WCVI stocks to a persistent LP-LB state after almost 20 years of application of the HCR suggests that whatever conditions allowed the recovery of the early 1970s did not prevail during the 2000s.

In practice the management procedure had changed over time due to changes in the assessment models and forecasting methodologies (see DFO 2015a) but the concept of preserving a threshold biomass and applying a 20% harvest rate has remained consistent. Modifications were made to the assessment model to acknowledge time-varying processes related to size-at-age and to improve statistical fit by allowing time-varying natural mortality. However, the goal of preserving an escapement of $0.25B_0$ as a productive base for rapid recovery was not achieved for the CC, HG and WCVI stocks, due in part to stock assessment and forecasting uncertainty common to all assessments, and potentially by treating the cutoff as an OCP that triggered cessation of commercial fishing rather than a level to avoid breaching with high probability.

A potentially troubling aspect of the decline to a persistent LP-LB state in the CC, HG and WCVI stocks is that, in all cases, it occurred rapidly from average, or above average, levels of spawning biomass to a state of persistently negative or low production. This feature was noted by Essington et al. (2015) in their meta-analysis of forage fish populations. Regardless of whether the transition could be predicted, the subsequent rapid decline of spawning biomass means that a LP-LB state can be reached within a Pacific Herring generation under the current HCR. It is not clear whether preserving spawning biomass levels higher than those estimated for the CC, HG, and WCVI for the mid-2000s to 2010s would have shortened the duration of the LP-LB state, or whether a LP-LB state could have been avoided by implementing harvest rate reductions at a higher estimated biomass level than the cutoff values. It is possible that the original premise of the harvest strategy could still be applicable if it was configured to reflect the conclusion that conditions today are not the same as those when the harvest strategy was originally developed (see Implementing Limit Reference Points section below).

Cox et al. (2015[1]) identified three categories of LRPs: model-based equilibrium LRPs, dynamic LRPs, and historical LRPs (Appendix G, Table 15). Their choices for candidate LRPs were not based on explicit consideration of serious harm, but were chosen because of historical reasons and best practice recommendations. Their closed-loop simulation analysis demonstrated how different LRP choices could help to discriminate among alternative management procedures. They recommended that work to identify LRPs for BC Pacific Herring should focus on fixed equilibrium reference points related to biomass and concluded that LRPs that track the dynamics of natural mortality and growth, or use limit fishing mortality rates based on F_{MSY}, were of little value. This recommendation may not have been expected *a priori*, given concerns about non-stationary growth and natural mortality processes, but was borne out by simulation results. Dynamic reference points were found to be vulnerable to reduction of the conservation threshold to progressively lower levels as stock biomass declined. They also concluded that historical LRPs based on the minimum estimated biomass in the reconstruction appeared to be too low for stocks in the PRD and SOG management areas or for populations that had decreased to low biomass (e.g., CC, WCVI).

Cox et al. (2015[1]) considered biomass-based equilibrium reference points of 0.25, 0.3 and $0.4B_0$. We estimated the upper biomass bound of a persistent LP-LB state for stocks in the CC, HG and WCVI management areas by plotting production based on spawning stock biomass as times series and as phase plots against estimated spawning biomass for both models AM1 (Table 3) and AM2 (Table 4). We interpreted a persistent LP-LB state to be consistent with signs of serious harm and therefore focussed on the maximum spawning biomass depletion levels within the LP-LB period. Stocks in the CC, HG, and WCVI management areas showed evidence of a persistent LP-LB state beginning by the mid-2000s, while stocks in the PRD and SOG management areas did not. Estimates of maximum B / B_0 within the LP-LB period ranged from 0.24 (WCVI) to 0.33 (HG) for model AM1 and 0.17 (CC) to 0.28 (HG) for model AM2. We note that there is no guarantee that stocks that transition to a persistent LP-LB state will always settle at the same level. For example, a given stock could persist in a LP-LB state at lower or higher levels than those estimated for the CC, HG, and WCVI stocks from the mid-2000s to mid-2010s.

Our examination of equilibrium-based reference points attempted to quantify recruitment overfishing (serious harm) based on estimating F_{rep} and associated proxies. Results reinforce the conclusions of Cox et al. (2015[1]) with respect to F-based limit fishing rates. Estimated F-based reference points were high, largely due to the value of M and juxtaposition of the selectivity and maturity schedules. Furthermore, in many instances, the results suggested that stocks need to be maintained close to B_0 to maintain viability, where the MPD estimate of B_0 is, for some stocks, well below the maximum observed biomass due to high recruitment variability and the assumption of non-equilibrium starting conditions in the stock assessment. Visual examination of the stock recruit curves for the CC, HG and WCVI stocks suggest a productive stock-recruit relationship exhibiting evidence of compensatory density-dependence (low recruitment at high stock size and vice versa). This seems counter-intuitive since these three stocks have entered a LP-LB state that persisted even in the absence of fishing pressure. We suggest that the LP-LB states for these stocks may not be driven by recruitment dynamics, and are being driven by mortality in the adult population. Indeed, this is how the recent stock assessments have interpreted the data, estimating increasing trends in adult natural mortality for the past decade (DFO 2016a). This suggests that focusing development of LRPs on stock-recruit dynamics may not be appropriate for BC Pacific Herring stocks, as other, non-stationary mechanisms are occurring in the adult population. Furthermore, non-stationary adult mortality and growth violate the assumptions of equilibrium fishing rate analyses, and there are no clear guidelines on how to account for these effects in the calculation of reference points. Since many of the structural uncertainties are related to non-stationary processes, it is difficult to support equilibrium-based BRPs based on yield-per-recruit, SPR or MSY for Pacific Herring, as stated in general by Hilborn (2002) and Hilborn and Stokes (2010).

Experience with the current harvest strategy since 1986 indicates that persistent LP-LB states can occur even when target harvest rates are set at, or below, 0.2 of the forecasted spawning biomass (e.g., the stocks in the CC, HG and WCVI management areas). For these stocks, the estimated harvest rates since 1983 are generally lower than $U \leq 0.225$ and much lower than the estimates of equilibrium fishing rate reference points we obtained. Schweigert et al. (1997) suggested that the harvest rate of 0.2 may prevent long-term declines but might not allow rebuilding of severely depleted stocks, an outcome that would be exacerbated by persistent periods of low production.

There are precedents in groundfish stock assessments for using historical minimum biomass-based reference points as LRPs to be avoided. Forrest et al. (2015) and Holt et al. (2016) used

LRPs based on the lowest estimated biomass from which the stock recovered to an above average level for Pacific Cod and Rock Sole, respectively. For these studies, there was no evaluation of whether lowest estimated biomass represented a state of serious harm; the lowest biomass was simply agreed to be an undesirable level of abundance. For BC Pacific Herring, we suggest the surplus production analysis shown here indicates a change in productivity in the recent period for at least the CC, HG and WCVI stocks, evidenced by a recent prolonged period in a LP-LB state. Therefore historical low biomass levels do not provide assurances of recovery for these stocks.

Cox et al. (2015[1]) found the estimated minimum biomass to be too low as an LRP for stocks recovering from low levels because this LRP choice failed to invoke management actions at abundance levels where risks were significant. Our analysis of surplus production showed that it is possible to transit to negative surplus production from relatively high biomass levels and rapidly attain persistently low spawning biomass depletion near historical minimums, below the intended threshold biomass level, and associated with persistent low productivity. Despite this result, historical biomass-based LRPs may be candidates provided they are selected to be above the level where there are signs of serious harm rather than at a historical minimum.

Interactions with predatory fish and marine mammals, possible loss of spatial diversity of spawn, climate effects, and reduced size-at-age leading to lower fecundity are possible factors that suggest lows from which the stock has recovered are not necessarily failsafe limits. Schweigert et al. (2010) argued there may be strong potential for increasing natural mortality due to increasing predation pressure on Pacific Herring along with potential changes in oceanographic regimes. Cox et al. (2015[1]) expressed concerns that a LRP of $0.25B_0$ may in future leave inadequate spawning biomass to service ecosystem requirements (Pikitch et al. 2012; Tyrrell et al. 2011).

Although we agree these are important considerations, we make no recommendations here because operating models to represent hypotheses regarding predator-prey dynamics or environmental drivers have not been developed for BC Pacific Herring. Furthermore, we are not aware of LRPs for forage fish that have been developed in consideration of dependent species that have been demonstrated through use to be effective; which is a criterion for establishing best practice (Sainsbury 2008). Hilborn et al. (2017) concluded that the impact on predators caused by fishing on forage species was generally less than indicated from trophic models, and that there is little evidence for strong connections between forage fish abundance and their predators for a range of US fisheries. They argued that any evaluation of harvest policies for forage fish needs to include these issues, and that models specific to individual species and ecosystems are needed to inform fisheries decision-making and policy.

4.2 IMPLEMENTING LIMIT REFERENCE POINTS FOR PACIFIC HERRING

Sustainable fisheries are defined by the procedural steps needed to establish a management system that accounts for catches, measures (relative) abundance, establishes rules about how to change catch in response to information about the stock, and enforces the changes in catch (FAO 1995; Hilborn 2002). Sustainability means being able to maintain a specified level of practical and effective use of a resource over the long-term, and defending such claims means that the management strategy meets current standards of acceptable scientific and management practice. Scientific defensibility of a choice of data, stock assessment method, and harvest control rule requires a systematic approach to defining objectives, investing in stock and fishery monitoring data, and responding to the results of new information and analyses. Defending fishery management plans requires demonstration that specific management measures can be expected to provide an acceptable trade-off between conservation, economic and socio-cultural benefits.

The choice of procedural steps is dependent on:

1. Defining a set of measurable objectives that correspond to both ecological and non-ecological sustainability goals;
2. Identifying performance measures for each objective that can be used to quantify how well each objective is met; and
3. Defining alternative management procedures that specify which choices of data, assessment methods and harvest control rules that are feasible for implementation.

Fishery managers and resource users can collaborate on steps (1-3). Science has a role in assisting this process since ultimately scientific data and methods will be used to evaluate the expected performance of proposed management procedures (de la Mare 1996). However, science has no means of coping with management uncertainty and, therefore, fisheries policy and decision-makers must specify acceptable levels of risk aversion to deviating from limits and targets. Where LRPs are available, their practical application is to use them as thresholds in measurable conservation objectives that discriminate which management options can satisfy the imperative goal of avoiding the possibility of serious harm (Cox et al. 2015[1]; Miller and Shelton 2010).

Science has a larger role in conducting prospective evaluation of proposed management procedures by:

4. Developing a set of operating models that represent alternative views (hypotheses) of stock and fishery dynamics that are used to generate realistic stock and fishery monitoring data consistent with each hypothesis;
5. Conducting simulation evaluation to "test drive" each proposed management procedure against data generated by each operating model to gather performance measures that can be used to rank the management procedures.

Finally, fishery managers and resource users need to participate in decision-making steps by:

6. Evaluating the trade-offs in management outcomes that result from application of each procedure to identify those management procedures that provide acceptable outcomes; and
7. Selecting and consistently applying the preferred management procedure.

These steps were proposed specifically for Pacific Herring in BC as a means of implementing the strategic stream (Landmark Fisheries Ltd. 2016[3]) and their application illustrated by Cleary et al. (2010) and Cox et al. (2015[1]). The steps (5-6) are in keeping with the FAO (1995) requirement for prospective simulation evaluation of proposed management options. The premise of the simulation evaluation is that management procedures that do not perform well in simulation are unlikely to perform well in actual application and can therefore be eliminated from further consideration. The DFO PA Framework identifies a need for evaluation "*… on a regular basis and it would normally take place after there is sufficient experience with the framework to conduct a proper evaluation of its performance (a period of 6 -10 years might provide enough time to gain appropriate experience with the framework).*" This implies testing of a management procedure on a real stock and fishery rather than simulation evaluation of alternative

[3] Cox, S.P. and Benson, A.J. 2016. Roadmap to more sustainable Pacific herring fisheries in Canada: a step-by-step guide to the management strategy evaluation approach. Landmark Fisheries Research. Unpublished report.

management procedures against a range of plausible uncertainty. Although prospective evaluation is not a guarantee of success, it is less risk-prone than testing the effectiveness of a management procedure on a real fish stock and fishery. This is because a range of uncertainties has been accounted for in a prospective analysis, as per DFO PA Framework element PA3. However, review of realized management procedure performance against the performance predicted by simulation after a period of application could be considered part of an ongoing iterative process specified by steps (1-7).

Goals can be translated to measurable objectives in step (1) by stating an acceptable probability of achieving the desired outcome (e.g., maintain spawning biomass above a LRP) and specifying the time period for evaluation (e.g., several Pacific Herring generations). The embedding of (limit) reference points in measurable objectives is a key step to understanding their role and also rendering them relevant to decision-making. Since an LRP must be avoided with high probability (e.g. 95%, DFO PA Framework, Annex Table 2B), they are distinguished from operational control points (OCPs), which are components of HCRs that define when management actions are taken to achieve the objectives (e.g., reducing fishing mortality) (Cox et al. 2013). This distinction is particularly important when reviewing analyses of the interactions between reference points and management options such as those conducted by Cleary et al. (2010) and Cox et al. (2015[1]), whose studies were concerned with evaluating the relative performances of MPs that differed in the choice of OCPs. Pacific Herring in BC is a particularly good example of the need to separate reference points from OCPs. Although the cutoff, first introduced in 1986 (Stocker 1993; Schweigert and Ware 1995[2]) to address biological goals is sometimes referred to in the Pacific Herring literature as a LRP, it has been applied in practice as an OCP.

The DFO PA Framework policy is not prescriptive on the time frame for evaluation except in cases where the need for stock rebuilding has been determined. In such instances 1.5 to 2 generations is suggested (Footnote 12 of the DFO PA Framework). The one-year forecast used by the current Pacific Herring management procedure for setting a TAC is too short a time to evaluate the effects of a specific choice of management actions relative to the generation time of Pacific Herring. For example, our analysis shows the persistence of average negative recruitment deviations for the WCVI stock assessment for a decade or more, increasing trends in natural mortality for some stocks since about 1990, and declining trends in weight-at-age over the same period. This suggests simulation-evaluation time frames of at least four to six generations to increase the detectability of undesirable transitory effects due to:

a) Lags in estimated stock dynamics that would not be apparent over a shorter time period (e.g., persistent over-estimation of stock abundance when in fact the true stock has changed trajectory and is in fact declining); and

b) Lags in management effects due to the time between when a management action is applied and recruitment of fish from that spawning biomass to the fishery.

For example, Cleary et al. (2010) used time frames of 10, 20, and 30 years for short-, mid- and long-term evaluations of HCR performance corresponding to 2, 4, and 6 Pacific Herring generations. Cox et al. (2015[1]) used a 20-year projection period (4 generations) for evaluating the relative performance of four management procedures against four hypotheses about trends in natural mortality. Furthermore, Hall et al. (1988) noted that the stock-recruitment relationship is a highly uncertain process that is best viewed as a probability distribution of recruitment for each level of spawning biomass. Thus, it is a description of how average recruitment changes with spawning biomass, and is therefore useful for characterizing the long-term response to candidate management procedures rather than for short-term prediction.

Performance measures in step (2) are needed for each objective to quantify how well each objective is met during the evaluation steps (5-6). Step (3) specifies the identification of candidate management procedures that are feasible to implement. One component of a management procedure is the HCR. Figure 13 illustrates a sequence of potential adjustments to the HCR that might be required to ensure that acceptable management outcomes are achieved against a given set of objectives including those that involve BRPs (Figure 13b). Deferring management action until the estimated stock size reaches the biomass at B_{LRP} poses more than a small risk of impaired production due to uncertainty in stock dynamics, assessment estimates, or variation in management effectiveness in controlling the catch. For example, the true stock status may be lower than estimated and the true B_{LRP} may be higher than estimated. Consequently, a precautionary approach requires actions to reduce harvest rate and increase the likelihood of stock growth *well before the* B_{LRP} *is reached* (Figure 13c).

Policy and management may take social and economic considerations into account in specifying the level of risk aversion and time horizon to be applied, but once the level of risk aversion is specified, the OCP where fishing is curtailed is chosen on the basis of performance measures identified in step (2). The current Pacific Herring HCR is conceptually most like Figure 13d; however, alternatives to the current HCR may be required if acceptable performance is not achieved with the current management procedure. For example, harvest rates may need to be reduced at a higher level than $0.31B_0$ to maintain spawning biomass above B_{LRP} and avoid commercial fishery closures leading to HCRs of the form represented in Figure 13e. Alternatively, the target harvest rate could be reduced to achieve similar outcomes.

Although LRPs are needed to help avoid biomass levels that potentially represent states of serious harm, they do not necessarily identify states that fulfill requirements for dependent species, socio-cultural needs, or economic opportunity. Therefore, a focus on LRPs places emphasis on conservation of a minimum biomass, as opposed to sustainable utilization that represents the aspirations of resource users or outcomes of ecosystem interest. Achieving objectives constructed around the target reference points that represent desirable states of the stock and fishery will produce trade-offs in outcomes related to conservation objectives defined using LRPs. Incomplete specification of elements (PA1, PA3) means that the evaluation required by element (PA4) cannot be fully met until the structured decision-making process proposed for renewal of the Pacific Herring management system matures. Thus, the efficacy of alternative management procedures in avoiding a LRP cannot be evaluated in isolation of other strategic components of the management system. Furthermore, LRPs in isolation are not the most important element of a management strategy. For example, Punt et al. (2008) demonstrated that the yield performance of three different management strategies that used threshold reference points were broadly insensitive to the actual thresholds. The key feature that produced acceptable performance of the management system was that catches were reduced as perceived stock size declined. In other words, the feedback control of catches in response to changes in stock abundance through consistent application of the management steps led to successful outcomes.

Consideration of step (7) of the process certainly requires consistent application of the selected MP to establish feedback control. However, moving from the current state of management to feedback control may require a phase-in (Punt et al. 2016) of the selected management procedure to achieve an acceptable trade-off of conservation, economic and socio-cultural outcomes.

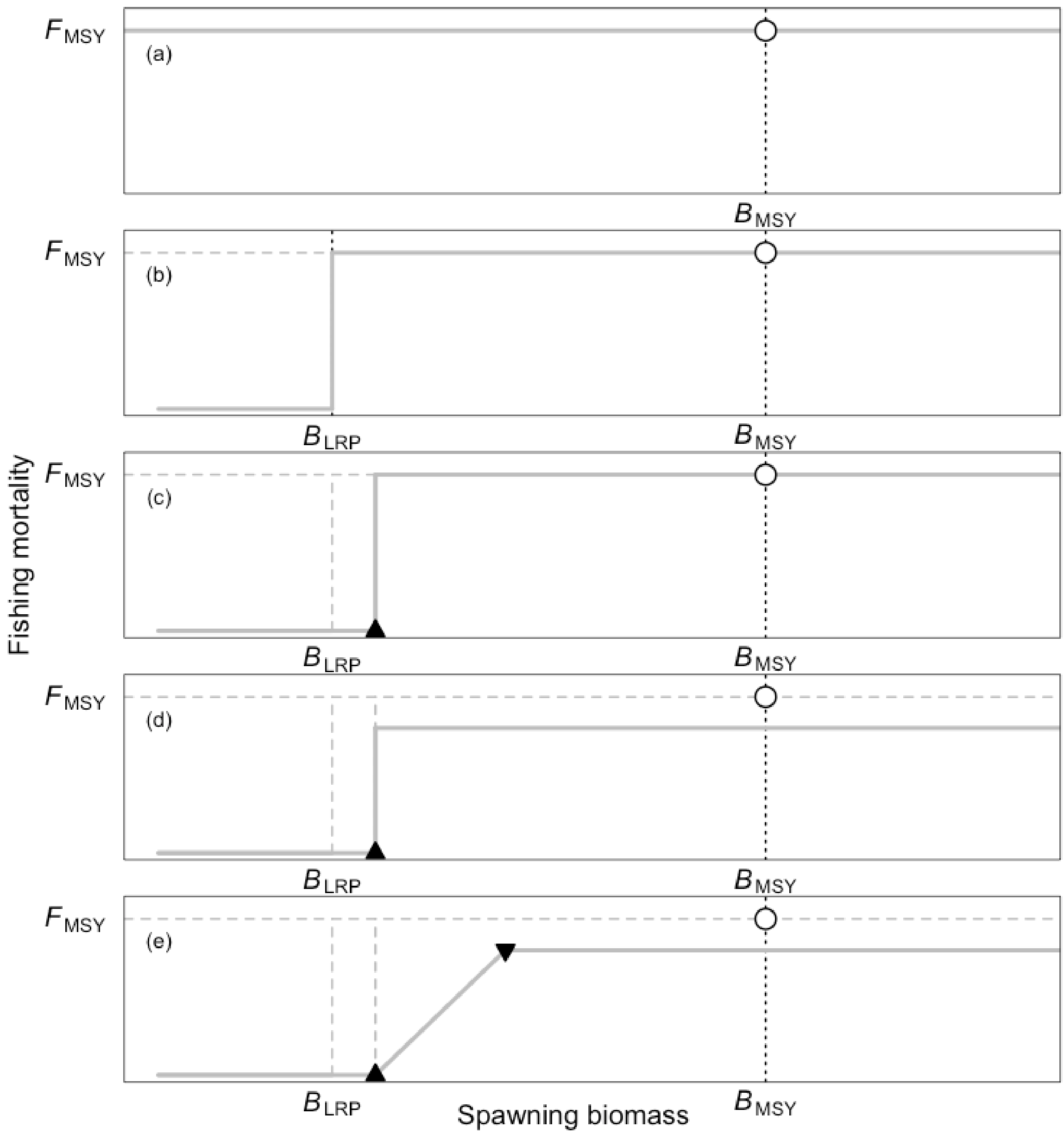

Figure 13. Separation of BRPs and operational control points (OCPs) in the design of a DFO PA Framework harvest control rule (HCR). International and domestic fisheries policy state that F_{MSY} is a limit fishing mortality rate and a biomass level of at least B_{MSY} is desirable. Fishing at F_{MSY} produces B_{MSY} under deterministic equilibrium conditions (thick grey line, panel a). Uncertain stock and fishery dynamics mean that adjustments to the HCR are needed to encourage desirable states and avoid deleterious states (panels b-e). A biomass-based limit at B_{LRP} is positioned above level where serious harm is a possibility and fishing mortality is set to 0 below this level (panel b). An OCP (black triangle) indicates where fishing is curtailed in order to avoid reaching B_{LRP} with high probability (panel c). A high probability of avoiding fishing mortalities exceeding F_{MSY} is ensured by specifying a target fishing mortality lower than F_{MSY} (panel d). Finally, fishing mortality is reduced below a second biomass-based OCP (inverted black triangle) to increase the likelihood of avoiding a fishery closure as B_{LRP} is approached (panel e). Note that the reference points B_{LRP} and F_{MSY} are unaffected by changes to the OCPs in the HCR and therefore management objectives do not change (modified from Cox et al. 2013).

4.3 LIMITATIONS

All results in this study are predicated on the two views of stock status for Pacific Herring in BC represented by models AM1 and AM2 with their associated prior probability distributions (e.g., related to steepness, survey catchability, natural mortality) and structural assumptions that condition assessment outcomes. The range of structural assumptions implemented by the two models is modest in comparison to possible alternative views of productivity, growth, natural mortality, and spatial dynamics. The only alternative hypotheses considered in the stock assessment are focused on parameter uncertainty related to dive survey catchability (DFO 2016a). Consideration of a wider range of structural dynamics could reveal quite different estimates of important model outcomes, including natural and fishing mortality, and other key parameters such as B_0 and steepness. Expression of alternative hypotheses will influence future development of conservation, economic and socio-cultural objectives and the operating models needed to evaluate the efficacy of management options.

The analysis of equilibrium reference points resulted in equilibrium fishing mortality rates that are too aggressive for BC Pacific Herring and imply relatively high spawning biomass levels can be achieved at those rates. This result is contrary to experience with all five major stocks. The estimation of sustainable fishing mortality rates depends on knowledge of the relationship between stock and recruitment. Unfortunately this relationship is one of the most uncertain processes in stock assessment and may be heavily influenced by prior probability distributions assumed for the steepness parameter. Difficulties of estimating the stock-recruit relationship and its relative influence on stock productivity is only exacerbated by non-stationary processes such as those that are likely to apply to BC Pacific Herring, particularly the magnitude of the rate of increase in M. The current structural assumptions that allow time-varying mortality may create parameter confounding with steepness and survey catchabilities, which is potentially masking the relative role of fishing mortality in explaining declines in spawning biomass that began in the mid-2000s for the CC, HG, and WCVI stocks. This challenge, coupled with the relative alignment of maturity-at-age and commercial fishery selectivity, means that future consideration of traditional equilibrium reference points should be based on results of simulation testing with operating models to represent a wide range of plausible structural hypotheses for Pacific Herring population, fishery and ecosystem dynamics.

4.4 RECOMMENDATIONS

Our analysis undertook an evidence-based approach to evaluating whether the major Pacific Herring stocks in BC show stock states consistent with signs of possible serious harm. We diagnosed persistent LP-LB states by inspecting production relationships and interpreted the frontier of such states as the threshold for possible serious harm. Stocks in the CC, HG, and WCVI management areas showed recent persistent LP-LB states that led to prolonged loss of benefits to resource users. For the CC, HG, and WCVI stocks the frontiers of the LP-LB states were estimated to be at spawning depletion levels of 0.244 (WCVI) to 0.328 (HG) for model AM1 and at 0.174 (CC) to 0.284 (HG) for model AM2. Lower levels of spawning biomass within the LP-LB state would represent increased probability of possible serious harm. We believe this approach to evaluating possible serious harm to be most appropriate for BC Pacific Herring, given current assessment model assumptions and outputs. We support the recommendation of Hilborn (2001) and Walters et al. (2008) to routinely conduct production analyses in stock assessments.

Sainsbury (2008) recommended $0.3B_{\text{unfished}}$ as a best practice LRP, however, use of a dynamic biomass-based LRP was rejected by Cox et al. (2015[1]) for BC Pacific Herring. Instead it was recommended that attention focus on reference points related to equilibrium unfished biomass to avoid the “ratcheting down” effect. Therefore, we adopt $0.3B_0$ as a proxy for Sainsbury’s

(2008) best practice recommendation that reduces the potential for progressive lowering of conservation thresholds.

The LRP of $0.3B_0$ is within the range of the estimated upper bound ("frontier") of the LP-LB states for the CC, HG and WCVI stocks based on model AM1. For model AM2, the frontier of the LP-LB state is estimated to be lower by approximately 0.11 to 0.02 depletion units depending on the stock. The DFO PA Framework is clear that LRPs must be positioned before a possible state of serious harm, i.e., at a higher spawning biomass level, or lower fishing mortality rate, than states coincident with possible serious harm. However, there is little policy or scientific guidance as to how far above the state of possible serious harm a biomass-based LRP should be positioned in order to avoid serious harm with high probability, particularly in the presence of non-stationary processes such as those that exist for BC Pacific Herring. We also note that the estimates of spawning stock depletion within the LP-LB state are subject to uncertainty, as are the OCPs that will be needed in MPs intended to avoid biomass-based LRPs with high probability. Therefore, we recommend that the biomass-based LRP of $0.3B_0$ be adopted for the CC, HG, and WCVI stocks.

We did not diagnose persistent LP-LB states for stocks in the PRD and SOG management areas and so by analogy, recommend the same LRP of $0.3B_0$, due to the common life history and evidence of recent persistent LP-LB states in geographically adjacent stock areas. Various authors have suggested differences in productivity among BC Pacific Herring stocks (e.g., Cleary et al. 2010; Cox et al. 2015[1]; Schweigert 1995). However, in the current stock assessment model, productivity is fundamentally driven by assumptions about natural mortality and steepness, and the observed changes in size-at-age. Confounding interactions with other model parameters can also be expected. Therefore, before research can be undertaken to evaluate stock-specific differences in productivity, we recommend that model development should focus on the parameterization of natural mortality, estimates of maturity-at-age, and the effects of prior probability distributions for steepness and survey catchability on model outcomes. This work is needed prior to embarking on simulation studies to extend the work of Cox et al. (2015[1]) to evaluate the performance of limit and target reference points, with a view to investigations of the effects of non-stationary processes.

We advise that under the strategic stream to renew the BC Pacific Herring management system, operating models that represent alternative views of stock and fishery dynamics will each be associated with a set of reference points. The utility of the recommended LRPs, and those associated with future operating models, should be subject to careful review of the objectives in which they are applied. The operating models may in future be extended to include ecosystem considerations related to predator communities. However, in the absence of operating models that represent these dependencies, we do not at this time recommend further adjustment of the biomass-based LRP to service ecosystem requirements.

The introduction of biomass-based LRPs for BC Pacific Herring will require deliberate design of MPs using simulation to identify the biomass-based and fishing mortality-based OCPs for harvest control rules. This process will require a fulsome statement of conservation, economic, and socio-cultural objectives in order to evaluate which MPs provide acceptable trade-offs in management outcomes that lead to sustained benefits for resource users. In agreement with the recommendations of Cox et al. (2015[1]), it will be critical during the evaluation to consider consequences such as the frequency of fishery closures while evaluating HCRs with higher biomass OCPs and/or lower harvest rate OCPs. Finally, in transitioning from the existing operational stream to the strategic stream, it is recommended that new MPs be phased-in to mitigate short-term consequences to resource users.

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APPENDIX A – SYNOPSIS OF STATUS BY STOCK

Stock status of Pacific Herring for the CC, HG, PRD, and SOG management areas is summarized graphically based on data and assessment model outputs (maximum posterior density (MPD) estimates) for models AM1 and AM2. Figure 14 through Figure 21 show results for models AM1 and AM2 for each stock in CC, HG, PRD, and SOG management areas. Results for the WCVI management area appear in the main body of the paper.

Each figure consists of six panels. Panel (a) shows the 1951-2016 time series of estimated spawning biomass (circles) and catch (bars). The lower 20% of spawning biomass estimates are shaded grey. Reference lines are shown at estimates of $0.1B_0$, (red dashed line) $0.25B_0$ (blue dashed line), $0.3B_0$ (green dashed line), the 1996 fixed cut-off value (thick blue long dashed line), and unfished spawning biomass (solid blue line). Panel (b) shows observed surface (grey squares) and dive (open circles) survey indices scaled to spawning stock biomass. A trailing 3-year moving average smoother indicates trend for the surface (solid blue line) and dive (solid red line) survey indices. The horizontal dashed line is positioned at the mean observed value for each survey series. Estimated survey catchabilities, q_1 (surface survey) and q_2 (dive survey), are shown in the panel. Recruitment deviations (grey bars) from a Beverton-Holt stock-recruitment function are shown in panel (c). A 3-year trailing moving average smoother (red line) shows the trend in deviations. Estimated natural mortality is shown in panel (d). Panel (e) shows a 3-year trailing moving average of observed weight-at-age for age classes 2-6. Estimated harvest rates are shown in panel (f). Reference lines are shown at the intended harvest rate of 0.2 (horizontal dotted line) and at the average harvest rate from 1983 to the first year of commercial fishery closure or 2016. All model estimates are the maximum posterior density (MPD) estimates.

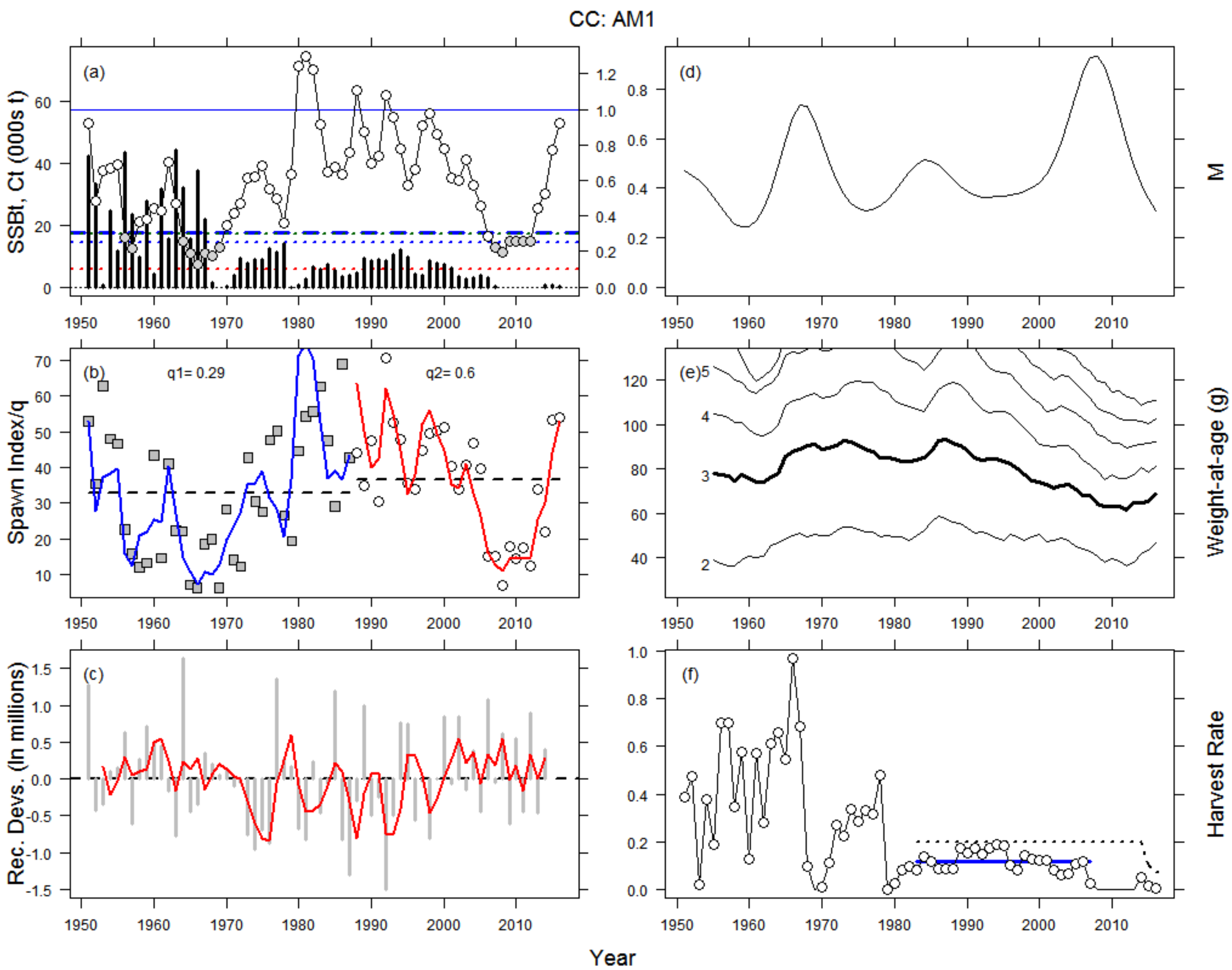

Figure 14. Stock reconstruction for CC Pacific Herring based on assessment model AM1 (DFO 2016a).

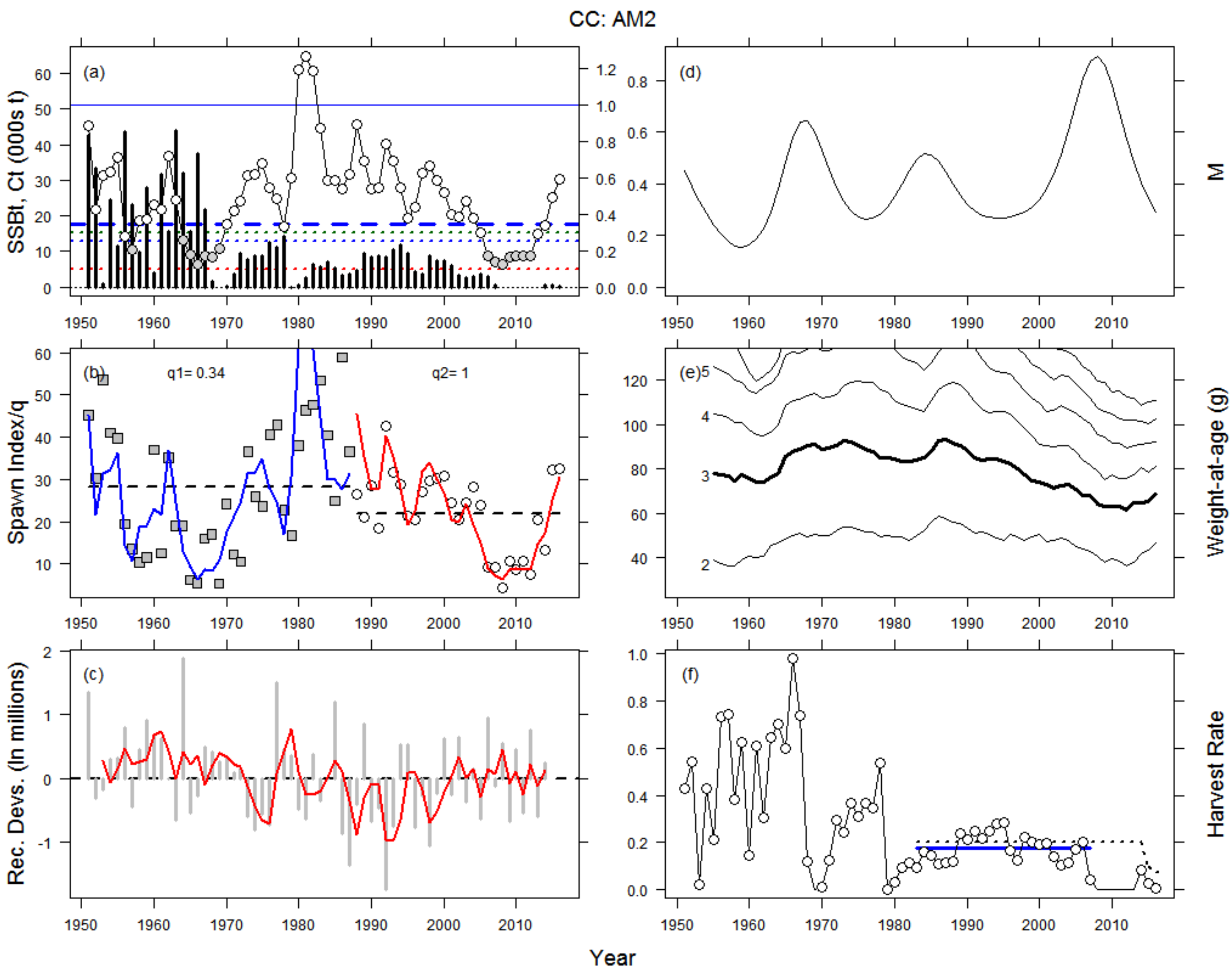

Figure 15. Stock reconstruction for CC Pacific Herring based on assessment model AM2 (DFO 2016a).

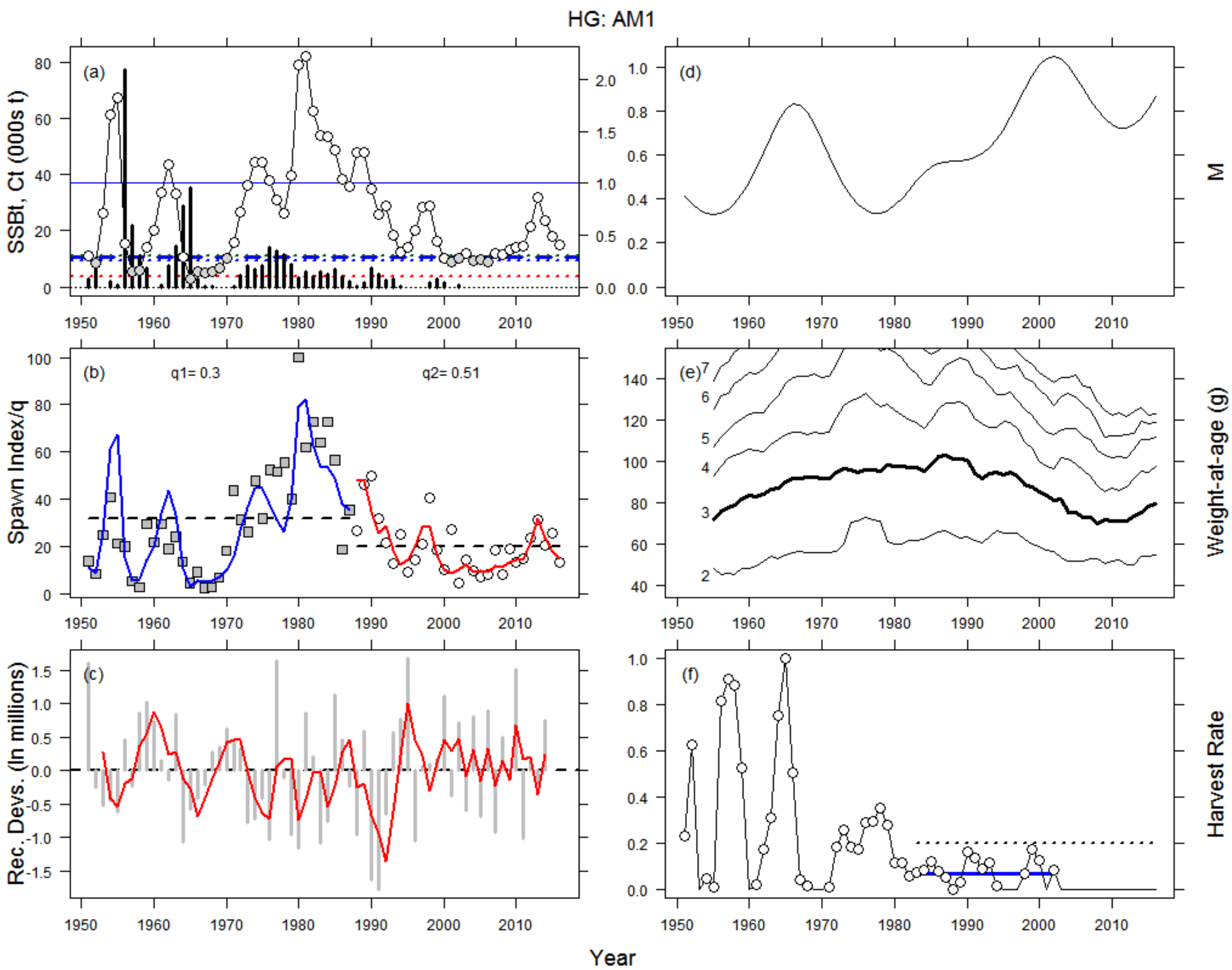

Figure 16. Stock reconstruction for HG Pacific Herring based on assessment model AM1 (DFO 2016a).

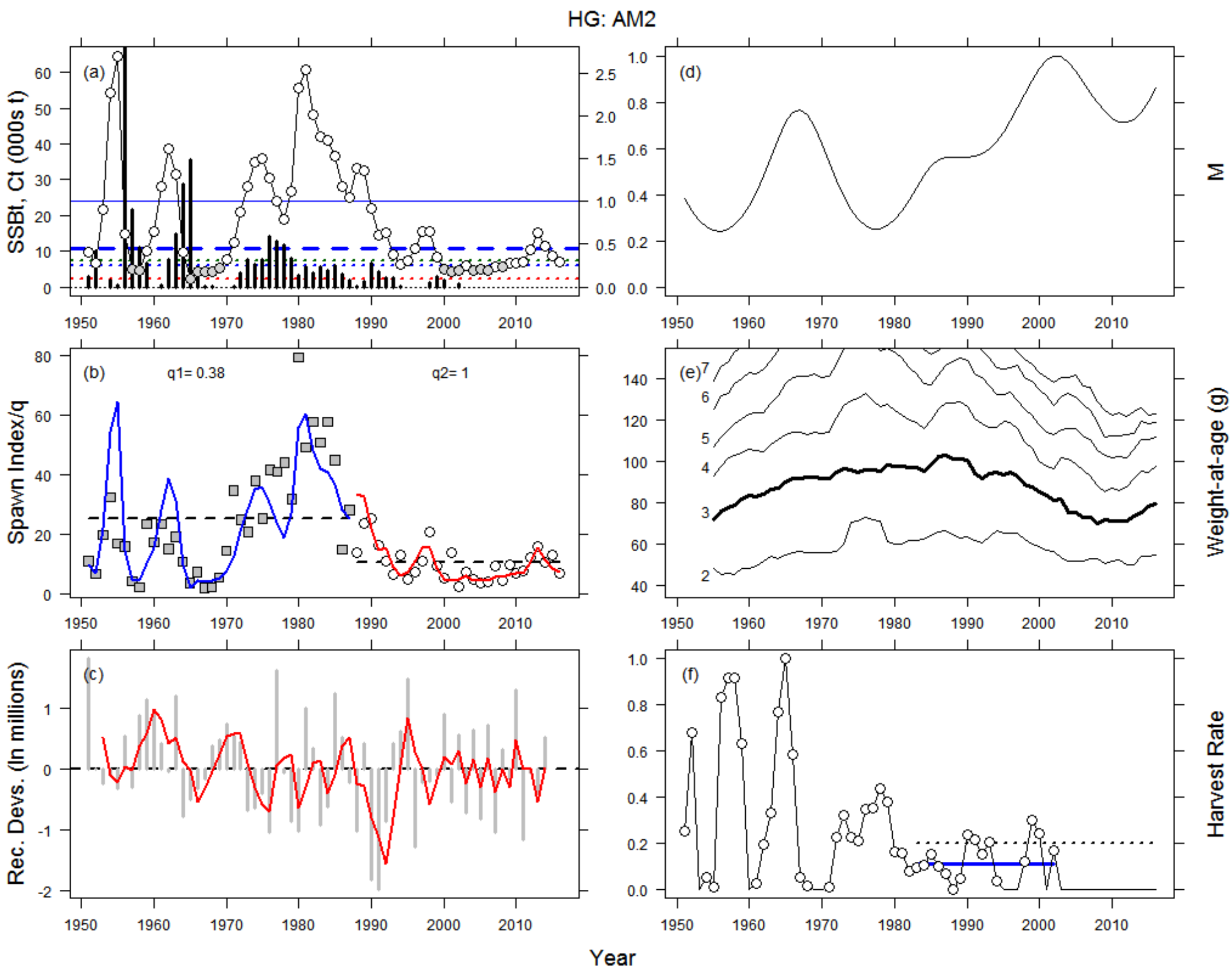

Figure 17. Stock reconstruction for HG Pacific Herring based on assessment model AM2 (DFO 2016a).

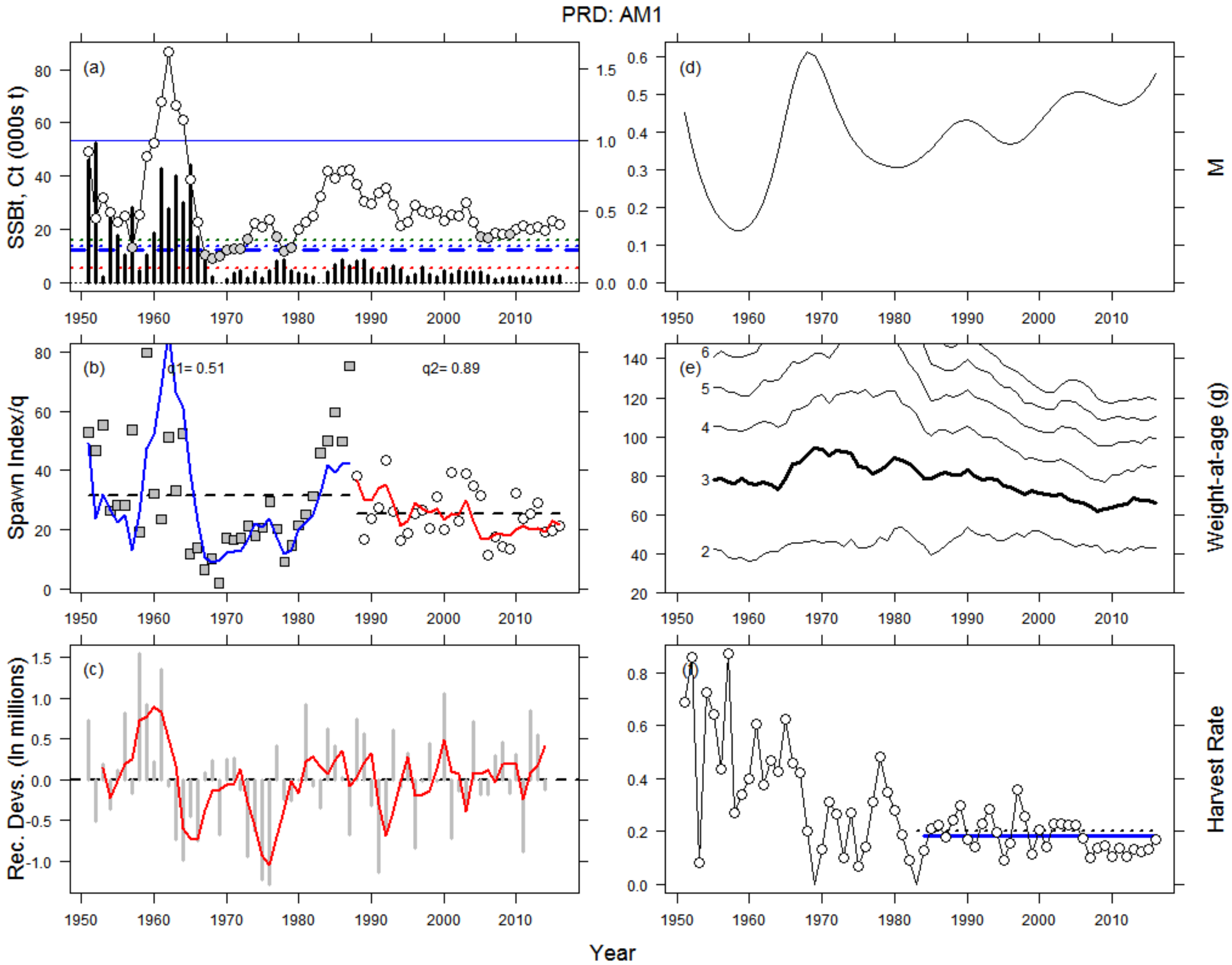

Figure 18. Stock reconstruction for PRD Pacific Herring based on assessment model AM1 (DFO 2016a).

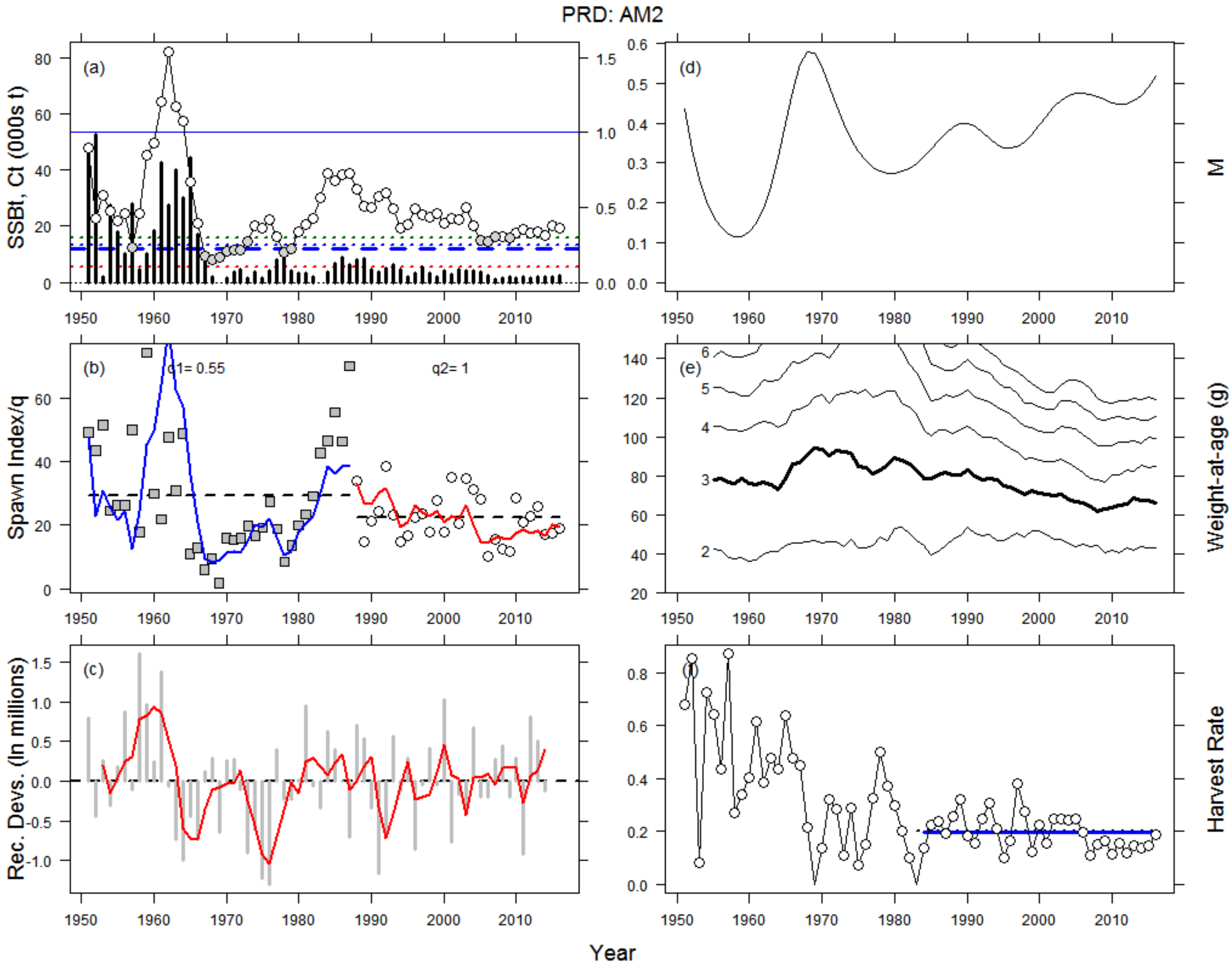

Figure 19. Stock reconstruction for PRD Pacific Herring based on assessment model AM2 (DFO 2016a).

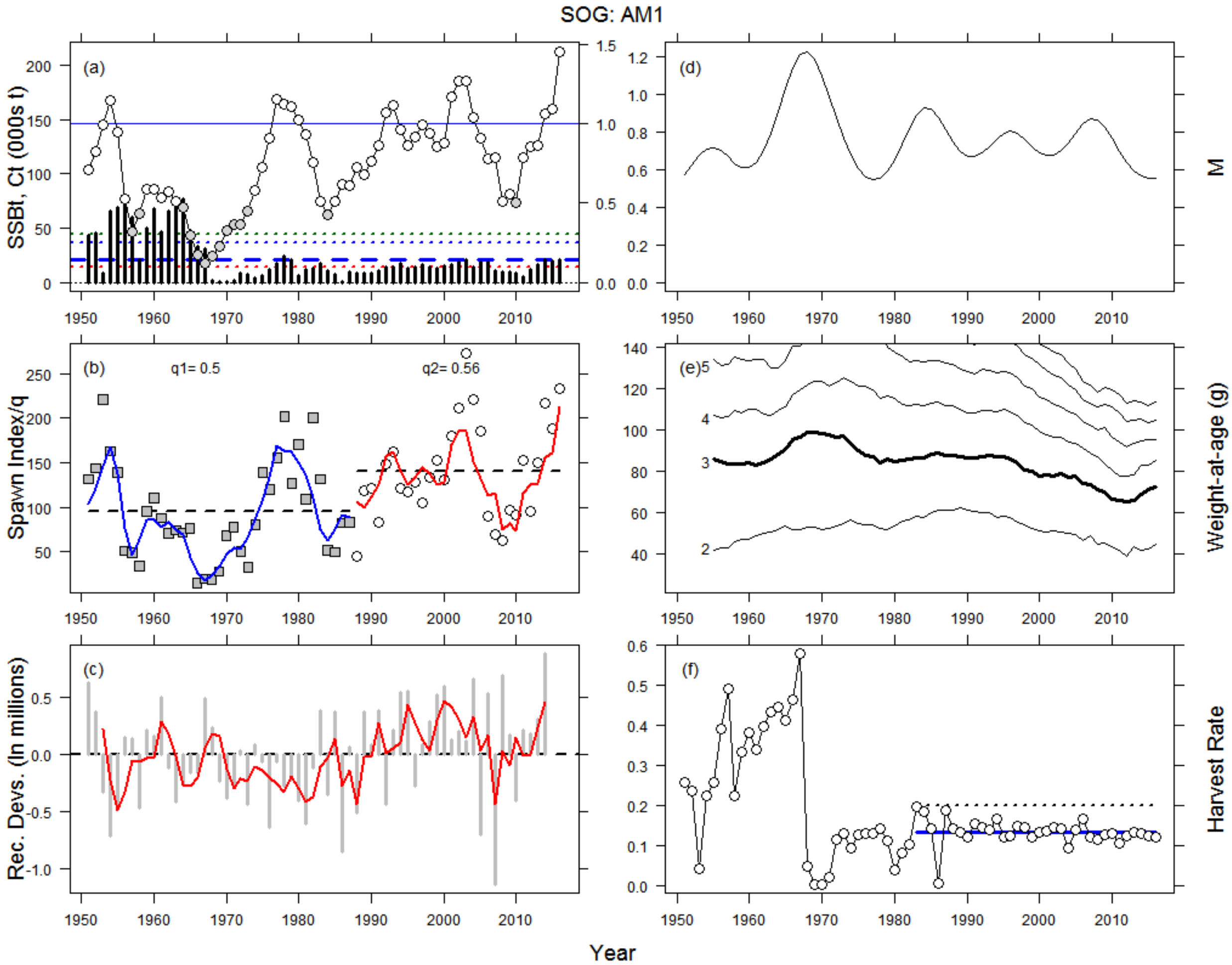

Figure 20. Stock reconstruction for SOG Pacific Herring based on assessment model AM1 (DFO 2016a).

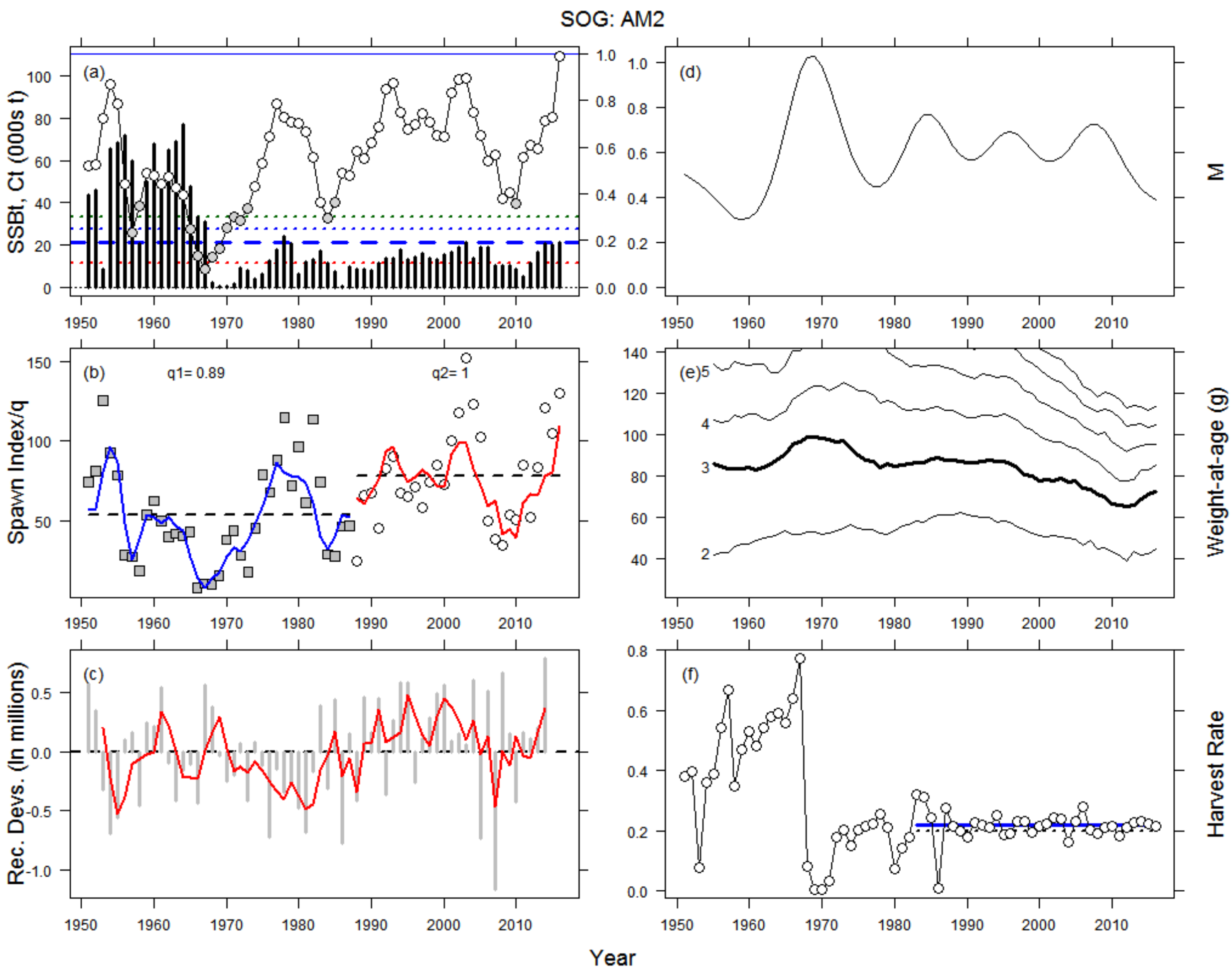

Figure 21. Stock reconstruction for SOG Pacific Herring based on assessment model AM2 (DFO 2016a).

APPENDIX B – SURPLUS PRODUCTION ANALYSIS

This appendix contains the results of surplus production analysis of Pacific Herring for the CC, HG, PRD, and SOG management areas based on catch data and assessment model estimates (maximum posterior density (MPD) estimates) of spawning stock biomass.

Table 7 through Table 14 report annual estimates of spawning stock biomass (B), depletion (D), surplus production (P), production rate (P / B) for the WCVI management area for model AM1. The spawn index value (I), scaled index value (I / q), catch (C), and harvest rate (U) are also shown. Results are provided for models AM1 and AM2.

Figure 22 through Figure 29 show plotted results for models AM1 and AM2 for each stock in CC, HG, PRD, and SOG management areas. Results for the WCVI management area appear in the main body of the paper.

Each figure consists of three panels that show the time series of observed spawn indices scaled by their respective estimates of catchability (panel a), SSB production (panel b), and SSB production rate (panel c). The 1951-2015 time series is broken into two periods corresponding to the surface survey (1951-1987, open squares) and dive survey (1988-2015, open circles). Years with SSB values in the lower 20% of the 1951-2015 series are shaded grey to allow comparison to corresponding values in Figure 14a to Figure 21a. Blue and red lines indicate a 3-year trailing moving average smoother applied to the plotted values for each survey period. Horizontal dashed lines in panel (a) indicate the mean survey index value for each survey period. Horizontal dotted lines are positioned at zero production and production rate in the two panel lower panels (b, c).

Table 7. *Annual estimates of spawning biomass (B, 000s t), spawning biomass depletion (D), spawn index (I), spawn index scaled by catchability (I/q), catch (C, 000s t), harvest rate (U), surplus production (P), and production rate (P/B) for the CC management area for model AM1. Years during the stock collapse from 1964-1969 are shaded light grey and years with a recent persistent LP-LB state from 2006-2011 are shaded dark grey.*

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1951	Surface	53.04	0.92	15.39	52.90	42.46	0.39	8.03	0.15
1952	Surface	27.87	0.49	10.30	35.38	33.20	0.47	10.31	0.37
1953	Surface	37.41	0.65	18.24	62.68	0.77	0.02	25.51	0.68
1954	Surface	38.29	0.67	13.97	48.01	24.62	0.38	12.84	0.34
1955	Surface	39.54	0.69	13.56	46.62	11.59	0.19	20.07	0.51
1956	Surface	15.99	0.28	6.63	22.77	43.63	0.70	19.64	1.23
1957	Surface	12.36	0.22	4.61	15.83	23.26	0.70	18.41	1.49
1958	Surface	20.92	0.36	3.55	12.20	9.85	0.35	28.89	1.38
1959	Surface	21.94	0.38	3.90	13.42	27.87	0.58	7.50	0.34
1960	Surface	25.41	0.44	12.62	43.36	4.04	0.13	30.97	1.22
1961	Surface	24.67	0.43	4.27	14.66	31.70	0.57	31.36	1.27
1962	Surface	40.33	0.70	11.95	41.07	15.71	0.28	30.69	0.76
1963	Surface	26.97	0.47	6.49	22.29	44.05	0.61	19.83	0.74
1964	Surface	14.91	0.26	6.46	22.22	31.90	0.66	11.56	0.78
1965	Surface	10.80	0.19	2.10	7.21	15.67	0.55	33.93	3.14
1966	Surface	7.24	0.13	1.86	6.40	37.48	0.97	25.47	3.52
1967	Surface	10.82	0.19	5.43	18.68	21.89	0.68	0.95	0.09
1968	Surface	10.24	0.18	5.79	19.90	1.53	0.10	2.53	0.25
1969	Surface	12.78	0.22	1.84	6.31	0.00	0.00	7.48	0.59
1970	Surface	20.05	0.35	8.23	28.29	0.21	0.01	7.33	0.37
1971	Surface	23.76	0.41	4.16	14.28	3.61	0.11	12.52	0.53
1972	Surface	27.00	0.47	3.57	12.28	9.28	0.27	15.90	0.59
1973	Surface	35.10	0.61	12.43	42.74	7.80	0.22	9.54	0.27
1974	Surface	35.76	0.62	8.85	30.42	8.89	0.34	11.94	0.33
1975	Surface	38.96	0.68	8.04	27.62	8.74	0.28	5.05	0.13
1976	Surface	31.61	0.55	13.85	47.60	12.41	0.33	8.05	0.25
1977	Surface	28.55	0.50	14.61	50.23	11.11	0.32	6.28	0.22
1978	Surface	20.79	0.36	7.75	26.63	14.05	0.48	15.73	0.76
1979	Surface	36.51	0.64	5.67	19.48	0.01	0.00	35.44	0.97
1980	Surface	71.41	1.24	12.96	44.53	0.54	0.03	5.73	0.08
1981	Surface	74.57	1.30	15.81	54.34	2.57	0.08	1.94	0.03
1982	Surface	70.13	1.22	16.22	55.73	6.37	0.10	-11.92	-0.17
1983	Surface	52.57	0.92	18.21	62.60	5.64	0.08	-8.31	-0.16
1984	Surface	37.09	0.65	13.79	47.39	7.17	0.14	6.96	0.19
1985	Surface	38.84	0.68	8.48	29.16	5.21	0.12	1.13	0.03
1986	Surface	36.57	0.64	20.06	68.93	3.39	0.09	10.37	0.28
1987	Surface	43.33	0.76	12.43	42.73	3.62	0.09	24.80	0.57
1988	Dive	63.60	1.11	26.47	43.96	4.53	0.09	-3.92	-0.06
1989	Dive	50.24	0.88	21.10	35.04	9.44	0.17	-1.98	-0.04

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1990	Dive	39.91	0.70	28.55	47.42	8.35	0.15	11.37	0.28
1991	Dive	42.37	0.74	18.43	30.61	8.90	0.18	27.92	0.66
1992	Dive	61.93	1.08	42.59	70.74	8.36	0.15	3.40	0.05
1993	Dive	54.81	0.96	31.72	52.68	10.52	0.17	1.87	0.03
1994	Dive	44.80	0.78	28.79	47.82	11.88	0.19	-2.54	-0.06
1995	Dive	32.68	0.57	21.34	35.45	9.58	0.19	9.49	0.29
1996	Dive	37.87	0.66	20.34	33.79	4.30	0.10	17.85	0.47
1997	Dive	52.10	0.91	27.02	44.87	3.62	0.08	12.38	0.24
1998	Dive	55.86	0.97	29.74	49.39	8.62	0.14	1.11	0.02
1999	Dive	49.44	0.86	30.21	50.17	7.52	0.13	2.75	0.06
2000	Dive	44.82	0.78	30.81	51.17	7.37	0.12	-3.49	-0.08
2001	Dive	35.20	0.61	24.33	40.41	6.13	0.12	2.49	0.07
2002	Dive	34.39	0.60	20.32	33.74	3.29	0.08	9.28	0.27
2003	Dive	41.09	0.72	24.40	40.53	2.59	0.06	-5.11	-0.12
2004	Dive	32.99	0.57	28.25	46.91	2.99	0.07	-2.91	-0.09
2005	Dive	26.30	0.46	23.90	39.70	3.78	0.10	-7.00	-0.27
2006	Dive	16.23	0.28	9.08	15.08	3.07	0.12	-3.09	-0.19
2007	Dive	12.74	0.22	9.26	15.39	0.40	0.02	-1.53	-0.12
2008	Dive	11.21	0.20	4.26	7.07	0.00	0.00	3.46	0.31
2009	Dive	14.68	0.26	10.77	17.89	0.00	0.00	0.27	0.02
2010	Dive	14.95	0.26	8.67	14.40	0.00	0.00	-0.28	-0.02
2011	Dive	14.67	0.26	10.53	17.50	0.00	0.00	0.13	0.01
2012	Dive	14.80	0.26	7.59	12.61	0.00	0.00	10.48	0.71
2013	Dive	25.28	0.44	20.37	33.83	0.00	0.00	5.64	0.22
2014	Dive	30.24	0.53	13.31	22.10	0.69	0.05	14.59	0.48
2015	Dive	44.21	0.77	32.15	53.39	0.63	0.02	9.06	0.21
2016	Dive	53.06	0.92	32.51	53.99	0.21	0.00	*NA*	*NA*

Table 8. Annual estimates of surplus production, production rate, and depletion for the CC management area for model AM2. LP-LB periods are shaded light grey from 1964-1969 and dark grey from 2006-2011. Description as for Table 7.

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1951	Surface	45.32	0.89	15.39	45.18	42.46	0.43	9.53	0.21
1952	Surface	21.65	0.42	10.30	30.22	33.20	0.54	10.44	0.48
1953	Surface	31.32	0.61	18.24	53.54	0.77	0.02	25.60	0.82
1954	Surface	32.30	0.63	13.97	41.00	24.62	0.43	15.57	0.48
1955	Surface	36.28	0.71	13.56	39.82	11.59	0.21	21.65	0.60
1956	Surface	14.31	0.28	6.63	19.45	43.63	0.74	19.53	1.36
1957	Surface	10.57	0.21	4.61	13.52	23.26	0.74	17.86	1.69
1958	Surface	18.58	0.36	3.55	10.42	9.85	0.38	28.17	1.52
1959	Surface	18.89	0.37	3.90	11.46	27.87	0.62	8.18	0.43
1960	Surface	23.03	0.45	12.62	37.03	4.04	0.14	30.36	1.32
1961	Surface	21.69	0.43	4.27	12.52	31.70	0.61	30.93	1.43
1962	Surface	36.91	0.72	11.95	35.07	15.71	0.30	31.50	0.85
1963	Surface	24.36	0.48	6.49	19.04	44.05	0.65	20.57	0.84
1964	Surface	13.04	0.26	6.46	18.98	31.90	0.70	11.88	0.91
1965	Surface	9.25	0.18	2.10	6.16	15.67	0.60	34.46	3.73
1966	Surface	6.22	0.12	1.86	5.47	37.48	0.98	24.45	3.93
1967	Surface	8.79	0.17	5.43	15.95	21.89	0.74	1.03	0.12
1968	Surface	8.29	0.16	5.79	17.00	1.53	0.12	2.52	0.30
1969	Surface	10.81	0.21	1.84	5.39	0.00	0.00	7.03	0.65
1970	Surface	17.63	0.35	8.23	24.16	0.21	0.01	7.34	0.42
1971	Surface	21.35	0.42	4.16	12.20	3.61	0.13	12.01	0.56
1972	Surface	24.08	0.47	3.57	10.49	9.28	0.29	15.03	0.62
1973	Surface	31.31	0.61	12.43	36.50	7.80	0.24	9.34	0.30
1974	Surface	31.76	0.62	8.85	25.99	8.89	0.37	11.71	0.37
1975	Surface	34.73	0.68	8.04	23.59	8.74	0.31	5.62	0.16
1976	Surface	27.94	0.55	13.85	40.65	12.41	0.36	7.97	0.29
1977	Surface	24.80	0.49	14.61	42.90	11.11	0.35	6.19	0.25
1978	Surface	16.94	0.33	7.75	22.74	14.05	0.54	13.69	0.81
1979	Surface	30.63	0.60	5.67	16.64	0.01	0.00	31.10	1.02
1980	Surface	61.19	1.20	12.96	38.04	0.54	0.03	6.21	0.10
1981	Surface	64.83	1.27	15.81	46.41	2.57	0.09	2.19	0.03
1982	Surface	60.65	1.19	16.22	47.60	6.37	0.11	-10.34	-0.17
1983	Surface	44.67	0.88	18.21	53.47	5.64	0.09	-7.43	-0.17
1984	Surface	30.06	0.59	13.79	40.48	7.17	0.16	5.23	0.17
1985	Surface	30.08	0.59	8.48	24.90	5.21	0.14	0.83	0.03
1986	Surface	27.52	0.54	20.06	58.88	3.39	0.11	7.60	0.28
1987	Surface	31.51	0.62	12.43	36.49	3.62	0.11	18.56	0.59
1988	Dive	45.54	0.89	26.47	26.51	4.53	0.12	-0.72	-0.02
1989	Dive	35.37	0.69	21.10	21.13	9.44	0.24	0.56	0.02
1990	Dive	27.59	0.54	28.55	28.60	8.35	0.21	9.08	0.33
1991	Dive	27.77	0.54	18.43	18.46	8.90	0.25	20.78	0.75

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1992	Dive	40.19	0.79	42.59	42.67	8.36	0.21	5.74	0.14
1993	Dive	35.41	0.69	31.72	31.77	10.52	0.25	4.49	0.13
1994	Dive	28.02	0.55	28.79	28.84	11.88	0.28	0.96	0.03
1995	Dive	19.40	0.38	21.34	21.38	9.58	0.29	7.41	0.38
1996	Dive	22.51	0.44	20.34	20.38	4.30	0.16	13.00	0.58
1997	Dive	31.89	0.63	27.02	27.06	3.62	0.13	10.60	0.33
1998	Dive	33.87	0.66	29.74	29.79	8.62	0.22	3.54	0.10
1999	Dive	29.88	0.59	30.21	30.26	7.52	0.20	4.13	0.14
2000	Dive	26.65	0.52	30.81	30.86	7.37	0.19	-0.20	-0.01
2001	Dive	20.32	0.40	24.33	24.38	6.13	0.19	2.71	0.13
2002	Dive	19.74	0.39	20.32	20.35	3.29	0.14	6.95	0.35
2003	Dive	24.09	0.47	24.40	24.44	2.59	0.10	-1.69	-0.07
2004	Dive	19.41	0.38	28.25	28.29	2.99	0.11	-0.52	-0.03
2005	Dive	15.12	0.30	23.90	23.94	3.78	0.17	-3.19	-0.21
2006	Dive	8.86	0.17	9.08	9.10	3.07	0.20	-1.32	-0.15
2007	Dive	7.15	0.14	9.26	9.28	0.40	0.04	-0.70	-0.10
2008	Dive	6.45	0.13	4.26	4.26	0.00	0.00	2.11	0.33
2009	Dive	8.56	0.17	10.77	10.79	0.00	0.00	0.25	0.03
2010	Dive	8.81	0.17	8.67	8.69	0.00	0.00	-0.11	-0.01
2011	Dive	8.70	0.17	10.53	10.55	0.00	0.00	0.05	0.01
2012	Dive	8.75	0.17	7.59	7.61	0.00	0.00	6.03	0.69
2013	Dive	14.78	0.29	20.37	20.40	0.00	0.00	3.24	0.22
2014	Dive	17.33	0.34	13.31	13.33	0.69	0.08	8.45	0.49
2015	Dive	25.16	0.49	32.15	32.20	0.63	0.03	5.47	0.22
2016	Dive	30.42	0.60	32.51	32.56	0.21	0.01	*NA*	*NA*

Table 9. Annual estimates of surplus production, production rate, and depletion for the HG management area for model AM1. LP-LB periods are shaded light grey from 1965-1969 and dark grey from 2000-2008. Description as for Table 7.

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1951	Surface	10.99	0.30	4.21	13.84	2.85	0.23	7.55	0.69
1952	Surface	8.40	0.23	2.58	8.47	10.15	0.62	17.84	2.12
1953	Surface	26.24	0.71	7.56	24.82	0.00	0.00	36.85	1.40
1954	Surface	61.30	1.66	12.41	40.76	1.79	0.05	6.56	0.11
1955	Surface	67.36	1.82	6.44	21.15	0.50	0.01	25.48	0.38
1956	Surface	15.38	0.42	6.04	19.85	77.46	0.82	11.76	0.76
1957	Surface	5.34	0.14	1.59	5.23	21.80	0.91	11.82	2.22
1958	Surface	6.01	0.16	0.82	2.68	11.15	0.88	15.08	2.51
1959	Surface	14.25	0.39	8.98	29.50	6.83	0.53	5.86	0.41
1960	Surface	20.11	0.54	6.60	21.68	0.00	0.00	14.33	0.71
1961	Surface	33.87	0.92	8.98	29.50	0.58	0.02	17.29	0.51
1962	Surface	43.53	1.18	5.73	18.82	7.63	0.17	4.56	0.10
1963	Surface	33.38	0.90	7.30	23.97	14.71	0.31	6.11	0.18
1964	Surface	10.71	0.29	4.10	13.48	28.77	0.75	27.61	2.58
1965	Surface	2.87	0.08	1.38	4.53	35.45	1.00	5.40	1.88
1966	Surface	5.53	0.15	2.82	9.28	2.75	0.50	-0.23	-0.04
1967	Surface	5.09	0.14	0.71	2.33	0.21	0.04	0.36	0.07
1968	Surface	5.37	0.15	0.83	2.74	0.08	0.01	1.56	0.29
1969	Surface	6.93	0.19	2.08	6.82	0.00	0.00	3.13	0.45
1970	Surface	10.06	0.27	5.55	18.24	0.00	0.00	6.04	0.60
1971	Surface	16.00	0.43	13.29	43.66	0.10	0.01	14.78	0.92
1972	Surface	26.81	0.73	9.54	31.34	3.97	0.19	16.95	0.63
1973	Surface	36.24	0.98	7.96	26.15	7.52	0.26	14.68	0.41
1974	Surface	44.60	1.21	14.51	47.66	6.32	0.18	7.76	0.17
1975	Surface	44.64	1.21	9.69	31.82	7.72	0.17	7.37	0.17
1976	Surface	37.90	1.02	15.99	52.51	14.12	0.29	5.91	0.16
1977	Surface	31.17	0.84	15.72	51.63	12.64	0.29	6.69	0.21
1978	Surface	26.14	0.71	16.89	55.47	11.73	0.35	21.50	0.82
1979	Surface	39.68	1.07	12.24	40.19	7.95	0.28	42.93	1.08
1980	Surface	79.30	2.14	30.46	100.04	3.32	0.11	8.63	0.11
1981	Surface	82.30	2.23	18.82	61.83	5.63	0.12	-15.81	-0.19
1982	Surface	62.71	1.70	22.16	72.79	3.78	0.06	-2.97	-0.05
1983	Surface	54.15	1.46	19.47	63.96	5.60	0.07	4.17	0.08
1984	Surface	53.67	1.45	22.12	72.66	4.65	0.08	1.45	0.03
1985	Surface	49.00	1.33	17.23	56.61	6.11	0.12	-7.29	-0.15
1986	Surface	38.21	1.03	5.68	18.66	3.50	0.08	-0.49	-0.01
1987	Surface	35.66	0.96	10.75	35.31	2.06	0.05	12.46	0.35
1988	Dive	48.09	1.30	13.63	26.62	0.03	0.00	1.33	0.03
1989	Dive	47.95	1.30	23.64	46.16	1.46	0.03	-6.28	-0.13
1990	Dive	34.96	0.95	25.40	49.61	6.71	0.16	-4.81	-0.14
1991	Dive	25.71	0.70	16.20	31.64	4.44	0.14	5.52	0.21

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1992	Dive	28.71	0.78	11.07	21.61	2.52	0.09	-7.69	-0.27
1993	Dive	18.32	0.50	6.46	12.62	2.70	0.11	-5.66	-0.31
1994	Dive	12.37	0.33	12.81	25.01	0.30	0.02	1.90	0.15
1995	Dive	14.27	0.39	4.70	9.18	0.00	0.00	5.84	0.41
1996	Dive	20.11	0.54	7.37	14.40	0.00	0.00	8.35	0.42
1997	Dive	28.46	0.77	10.78	21.05	0.00	0.00	1.64	0.06
1998	Dive	28.73	0.78	20.68	40.39	1.37	0.07	-9.26	-0.32
1999	Dive	16.49	0.45	9.47	18.50	2.98	0.17	-4.32	-0.26
2000	Dive	10.41	0.28	5.34	10.43	1.77	0.13	-1.59	-0.15
2001	Dive	8.82	0.24	13.86	27.06	0.00	0.00	2.00	0.23
2002	Dive	10.12	0.27	2.29	4.46	0.71	0.09	2.00	0.20
2003	Dive	12.12	0.33	7.40	14.45	0.00	0.00	-2.84	-0.23
2004	Dive	9.28	0.25	4.91	9.58	0.00	0.00	0.64	0.07
2005	Dive	9.91	0.27	3.61	7.06	0.00	0.00	-0.85	-0.09
2006	Dive	9.07	0.25	4.10	8.00	0.00	0.00	2.68	0.30
2007	Dive	11.75	0.32	9.44	18.43	0.00	0.00	-0.37	-0.03
2008	Dive	11.38	0.31	4.21	8.23	0.00	0.00	2.09	0.18
2009	Dive	13.47	0.36	9.79	19.13	0.00	0.00	0.60	0.04
2010	Dive	14.07	0.38	6.85	13.37	0.00	0.00	0.48	0.03
2011	Dive	14.55	0.39	7.55	14.75	0.00	0.00	7.04	0.48
2012	Dive	21.59	0.58	11.98	23.40	0.00	0.00	10.34	0.48
2013	Dive	31.93	0.86	16.03	31.29	0.00	0.00	-8.16	-0.26
2014	Dive	23.77	0.64	10.57	20.63	0.00	0.00	-5.77	-0.24
2015	Dive	17.99	0.49	13.10	25.59	0.00	0.00	-2.96	-0.16
2016	Dive	15.04	0.41	6.89	13.45	0.00	0.00	*NA*	*NA*

Table 10. Annual estimates of surplus production, production rate, and depletion for the HG management area for model AM2. LP-LB periods are shaded light grey from 1965-1969 and dark grey from 2000-2010. Description as for Table 7.

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1951	Surface	9.77	0.41	4.21	10.99	2.85	0.25	7.04	0.72
1952	Surface	6.67	0.28	2.58	6.73	10.15	0.68	15.14	2.27
1953	Surface	21.80	0.91	7.56	19.72	0.00	0.00	34.40	1.58
1954	Surface	54.41	2.27	12.41	32.38	1.79	0.05	10.65	0.20
1955	Surface	64.57	2.70	6.44	16.80	0.50	0.01	27.56	0.43
1956	Surface	14.67	0.61	6.04	15.77	77.46	0.83	12.06	0.82
1957	Surface	4.92	0.21	1.59	4.15	21.80	0.92	10.71	2.18
1958	Surface	4.49	0.19	0.82	2.13	11.15	0.92	12.37	2.75
1959	Surface	10.03	0.42	8.98	23.44	6.83	0.63	5.41	0.54
1960	Surface	15.44	0.65	6.60	17.22	0.00	0.00	13.22	0.86
1961	Surface	28.08	1.17	8.98	23.44	0.58	0.03	18.19	0.65
1962	Surface	38.64	1.61	5.73	14.95	7.63	0.20	7.56	0.20
1963	Surface	31.49	1.32	7.30	19.04	14.71	0.33	7.12	0.23
1964	Surface	9.84	0.41	4.10	10.71	28.77	0.77	27.69	2.81
1965	Surface	2.08	0.09	1.38	3.60	35.45	1.00	4.90	2.35
1966	Surface	4.24	0.18	2.82	7.37	2.75	0.58	0.12	0.03
1967	Surface	4.15	0.17	0.71	1.85	0.21	0.05	0.21	0.05
1968	Surface	4.28	0.18	0.83	2.17	0.08	0.02	1.12	0.26
1969	Surface	5.40	0.23	2.08	5.41	0.00	0.00	2.34	0.43
1970	Surface	7.73	0.32	5.55	14.49	0.00	0.00	4.84	0.63
1971	Surface	12.47	0.52	13.29	34.68	0.10	0.01	12.42	1.00
1972	Surface	20.92	0.87	9.54	24.90	3.97	0.23	14.69	0.70
1973	Surface	28.09	1.17	7.96	20.77	7.52	0.32	13.24	0.47
1974	Surface	35.02	1.46	14.51	37.87	6.32	0.22	8.78	0.25
1975	Surface	36.07	1.51	9.69	25.28	7.72	0.21	8.46	0.23
1976	Surface	30.42	1.27	15.99	41.72	14.12	0.34	6.41	0.21
1977	Surface	24.19	1.01	15.72	41.01	12.64	0.35	6.32	0.26
1978	Surface	18.79	0.78	16.89	44.06	11.73	0.44	15.92	0.85
1979	Surface	26.76	1.12	12.24	31.93	7.95	0.38	32.34	1.21
1980	Surface	55.78	2.33	30.46	79.48	3.32	0.16	10.47	0.19
1981	Surface	60.63	2.53	18.82	49.12	5.63	0.16	-8.62	-0.14
1982	Surface	48.23	2.01	22.16	57.83	3.78	0.08	-0.62	-0.01
1983	Surface	42.01	1.75	19.47	50.81	5.60	0.09	3.64	0.09
1984	Surface	41.00	1.71	22.12	57.72	4.65	0.11	1.70	0.04
1985	Surface	36.59	1.53	17.23	44.97	6.11	0.15	-5.12	-0.14
1986	Surface	27.97	1.17	5.68	14.82	3.50	0.10	-0.71	-0.03
1987	Surface	25.20	1.05	10.75	28.05	2.06	0.07	8.18	0.32
1988	Dive	33.35	1.39	13.63	13.65	0.03	0.00	0.61	0.02
1989	Dive	32.50	1.36	23.64	23.68	1.46	0.05	-3.90	-0.12
1990	Dive	21.88	0.91	25.40	25.44	6.71	0.23	-2.95	-0.13
1991	Dive	14.50	0.61	16.20	16.23	4.44	0.22	3.27	0.23

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1992	Dive	15.25	0.64	11.07	11.09	2.52	0.15	-3.47	-0.23
1993	Dive	9.07	0.38	6.46	6.47	2.70	0.21	-2.55	-0.28
1994	Dive	6.23	0.26	12.81	12.83	0.30	0.03	1.20	0.19
1995	Dive	7.43	0.31	4.70	4.71	0.00	0.00	3.40	0.46
1996	Dive	10.83	0.45	7.37	7.39	0.00	0.00	4.58	0.42
1997	Dive	15.41	0.64	10.78	10.80	0.00	0.00	1.38	0.09
1998	Dive	15.42	0.64	20.68	20.71	1.37	0.12	-4.08	-0.26
1999	Dive	8.36	0.35	9.47	9.49	2.98	0.30	-1.72	-0.21
2000	Dive	4.87	0.20	5.34	5.35	1.77	0.24	-0.60	-0.12
2001	Dive	4.28	0.18	13.86	13.88	0.00	0.00	1.19	0.28
2002	Dive	4.76	0.20	2.29	2.29	0.71	0.17	1.13	0.24
2003	Dive	5.89	0.25	7.40	7.41	0.00	0.00	-1.30	-0.22
2004	Dive	4.59	0.19	4.91	4.91	0.00	0.00	0.34	0.08
2005	Dive	4.93	0.21	3.61	3.62	0.00	0.00	-0.45	-0.09
2006	Dive	4.48	0.19	4.10	4.10	0.00	0.00	1.26	0.28
2007	Dive	5.74	0.24	9.44	9.45	0.00	0.00	-0.21	-0.04
2008	Dive	5.53	0.23	4.21	4.22	0.00	0.00	0.97	0.18
2009	Dive	6.50	0.27	9.79	9.81	0.00	0.00	0.30	0.05
2010	Dive	6.80	0.28	6.85	6.86	0.00	0.00	0.23	0.03
2011	Dive	7.03	0.29	7.55	7.57	0.00	0.00	3.36	0.48
2012	Dive	10.39	0.43	11.98	12.00	0.00	0.00	4.95	0.48
2013	Dive	15.34	0.64	16.03	16.05	0.00	0.00	-3.86	-0.25
2014	Dive	11.48	0.48	10.57	10.58	0.00	0.00	-2.82	-0.25
2015	Dive	8.66	0.36	13.10	13.12	0.00	0.00	-1.54	-0.18
2016	Dive	7.12	0.30	6.89	6.90	0.00	0.00	*NA*	*NA*

Table 11. Annual estimates of surplus production, production rate, and depletion for the PRD management area for model AM1. LP-LB periods are shaded light grey from 1967-1972. Description as for Table 7.

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1951	Surface	49.07	0.92	27.15	52.88	45.87	0.69	27.19	0.55
1952	Surface	23.88	0.45	24.05	46.83	52.38	0.86	9.96	0.42
1953	Surface	31.98	0.60	28.47	55.45	1.87	0.08	21.71	0.68
1954	Surface	26.42	0.49	13.54	26.36	27.28	0.73	13.89	0.53
1955	Surface	22.50	0.42	14.48	28.21	17.81	0.64	12.82	0.57
1956	Surface	25.14	0.47	14.53	28.30	10.18	0.44	16.07	0.64
1957	Surface	13.17	0.25	27.52	53.59	28.04	0.87	16.99	1.29
1958	Surface	25.64	0.48	9.88	19.25	4.52	0.27	32.17	1.25
1959	Surface	47.59	0.89	40.96	79.78	10.22	0.34	23.09	0.49
1960	Surface	52.20	0.98	16.55	32.22	18.48	0.40	58.70	1.12
1961	Surface	68.16	1.27	12.06	23.49	42.75	0.61	46.29	0.68
1962	Surface	86.79	1.62	26.33	51.28	27.66	0.38	19.88	0.23
1963	Surface	66.44	1.24	16.98	33.07	40.23	0.47	24.60	0.37
1964	Surface	61.11	1.14	26.92	52.43	29.93	0.43	21.84	0.36
1965	Surface	38.74	0.72	6.06	11.79	44.21	0.62	1.30	0.03
1966	Surface	22.75	0.43	7.11	13.84	17.30	0.46	-4.27	-0.19
1967	Surface	10.48	0.20	3.39	6.59	8.00	0.42	0.63	0.06
1968	Surface	9.04	0.17	5.20	10.12	2.07	0.20	0.71	0.08
1969	Surface	9.75	0.18	0.97	1.88	0.00	0.00	3.67	0.38
1970	Surface	12.09	0.23	8.81	17.17	1.33	0.13	4.00	0.33
1971	Surface	12.59	0.24	8.48	16.52	3.50	0.31	4.64	0.37
1972	Surface	12.74	0.24	8.77	17.09	4.49	0.27	5.03	0.40
1973	Surface	16.16	0.30	10.96	21.34	1.61	0.10	9.72	0.60
1974	Surface	22.07	0.41	9.24	18.00	3.82	0.27	0.52	0.02
1975	Surface	20.88	0.39	10.57	20.58	1.70	0.07	7.01	0.34
1976	Surface	23.59	0.44	15.20	29.60	4.31	0.14	1.97	0.08
1977	Surface	17.42	0.33	10.43	20.30	8.14	0.31	2.91	0.17
1978	Surface	11.74	0.22	4.73	9.22	8.59	0.48	5.74	0.49
1979	Surface	13.16	0.25	7.60	14.80	4.32	0.35	10.23	0.78
1980	Surface	19.97	0.37	11.00	21.43	3.43	0.28	5.65	0.28
1981	Surface	22.53	0.42	12.94	25.20	3.09	0.19	4.37	0.19
1982	Surface	24.91	0.47	16.11	31.37	1.99	0.09	7.59	0.30
1983	Surface	32.50	0.61	23.58	45.92	0.00	0.00	13.00	0.40
1984	Surface	41.80	0.78	25.70	50.06	3.71	0.13	4.23	0.10
1985	Surface	39.28	0.73	30.68	59.74	6.75	0.21	11.41	0.29
1986	Surface	42.01	0.79	25.58	49.82	8.68	0.22	6.71	0.16
1987	Surface	42.44	0.79	38.67	75.32	6.27	0.18	2.36	0.06
1988	Dive	36.84	0.69	33.96	38.22	7.97	0.24	1.91	0.05
1989	Dive	30.27	0.57	14.88	16.74	8.47	0.30	4.12	0.14
1990	Dive	29.70	0.56	21.18	23.84	4.69	0.17	7.66	0.26
1991	Dive	33.85	0.63	24.31	27.36	3.51	0.14	6.61	0.20

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1992	Dive	35.29	0.66	38.59	43.43	5.18	0.23	0.03	0.00
1993	Dive	28.99	0.54	23.33	26.26	6.32	0.28	-2.95	-0.10
1994	Dive	21.35	0.40	14.68	16.53	4.69	0.20	3.60	0.17
1995	Dive	22.90	0.43	16.88	19.00	2.06	0.09	9.32	0.41
1996	Dive	29.13	0.54	22.66	25.51	3.09	0.15	3.35	0.12
1997	Dive	26.94	0.50	23.57	26.52	5.54	0.36	2.13	0.08
1998	Dive	25.85	0.48	18.00	20.26	3.22	0.26	3.26	0.13
1999	Dive	27.00	0.50	27.74	31.22	2.12	0.12	0.66	0.02
2000	Dive	23.35	0.44	17.94	20.20	4.32	0.21	4.94	0.21
2001	Dive	25.37	0.47	35.07	39.47	2.92	0.14	4.16	0.16
2002	Dive	25.03	0.47	20.50	23.08	4.49	0.23	8.90	0.36
2003	Dive	29.92	0.56	34.63	38.98	4.01	0.23	-2.98	-0.10
2004	Dive	22.83	0.43	31.10	35.01	4.11	0.22	-1.99	-0.09
2005	Dive	17.03	0.32	28.17	31.71	3.80	0.23	2.25	0.13
2006	Dive	16.67	0.31	10.26	11.54	2.62	0.18	2.70	0.16
2007	Dive	18.40	0.34	15.70	17.67	0.97	0.10	1.59	0.09
2008	Dive	18.33	0.34	12.73	14.33	1.66	0.14	1.54	0.08
2009	Dive	17.87	0.33	11.96	13.46	2.00	0.15	3.67	0.21
2010	Dive	20.06	0.38	28.61	32.20	1.49	0.10	3.55	0.18
2011	Dive	21.46	0.40	21.10	23.75	2.15	0.14	-0.04	0.00
2012	Dive	20.04	0.37	22.72	25.57	1.38	0.11	2.62	0.13
2013	Dive	20.64	0.39	25.76	28.99	2.03	0.13	0.62	0.03
2014	Dive	19.26	0.36	17.13	19.27	2.00	0.13	5.92	0.31
2015	Dive	23.01	0.43	17.41	19.59	2.16	0.13	1.21	0.05
2016	Dive	21.80	0.41	18.99	21.37	2.43	0.17	*NA*	*NA*

Table 12. Annual estimates of surplus production, production rate, and depletion for the PRD management area for model AM2. LP-LB periods are shaded grey from 1967-1972. Description as for Table 7.

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1951	Surface	47.72	0.90	27.15	49.24	45.87	0.68	27.34	0.57
1952	Surface	22.69	0.43	24.05	43.61	52.38	0.86	10.19	0.45
1953	Surface	31.01	0.58	28.47	51.63	1.87	0.08	21.77	0.70
1954	Surface	25.50	0.48	13.54	24.55	27.28	0.73	14.03	0.55
1955	Surface	21.72	0.41	14.48	26.26	17.81	0.64	12.95	0.60
1956	Surface	24.49	0.46	14.53	26.36	10.18	0.44	16.00	0.65
1957	Surface	12.45	0.23	27.52	49.91	28.04	0.87	16.38	1.32
1958	Surface	24.31	0.46	9.88	17.92	4.52	0.27	31.20	1.28
1959	Surface	45.28	0.85	40.96	74.29	10.22	0.34	22.90	0.51
1960	Surface	49.71	0.93	16.55	30.01	18.48	0.40	57.10	1.15
1961	Surface	64.06	1.20	12.06	21.87	42.75	0.62	45.62	0.71
1962	Surface	82.02	1.54	26.33	47.75	27.66	0.38	20.79	0.25
1963	Surface	62.58	1.18	16.98	30.80	40.23	0.48	24.73	0.40
1964	Surface	57.38	1.08	26.92	48.82	29.93	0.44	22.62	0.39
1965	Surface	35.79	0.67	6.06	10.98	44.21	0.64	2.38	0.07
1966	Surface	20.88	0.39	7.11	12.89	17.30	0.48	-3.42	-0.16
1967	Surface	9.46	0.18	3.39	6.14	8.00	0.45	0.83	0.09
1968	Surface	8.21	0.15	5.20	9.43	2.07	0.21	0.80	0.10
1969	Surface	9.01	0.17	0.97	1.75	0.00	0.00	3.53	0.39
1970	Surface	11.22	0.21	8.81	15.98	1.33	0.14	3.88	0.35
1971	Surface	11.60	0.22	8.48	15.38	3.50	0.32	4.47	0.39
1972	Surface	11.58	0.22	8.77	15.91	4.49	0.28	4.75	0.41
1973	Surface	14.72	0.28	10.96	19.87	1.61	0.11	9.16	0.62
1974	Surface	20.07	0.38	9.24	16.76	3.82	0.29	1.01	0.05
1975	Surface	19.37	0.36	10.57	19.16	1.70	0.07	7.13	0.37
1976	Surface	22.20	0.42	15.20	27.56	4.31	0.15	2.30	0.10
1977	Surface	16.36	0.31	10.43	18.91	8.14	0.33	3.04	0.19
1978	Surface	10.81	0.20	4.73	8.59	8.59	0.50	5.39	0.50
1979	Surface	11.88	0.22	7.60	13.78	4.32	0.37	9.52	0.80
1980	Surface	17.98	0.34	11.00	19.95	3.43	0.30	5.53	0.31
1981	Surface	20.42	0.38	12.94	23.47	3.09	0.20	4.37	0.21
1982	Surface	22.81	0.43	16.11	29.21	1.99	0.10	7.25	0.32
1983	Surface	30.06	0.56	23.58	42.76	0.00	0.00	12.21	0.41
1984	Surface	38.56	0.72	25.70	46.61	3.71	0.14	4.43	0.11
1985	Surface	36.24	0.68	30.68	55.63	6.75	0.22	10.93	0.30
1986	Surface	38.49	0.72	25.58	46.39	8.68	0.24	6.32	0.16
1987	Surface	38.54	0.72	38.67	70.14	6.27	0.19	2.66	0.07
1988	Dive	33.23	0.62	33.96	33.95	7.97	0.26	2.39	0.07
1989	Dive	27.14	0.51	14.88	14.87	8.47	0.32	4.19	0.15
1990	Dive	26.64	0.50	21.18	21.17	4.69	0.18	7.18	0.27
1991	Dive	30.31	0.57	24.31	24.30	3.51	0.15	6.60	0.22

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1992	Dive	31.73	0.60	38.59	38.57	5.18	0.25	0.78	0.02
1993	Dive	26.19	0.49	23.33	23.32	6.32	0.31	-2.11	-0.08
1994	Dive	19.39	0.36	14.68	14.68	4.69	0.21	3.45	0.18
1995	Dive	20.78	0.39	16.88	16.87	2.06	0.10	8.49	0.41
1996	Dive	26.19	0.49	22.66	22.66	3.09	0.16	3.32	0.13
1997	Dive	23.97	0.45	23.57	23.56	5.54	0.38	2.27	0.09
1998	Dive	23.03	0.43	18.00	17.99	3.22	0.28	3.38	0.15
1999	Dive	24.29	0.46	27.74	27.73	2.12	0.13	0.91	0.04
2000	Dive	20.89	0.39	17.94	17.94	4.32	0.22	4.75	0.23
2001	Dive	22.72	0.43	35.07	35.06	2.92	0.15	3.96	0.17
2002	Dive	22.19	0.42	20.50	20.50	4.49	0.25	8.30	0.37
2003	Dive	26.48	0.50	34.63	34.62	4.01	0.25	-2.12	-0.08
2004	Dive	20.24	0.38	31.10	31.09	4.11	0.24	-1.42	-0.07
2005	Dive	15.02	0.28	28.17	28.16	3.80	0.25	2.11	0.14
2006	Dive	14.52	0.27	10.26	10.25	2.62	0.20	2.53	0.17
2007	Dive	16.08	0.30	15.70	15.70	0.97	0.11	1.70	0.11
2008	Dive	16.12	0.30	12.73	12.72	1.66	0.15	1.59	0.10
2009	Dive	15.72	0.30	11.96	11.96	2.00	0.16	3.34	0.21
2010	Dive	17.57	0.33	28.61	28.60	1.49	0.12	3.32	0.19
2011	Dive	18.75	0.35	21.10	21.09	2.15	0.15	0.17	0.01
2012	Dive	17.54	0.33	22.72	22.71	1.38	0.12	2.59	0.15
2013	Dive	18.10	0.34	25.76	25.75	2.03	0.15	0.78	0.04
2014	Dive	16.88	0.32	17.13	17.12	2.00	0.14	5.42	0.32
2015	Dive	20.14	0.38	17.41	17.40	2.16	0.15	1.53	0.08
2016	Dive	19.25	0.36	18.99	18.98	2.43	0.19	*NA*	*NA*

Table 13. Annual estimates of surplus production, production rate, and depletion for the SOG management area for model AM1. LP-LB periods are shaded grey from 1966-1969. Description as for Table 7.

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1951	Surface	104.15	0.71	66.14	131.30	43.80	0.26	62.68	0.60
1952	Surface	120.95	0.83	72.38	143.68	45.89	0.23	32.80	0.27
1953	Surface	145.33	0.99	111.31	220.96	8.43	0.04	88.46	0.61
1954	Surface	168.02	1.15	82.14	163.06	65.77	0.22	38.91	0.23
1955	Surface	138.29	0.95	69.85	138.67	68.64	0.26	10.50	0.08
1956	Surface	76.73	0.52	25.67	50.95	72.06	0.39	29.58	0.39
1957	Surface	46.70	0.32	24.47	48.57	59.61	0.49	37.80	0.81
1958	Surface	63.86	0.44	16.91	33.57	20.63	0.22	72.39	1.13
1959	Surface	86.23	0.59	47.86	95.02	50.03	0.33	67.60	0.78
1960	Surface	85.79	0.59	55.71	110.59	68.04	0.38	38.40	0.45
1961	Surface	77.98	0.53	44.33	87.99	46.22	0.34	70.97	0.91
1962	Surface	83.64	0.57	35.60	70.66	65.30	0.40	59.97	0.72
1963	Surface	74.76	0.51	37.38	74.21	68.85	0.43	71.70	0.96
1964	Surface	69.59	0.48	35.95	71.37	76.88	0.44	21.71	0.31
1965	Surface	43.47	0.30	38.39	76.21	47.82	0.41	15.32	0.35
1966	Surface	25.45	0.17	7.21	14.31	33.34	0.46	23.02	0.90
1967	Surface	17.43	0.12	9.65	19.15	31.04	0.58	8.88	0.51
1968	Surface	24.41	0.17	9.44	18.74	1.89	0.05	9.09	0.37
1969	Surface	33.31	0.23	14.04	27.87	0.19	0.00	14.41	0.43
1970	Surface	47.48	0.32	34.16	67.82	0.24	0.00	8.13	0.17
1971	Surface	53.90	0.37	38.92	77.26	1.70	0.02	7.72	0.14
1972	Surface	52.81	0.36	25.14	49.90	8.81	0.12	20.09	0.38
1973	Surface	65.25	0.45	16.19	32.14	7.65	0.13	24.02	0.37
1974	Surface	85.25	0.58	40.57	80.54	4.02	0.09	26.47	0.31
1975	Surface	105.55	0.72	70.21	139.37	6.18	0.13	39.64	0.38
1976	Surface	132.95	0.91	60.51	120.12	12.24	0.13	53.54	0.40
1977	Surface	168.98	1.15	78.11	155.06	17.51	0.13	19.11	0.11
1978	Surface	164.09	1.12	101.78	202.05	24.00	0.14	18.39	0.11
1979	Surface	162.14	1.11	63.97	126.99	20.34	0.11	-6.35	-0.04
1980	Surface	149.98	1.02	85.68	170.08	5.82	0.04	-1.08	-0.01
1981	Surface	136.85	0.94	54.75	108.69	12.05	0.08	-13.41	-0.10
1982	Surface	110.61	0.76	101.03	200.55	12.83	0.10	-18.76	-0.17
1983	Surface	74.63	0.51	66.20	131.42	17.22	0.20	-1.74	-0.02
1984	Surface	61.85	0.42	26.05	51.72	11.04	0.18	19.34	0.31
1985	Surface	74.16	0.51	25.02	49.68	7.03	0.14	17.30	0.23
1986	Surface	90.87	0.62	41.58	82.53	0.59	0.00	7.40	0.08
1987	Surface	88.92	0.61	41.74	82.85	9.35	0.19	25.55	0.29
1988	Dive	106.25	0.73	24.98	44.92	8.22	0.14	1.47	0.01
1989	Dive	99.36	0.68	66.05	118.81	8.37	0.13	20.32	0.20
1990	Dive	111.55	0.76	67.15	120.78	8.12	0.12	25.37	0.23
1991	Dive	125.82	0.86	45.83	82.43	11.10	0.15	44.68	0.36

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1992	Dive	156.81	1.07	82.71	148.77	13.70	0.14	19.63	0.13
1993	Dive	162.69	1.11	90.20	162.24	13.74	0.14	-4.28	-0.03
1994	Dive	140.68	0.96	67.14	120.76	17.74	0.17	-1.50	-0.01
1995	Dive	125.99	0.86	64.90	116.73	13.19	0.12	21.96	0.17
1996	Dive	133.83	0.91	71.33	128.29	14.11	0.12	26.78	0.20
1997	Dive	144.79	0.99	58.18	104.65	15.82	0.15	6.09	0.04
1998	Dive	137.28	0.94	74.62	134.21	13.60	0.15	1.30	0.01
1999	Dive	125.38	0.86	85.09	153.06	13.20	0.12	17.91	0.14
2000	Dive	128.09	0.88	72.69	130.74	15.20	0.13	58.86	0.46
2001	Dive	170.57	1.17	100.25	180.32	16.38	0.14	33.20	0.19
2002	Dive	185.15	1.27	117.86	212.00	18.61	0.15	21.70	0.12
2003	Dive	185.98	1.27	152.15	273.67	20.88	0.14	-20.38	-0.11
2004	Dive	152.00	1.04	122.84	220.95	13.60	0.09	-0.36	0.00
2005	Dive	132.77	0.91	102.76	184.84	18.87	0.14	-0.47	0.00
2006	Dive	113.53	0.78	50.26	90.40	18.76	0.17	12.08	0.11
2007	Dive	115.39	0.79	38.52	69.29	10.22	0.12	-30.38	-0.26
2008	Dive	75.00	0.51	34.51	62.07	10.00	0.11	16.99	0.23
2009	Dive	81.82	0.56	53.65	96.50	10.17	0.13	0.49	0.01
2010	Dive	73.99	0.51	50.45	90.75	8.32	0.13	46.39	0.63
2011	Dive	115.25	0.79	85.00	152.89	5.13	0.11	21.59	0.19
2012	Dive	125.50	0.86	52.64	94.68	11.34	0.12	17.68	0.14
2013	Dive	126.63	0.87	83.69	150.54	16.55	0.13	49.17	0.39
2014	Dive	155.50	1.06	120.47	216.69	20.31	0.13	24.82	0.16
2015	Dive	160.34	1.10	104.48	187.93	19.97	0.12	73.72	0.46
2016	Dive	212.76	1.45	129.50	232.94	21.31	0.12	*NA*	*NA*

Table 14. Annual estimates of surplus production, production rate, and depletion for the SOG management area for model AM2. LP-LB periods are shaded grey from 1965-1970. Description as for Table 7.

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1951	Surface	57.28	0.52	66.14	74.48	43.80	0.38	46.65	0.81
1952	Surface	58.05	0.53	72.38	81.50	45.89	0.40	30.71	0.53
1953	Surface	80.34	0.73	111.31	125.33	8.43	0.08	81.62	1.02
1954	Surface	96.19	0.87	82.14	92.49	65.77	0.36	59.14	0.61
1955	Surface	86.69	0.78	69.85	78.66	68.64	0.39	34.00	0.39
1956	Surface	48.63	0.44	25.67	28.90	72.06	0.54	36.47	0.75
1957	Surface	25.50	0.23	24.47	27.55	59.61	0.67	33.60	1.32
1958	Surface	38.47	0.35	16.91	19.04	20.63	0.35	65.37	1.70
1959	Surface	53.81	0.49	47.86	53.89	50.03	0.47	66.99	1.25
1960	Surface	52.77	0.48	55.71	62.73	68.04	0.53	42.08	0.80
1961	Surface	48.63	0.44	44.33	49.91	46.22	0.48	69.18	1.42
1962	Surface	52.51	0.48	35.60	40.08	65.30	0.54	63.27	1.20
1963	Surface	46.93	0.42	37.38	42.09	68.85	0.58	73.69	1.57
1964	Surface	43.75	0.40	35.95	40.48	76.88	0.59	31.28	0.72
1965	Surface	27.21	0.25	38.39	43.23	47.82	0.56	20.75	0.76
1966	Surface	14.62	0.13	7.21	8.12	33.34	0.64	24.94	1.71
1967	Surface	8.52	0.08	9.65	10.86	31.04	0.77	7.39	0.87
1968	Surface	14.01	0.13	9.44	10.63	1.89	0.08	4.51	0.32
1969	Surface	18.32	0.17	14.04	15.81	0.19	0.01	9.79	0.53
1970	Surface	27.87	0.25	34.16	38.47	0.24	0.01	7.15	0.26
1971	Surface	33.33	0.30	38.92	43.83	1.70	0.03	6.95	0.21
1972	Surface	31.46	0.28	25.14	28.31	8.81	0.18	13.39	0.43
1973	Surface	37.21	0.34	16.19	18.23	7.65	0.20	14.47	0.39
1974	Surface	47.66	0.43	40.57	45.68	4.02	0.15	17.02	0.36
1975	Surface	58.50	0.53	70.21	79.05	6.18	0.20	25.27	0.43
1976	Surface	71.53	0.65	60.51	68.14	12.24	0.22	32.84	0.46
1977	Surface	86.86	0.79	78.11	87.96	17.51	0.22	17.88	0.21
1978	Surface	80.74	0.73	101.78	114.61	24.00	0.26	18.01	0.22
1979	Surface	78.41	0.71	63.97	72.03	20.34	0.21	5.08	0.06
1980	Surface	77.67	0.70	85.68	96.47	5.82	0.08	8.17	0.11
1981	Surface	73.79	0.67	54.75	61.65	12.05	0.14	0.44	0.01
1982	Surface	61.39	0.56	101.03	113.75	12.83	0.18	-3.76	-0.06
1983	Surface	40.41	0.37	66.20	74.54	17.22	0.32	3.56	0.09
1984	Surface	32.94	0.30	26.05	29.34	11.04	0.31	14.44	0.44
1985	Surface	40.35	0.37	25.02	28.18	7.03	0.24	14.25	0.35
1986	Surface	54.00	0.49	41.58	46.81	0.59	0.01	8.11	0.15
1987	Surface	52.76	0.48	41.74	47.00	9.35	0.28	20.15	0.38
1988	Dive	64.69	0.59	24.98	25.01	8.22	0.21	4.52	0.07
1989	Dive	60.84	0.55	66.05	66.15	8.37	0.20	15.98	0.26
1990	Dive	68.70	0.62	67.15	67.25	8.12	0.18	18.62	0.27
1991	Dive	76.22	0.69	45.83	45.90	11.10	0.23	31.29	0.41

Year	Period	*B*	*D*	*I*	*I/q*	*C*	*U*	*P*	*P/B*
1992	Dive	93.81	0.85	82.71	82.83	13.70	0.22	16.78	0.18
1993	Dive	96.85	0.88	90.20	90.33	13.74	0.21	3.91	0.04
1994	Dive	83.02	0.75	67.14	67.24	17.74	0.25	4.83	0.06
1995	Dive	74.66	0.68	64.90	64.99	13.19	0.19	16.64	0.22
1996	Dive	77.19	0.70	71.33	71.43	14.11	0.19	20.74	0.27
1997	Dive	82.11	0.74	58.18	58.27	15.82	0.23	9.67	0.12
1998	Dive	78.17	0.71	74.62	74.73	13.60	0.23	6.86	0.09
1999	Dive	71.83	0.65	85.09	85.22	13.20	0.20	14.50	0.20
2000	Dive	71.13	0.64	72.69	72.80	15.20	0.22	37.44	0.53
2001	Dive	92.18	0.83	100.25	100.40	16.38	0.22	25.00	0.27
2002	Dive	98.57	0.89	117.86	118.04	18.61	0.24	21.20	0.22
2003	Dive	98.90	0.89	152.15	152.38	20.88	0.24	-2.13	-0.02
2004	Dive	83.16	0.75	122.84	123.02	13.60	0.16	7.78	0.09
2005	Dive	72.07	0.65	102.76	102.92	18.87	0.23	6.27	0.09
2006	Dive	59.58	0.54	50.26	50.33	18.76	0.28	13.49	0.23
2007	Dive	62.84	0.57	38.52	38.58	10.22	0.20	-11.15	-0.18
2008	Dive	41.69	0.38	34.51	34.56	10.00	0.19	13.46	0.32
2009	Dive	44.98	0.41	53.65	53.73	10.17	0.21	2.77	0.06
2010	Dive	39.42	0.36	50.45	50.53	8.32	0.22	27.31	0.69
2011	Dive	61.60	0.56	85.00	85.13	5.13	0.18	16.88	0.27
2012	Dive	67.14	0.61	52.64	52.71	11.34	0.21	15.19	0.23
2013	Dive	65.78	0.60	83.69	83.82	16.55	0.23	33.15	0.50
2014	Dive	78.62	0.71	120.47	120.65	20.31	0.23	21.89	0.28
2015	Dive	80.54	0.73	104.48	104.64	19.97	0.23	50.36	0.63
2016	Dive	109.59	0.99	129.50	129.70	21.31	0.22	NA	NA

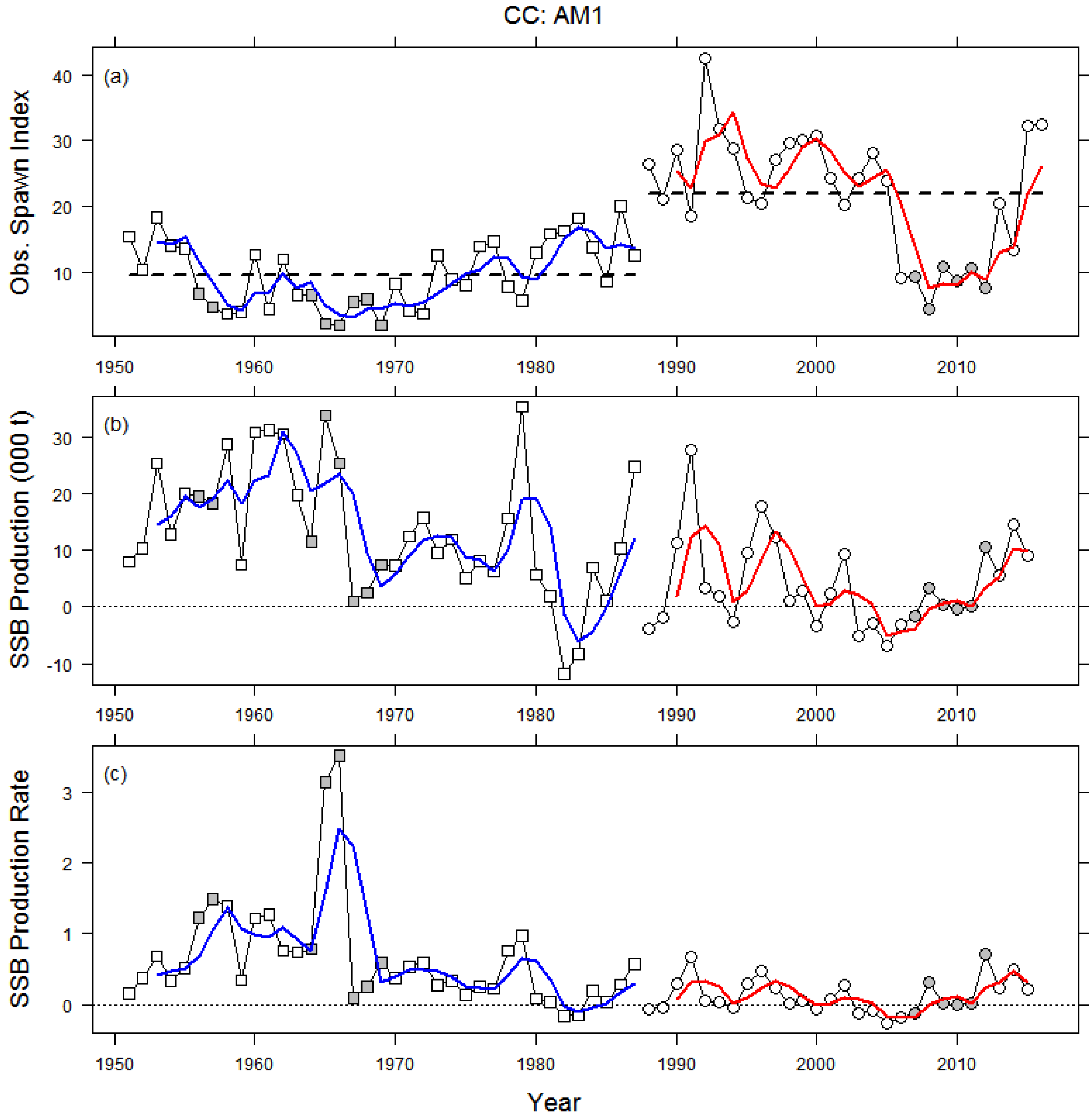

Figure 22. Time series of observed spawn indices (panel a), spawning stock biomass (SSB) production (panel b), and SSB production rate (panel c) for CC Pacific Herring based on model AM1.

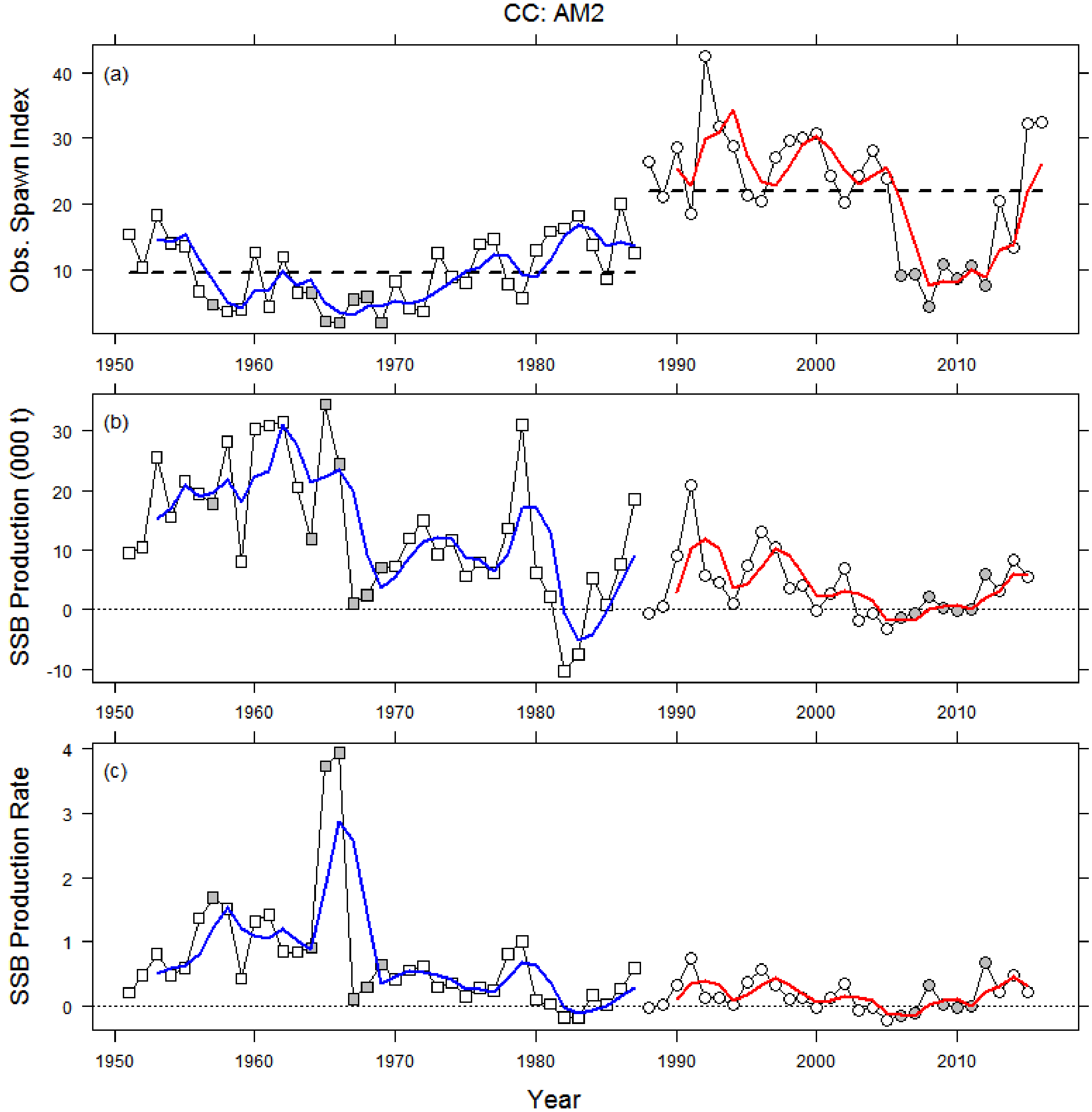

Figure 23. Time series of observed spawn indices (panel a), spawning stock biomass (SSB) production (panel b), and SSB production rate (panel c) for CC Pacific Herring based on model AM2.

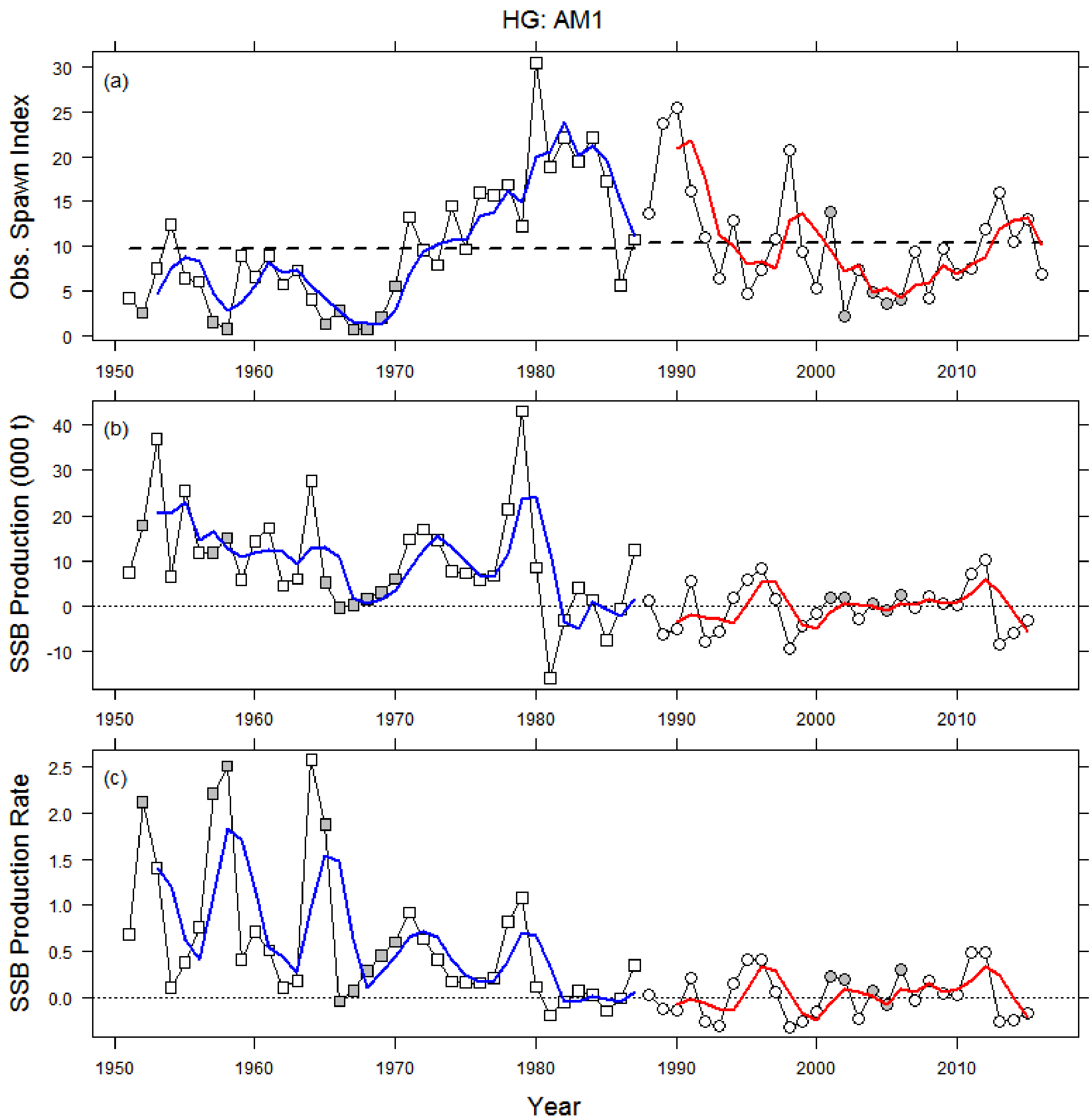

Figure 24. Time series of observed spawn indices (panel a), spawning stock biomass (SSB) production (panel b), and SSB production rate (panel c) for HG Pacific Herring based on model AM1.

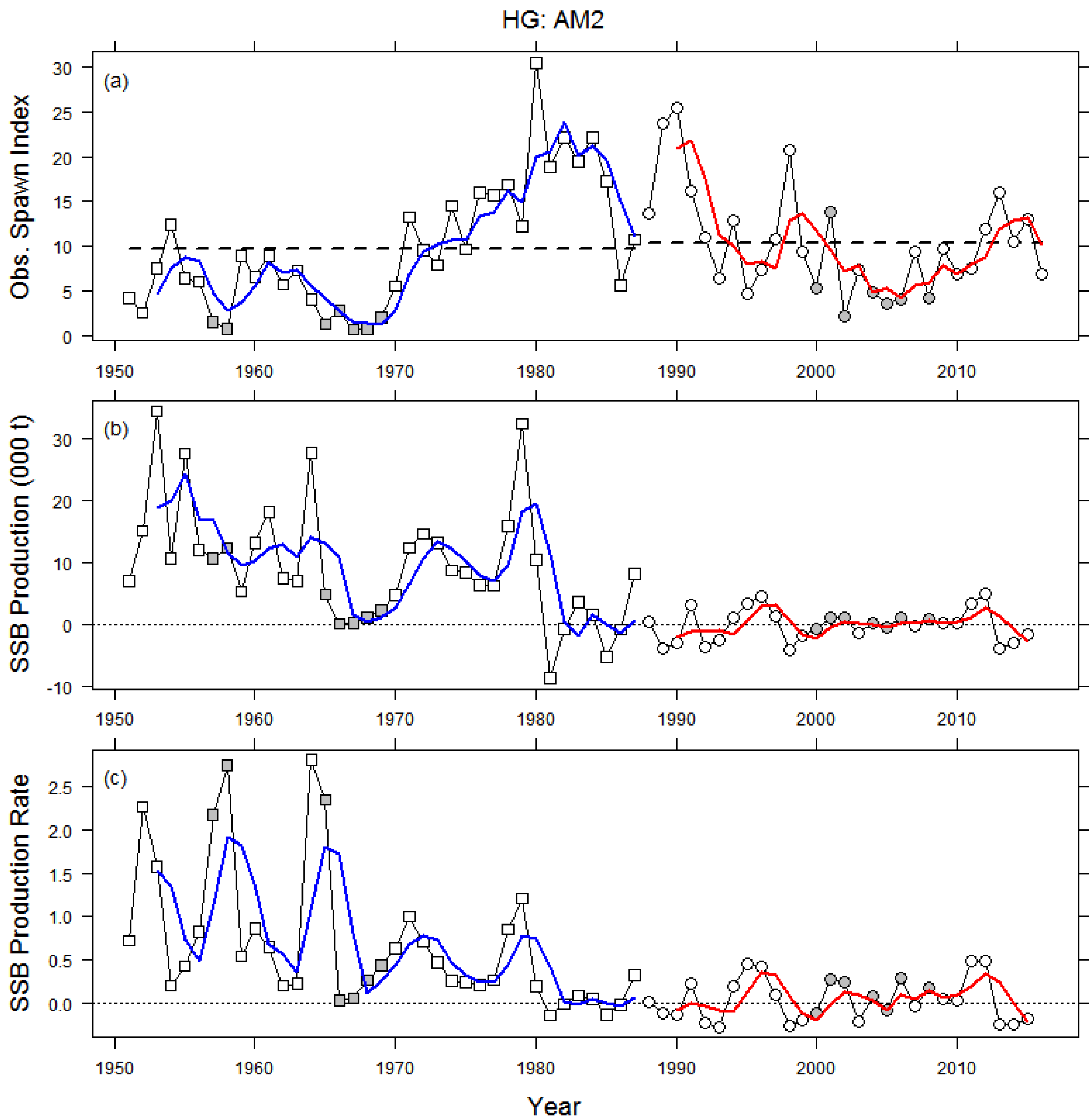

Figure 25. Time series of observed spawn indices (panel a), spawning stock biomass (SSB) production (panel b), and SSB production rate (panel c) for HG Pacific Herring based on model AM2.

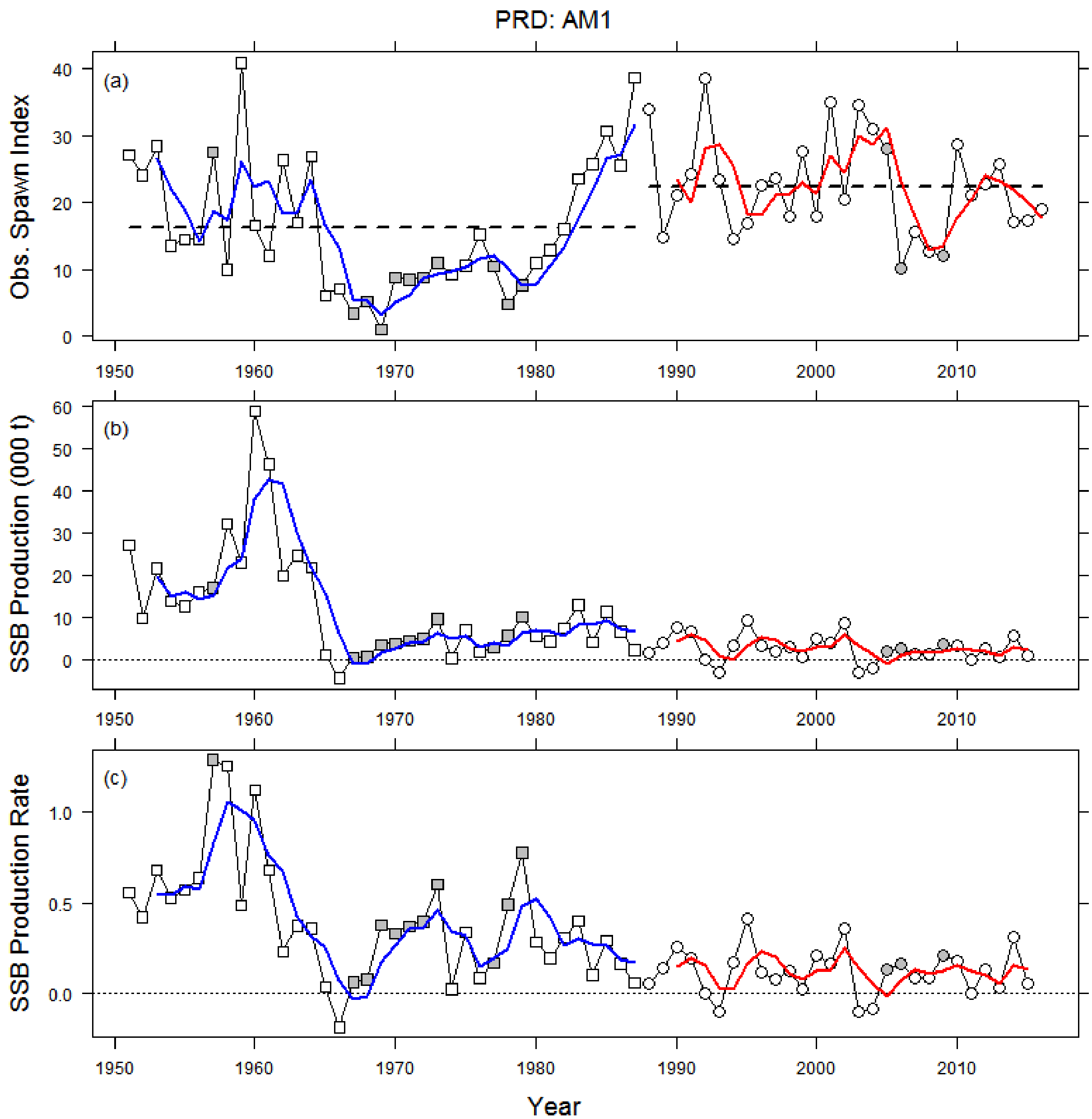

Figure 26. Time series of observed spawn indices (panel a), spawning stock biomass (SSB) production (panel b), and SSB production rate (panel c) for PRD Pacific Herring based on model AM1.

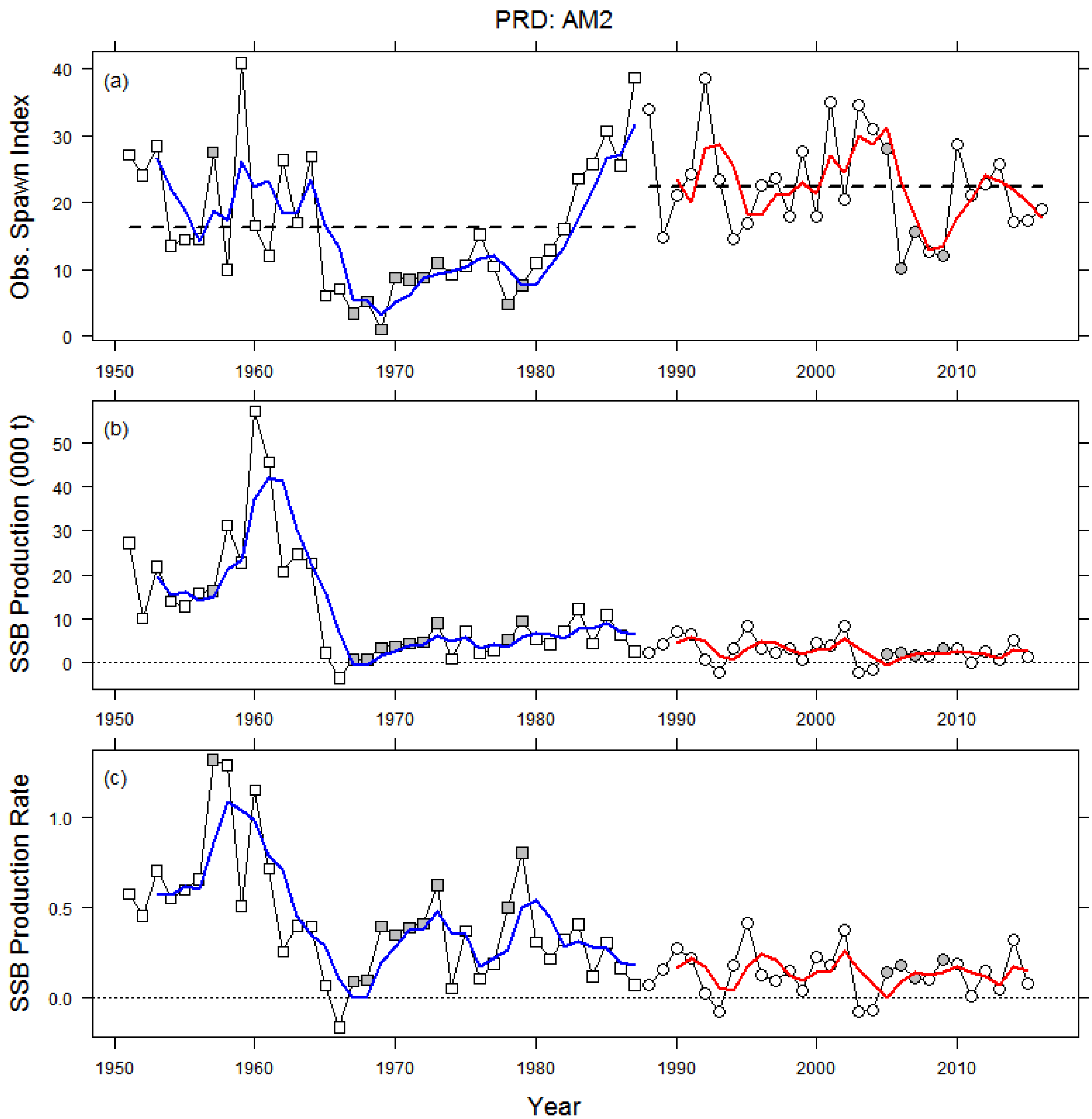

Figure 27. Time series of observed spawn indices (panel a), spawning stock biomass (SSB) production (panel b), and SSB production rate (panel c) for PRD Pacific Herring based on model AM2.

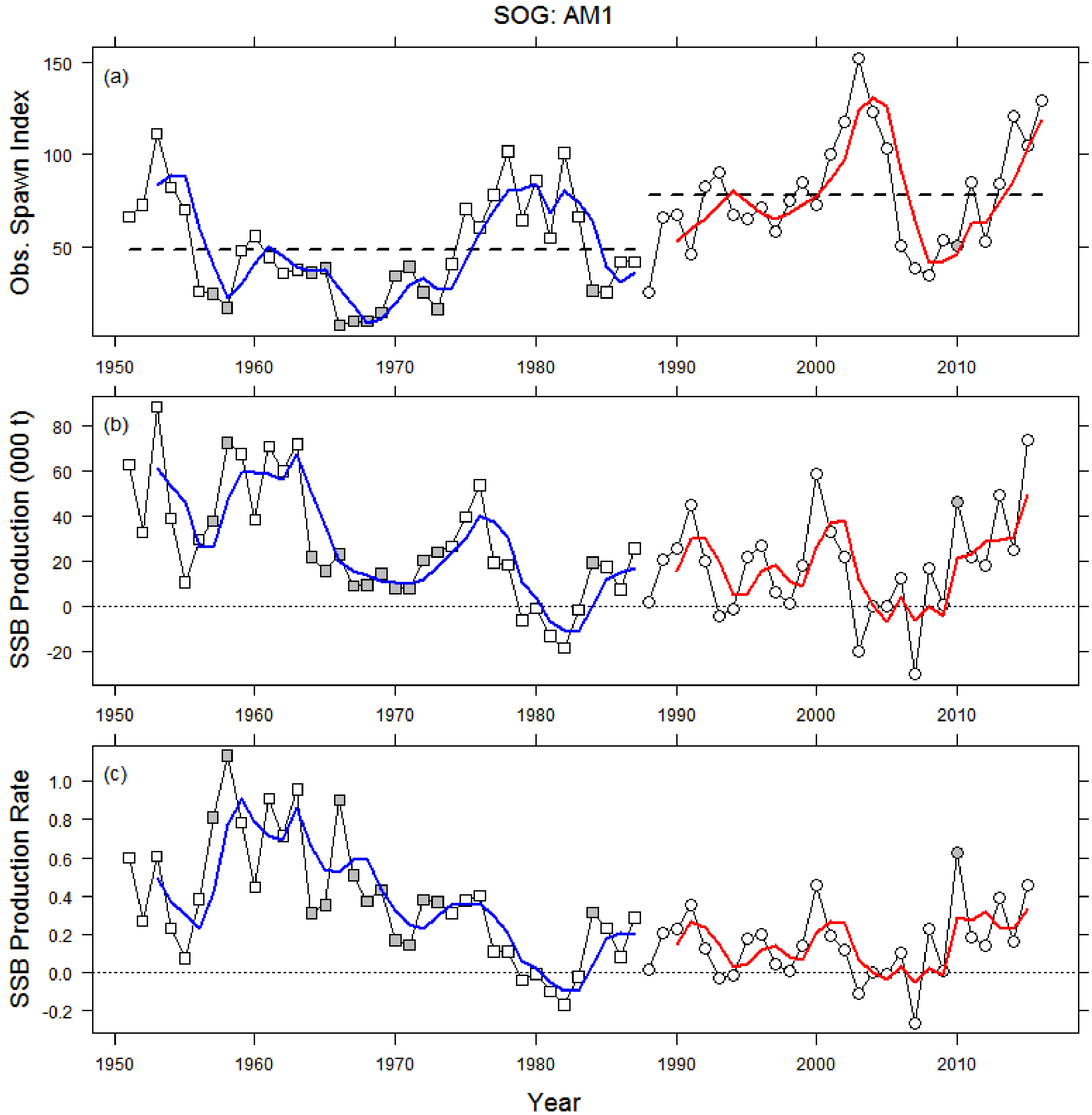

Figure 28. Time series of observed spawn indices (panel a), spawning stock biomass (SSB) production (panel b), and SSB production rate (panel c) for SOG Pacific Herring based on model AM1.

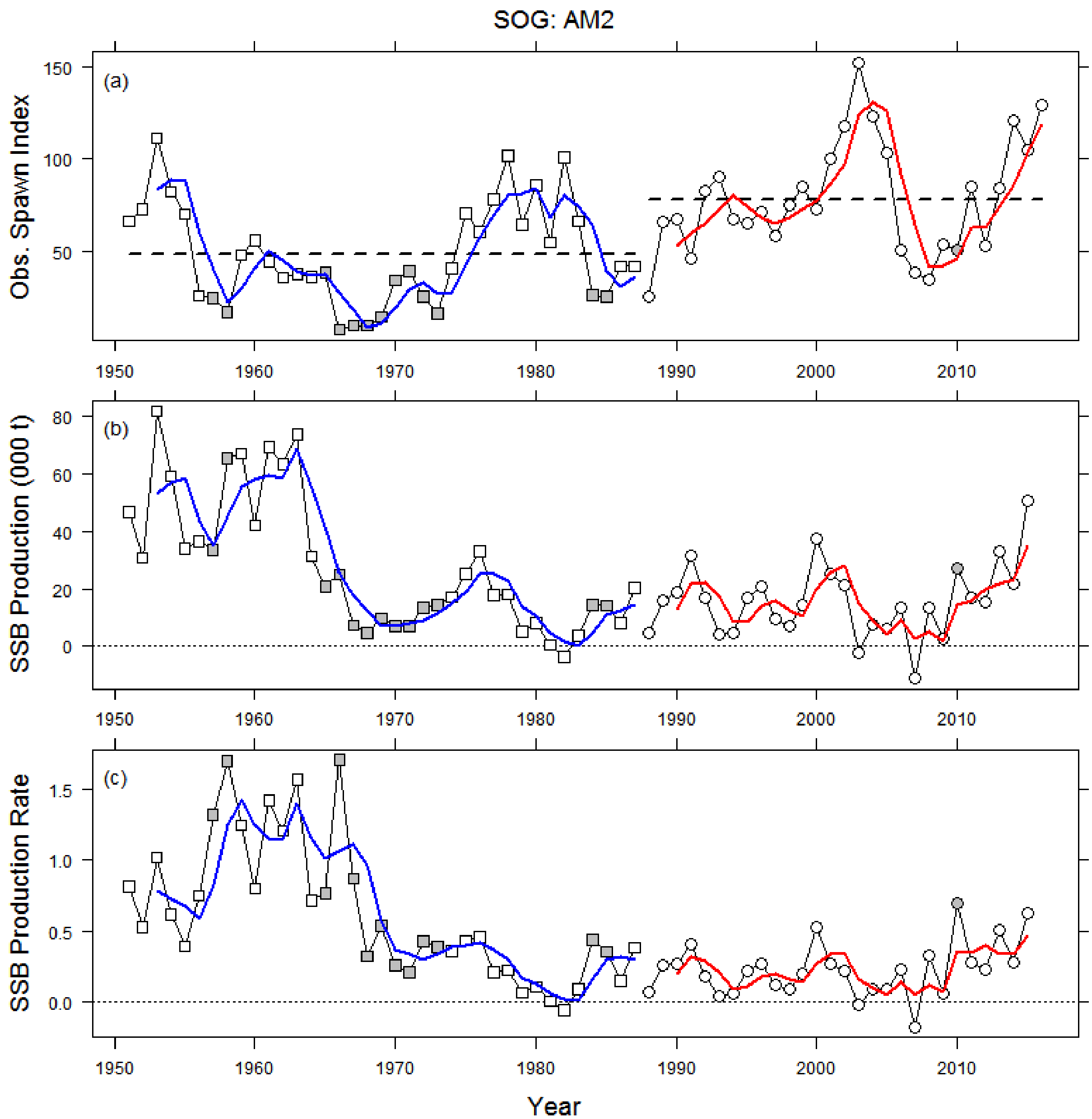

Figure 29. Time series of observed spawn indices (panel a), spawning stock biomass (SSB) production (panel b), and SSB production rate (panel c) for SOG Pacific Herring based on model AM2.

APPENDIX C – PRODUCTION PHASE PLOTS

Surplus production analysis of Pacific Herring for the CC, HG, PRD, and SOG management areas based on catch data and assessment model estimates (maximum posterior density (MPD) estimates) of spawning stock biomass (SSB) for models AM1 and AM2. Figure 30 through Figure 37 show results for models AM1 and AM2 for each stock in CC, HG, PRD, and SOG management areas. Results for the WCVI management area appear in the main body of the paper.

Each figures shows phase plots of SSB production (upper panels) and SSB production rate (panel c) against SSB for a management area. The time series is broken into two periods corresponding to surface (1951-1987, left panels) and dive (1988-2015, right panels) survey periods. The start and end of each time series is indicated by a black circle and black diamond, respectively. Grey shading of the circles becomes darker from in chronological order. Calendar years are indicated above each symbol. The axis scales at the top of each panel are in units of spawning biomass depletion, i.e., divided by the estimated unfished spawning biomass from the assessment model. Estimated values of surface and dive survey catchability parameters are reported above the left and right panels, respectively. Vertical reference lines are positioned at estimates of $0.1B_0$, (red dashed line) $0.25B_0$ (blue dashed line), $0.3B_0$ (black dashed line), and the 1996 fixed cut-off value (thick blue long dashed line).

Figure 38 through Figure 41 show phase plots of spawning stock biomass (SSB) production against SSB for model AM1 (upper panels) and model AM2 (lower panels) for the CC, HG, PRD, and SOG management area. Panels have been scaled to emphasize the relative positions of points in the lowest 20th percentile of SSB values (grey circles). The first (black circle) and last year (black diamond) of the surface survey (1951-1987) and dive survey periods (1988-2015) are indicated. Vertical reference lines are positioned at the model estimates of $0.1B_0$ (red dotted), $0.25B_0$ (blue dotted), $0.3B_0$ (green dotted) and the fixed cut-off estimated in 1996 (thick blue dashed).

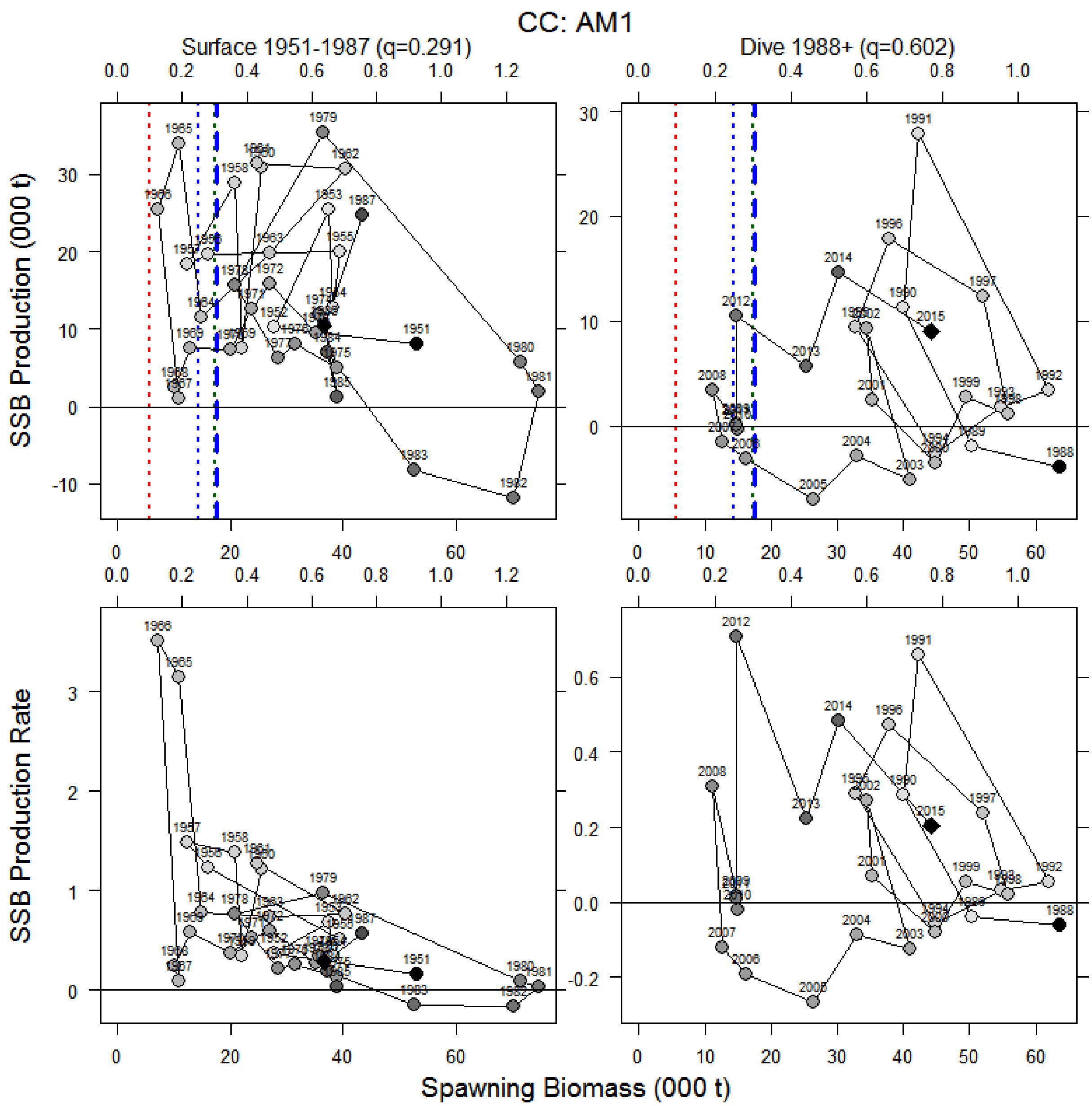

Figure 30. Phase plot of SSB production (upper panels) and SSB production rate (lower panels) against SSB for CC Pacific Herring based on model AM1. See appendix text for description.

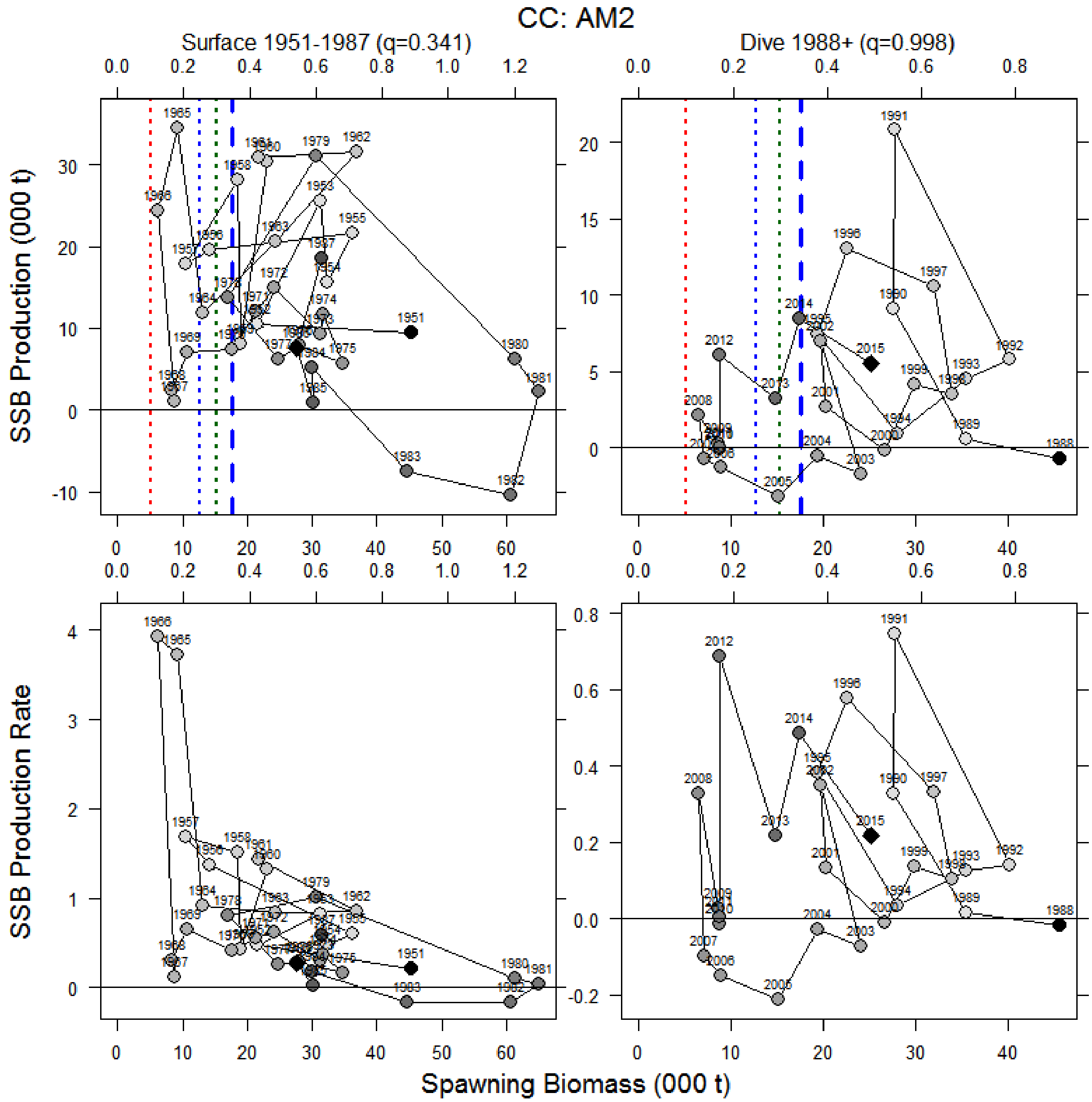

Figure 31. Phase plot of SSB production (upper panels) and SSB production rate (lower panels) against SSB for CC Pacific Herring based on model AM2. See appendix text for description.

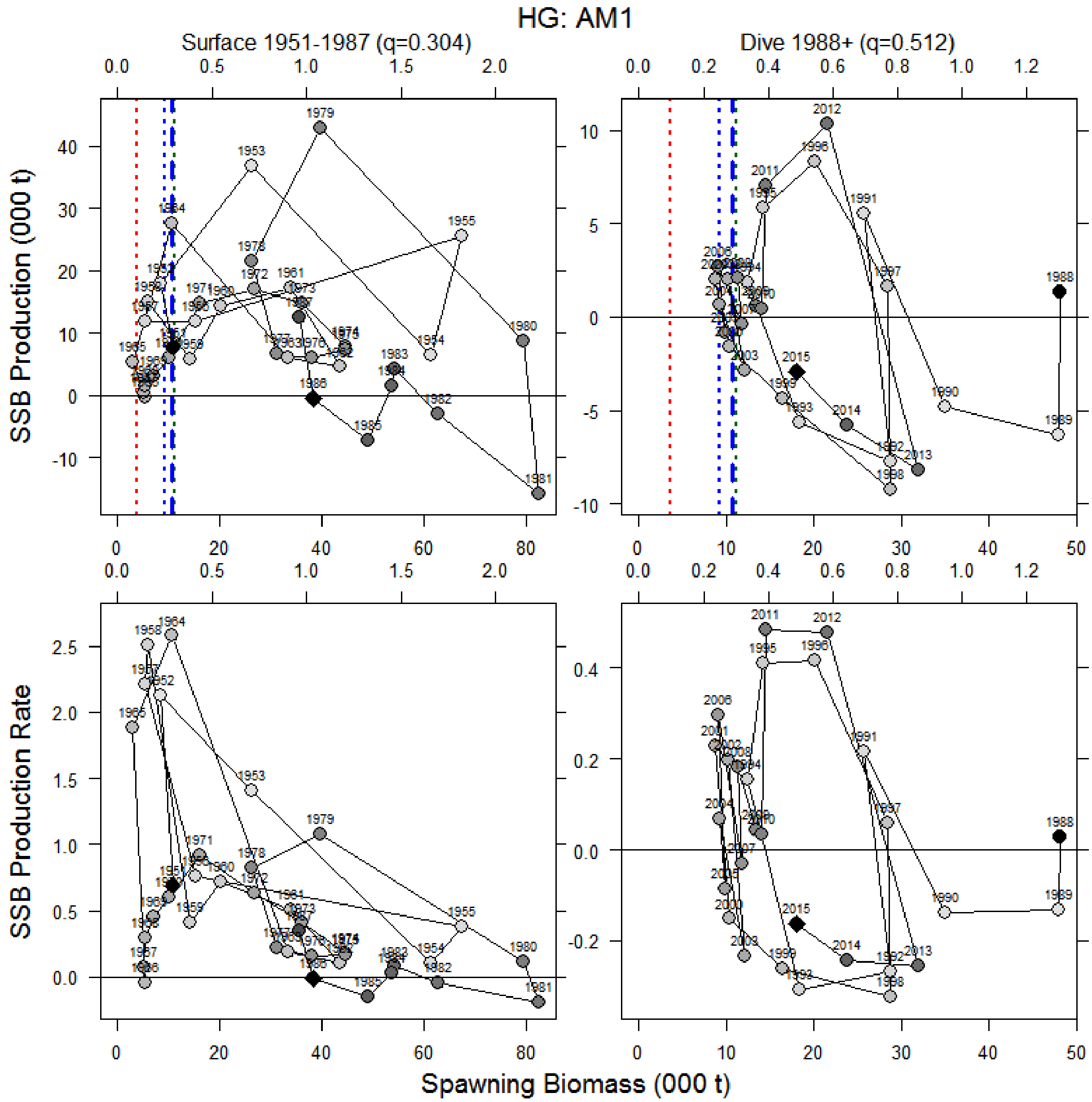

Figure 32. Phase plot of SSB production (upper panels) and SSB production rate (lower panels) against SSB for HG Pacific Herring based on model AM1. See appendix text for description.

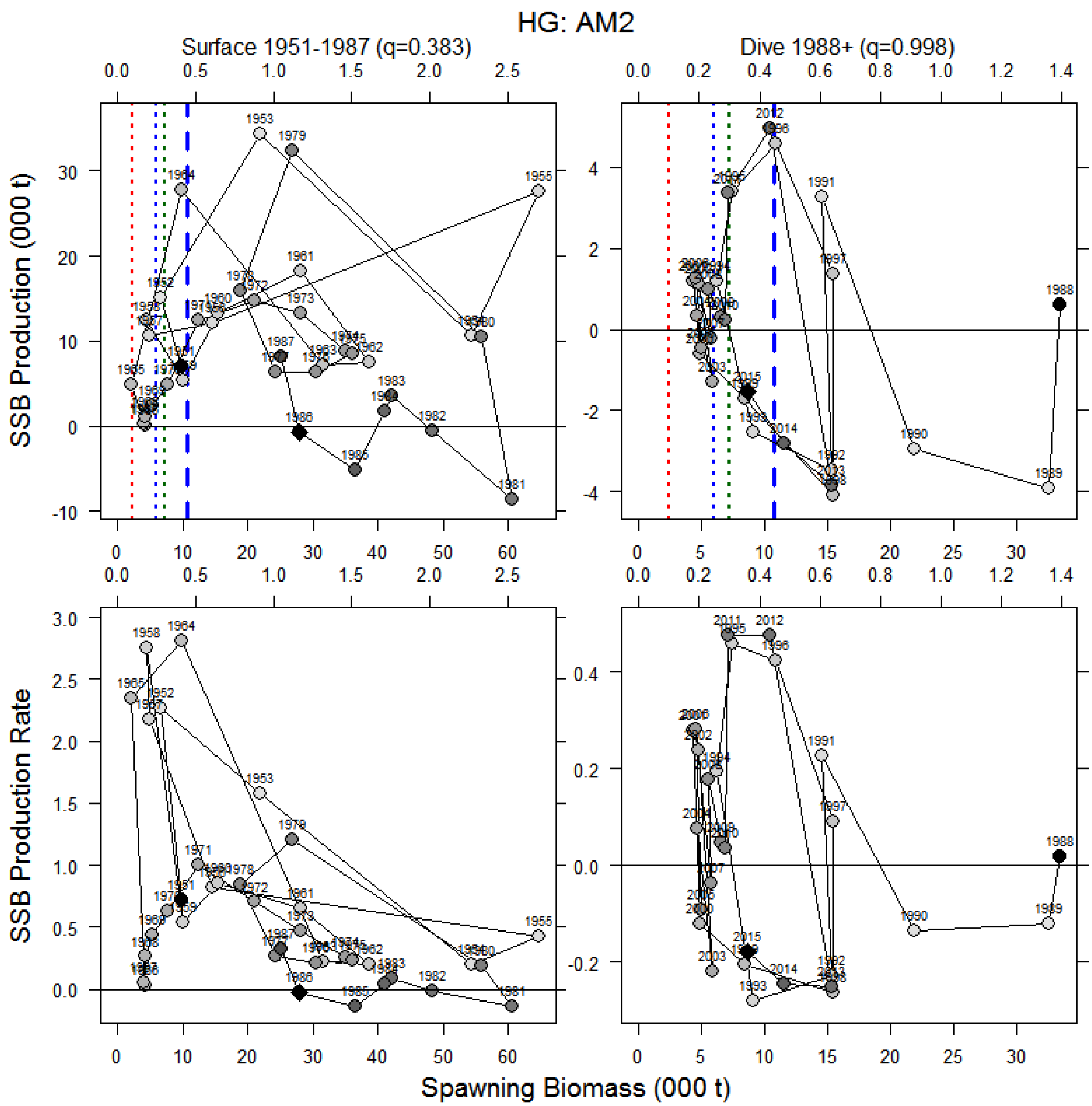

Figure 33. Phase plot of SSB production (upper panels) and SSB production rate (lower panels) against SSB for HG Pacific Herring based on model AM2. See appendix text for description.

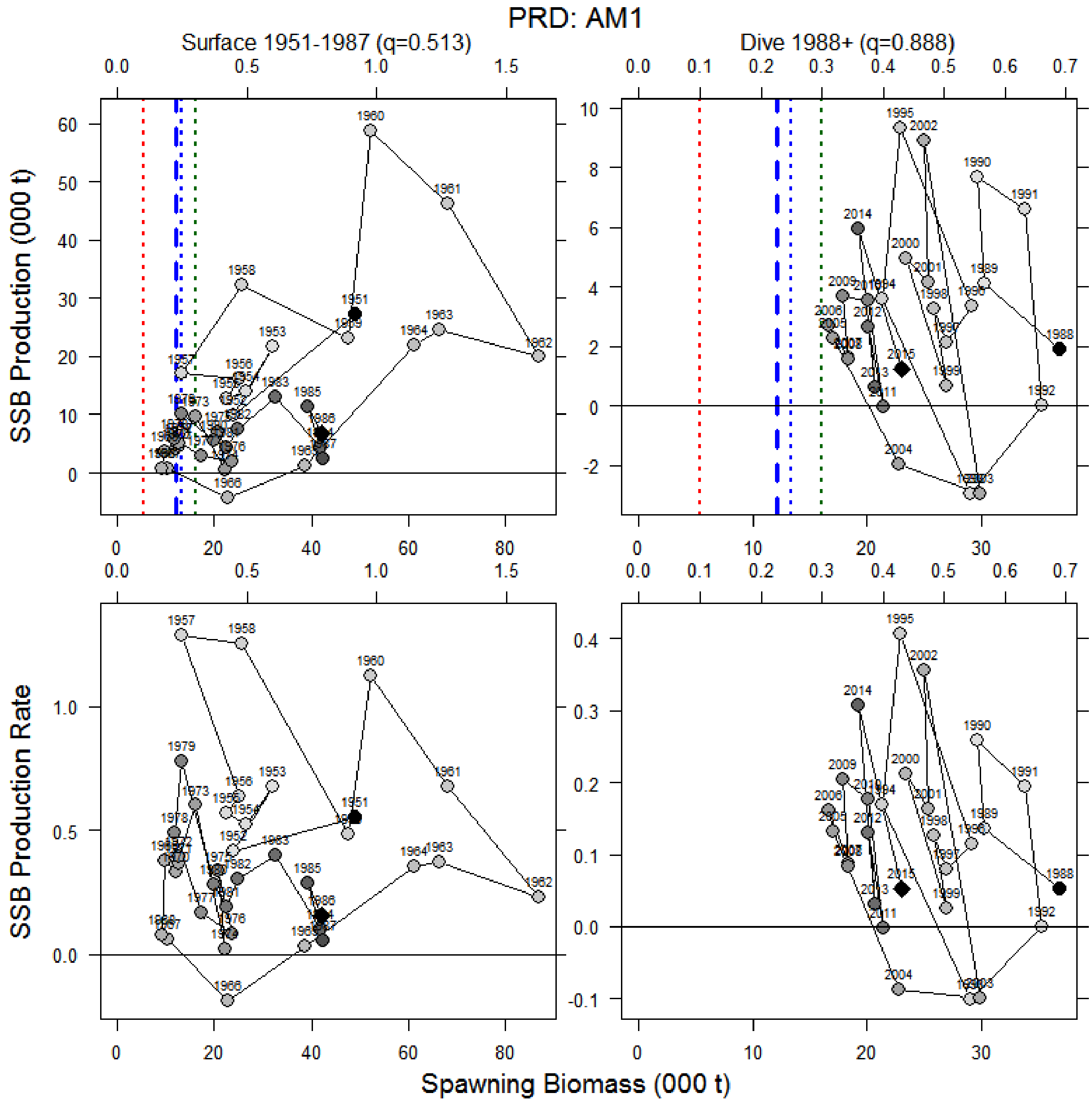

Figure 34. Phase plot of SSB production (upper panels) and SSB production rate (lower panels) against SSB for PRD Pacific Herring based on model AM1. See appendix text for description.

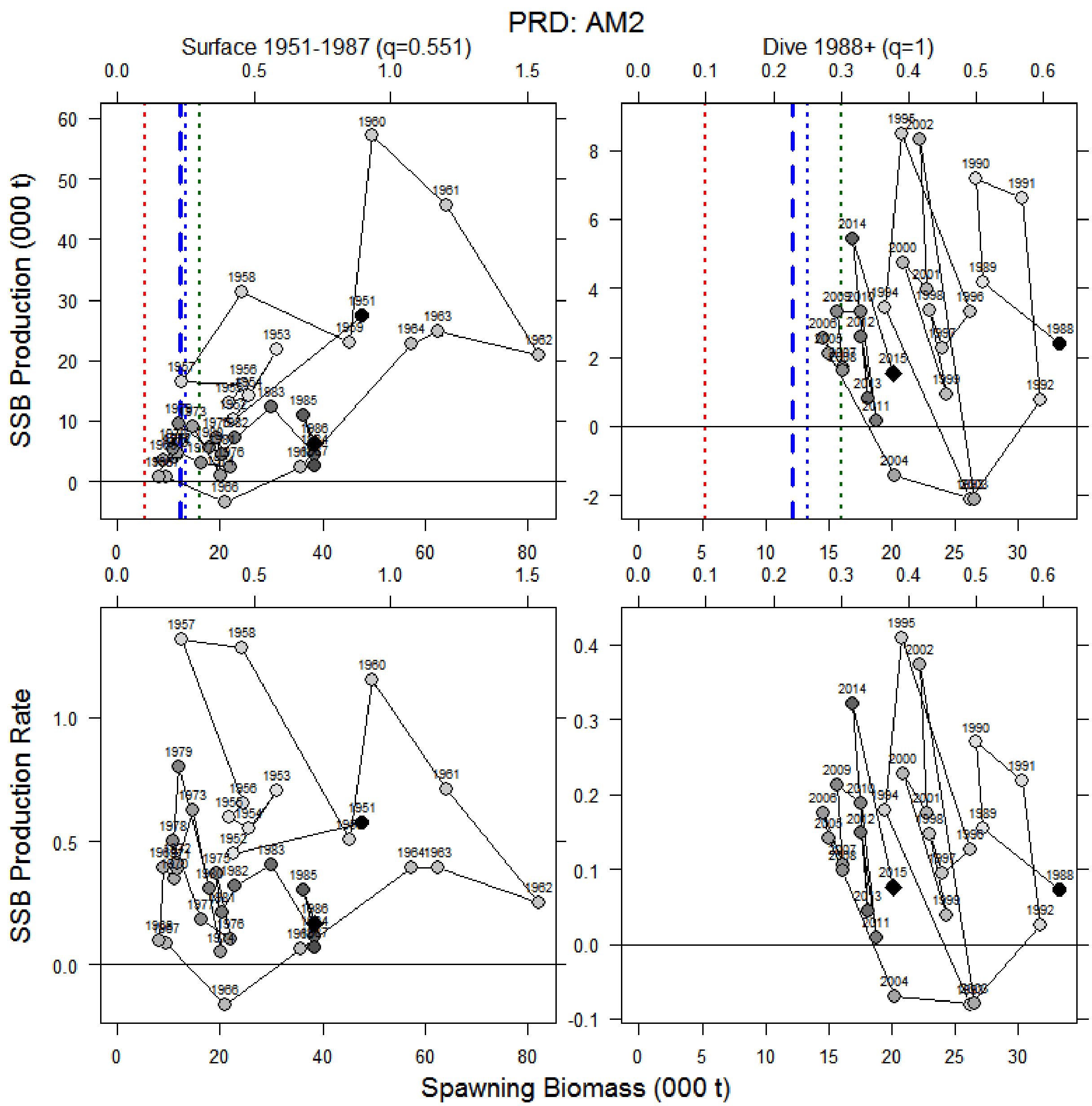

Figure 35. Phase plot of SSB production (upper panels) and SSB production rate (lower panels) against SSB for PRD Pacific Herring based on model AM2. See appendix text for description.

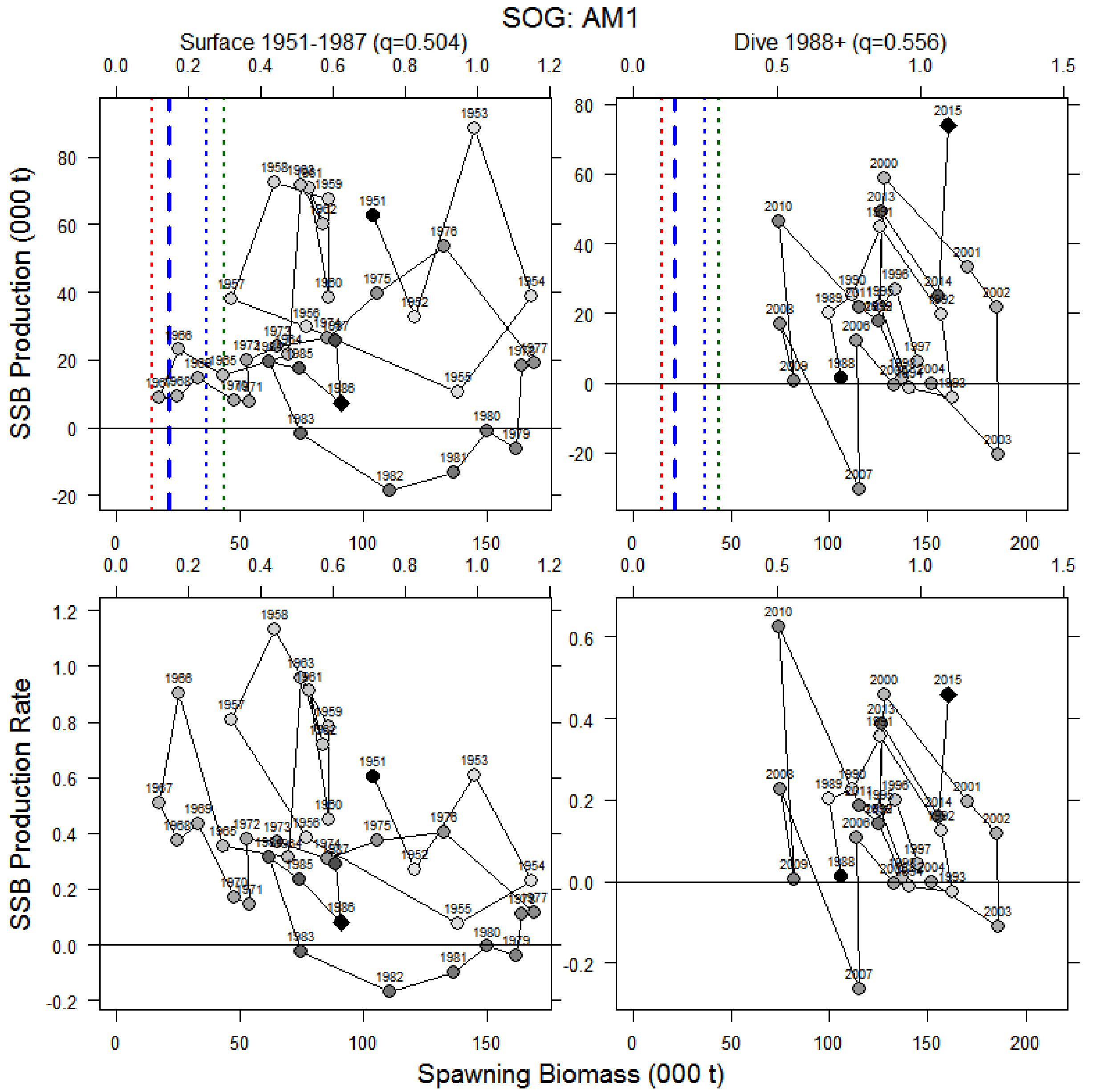

Figure 36. Phase plot of SSB production (upper panels) and SSB production rate (lower panels) against SSB for SOG Pacific Herring based on model AM1. See appendix text for description.

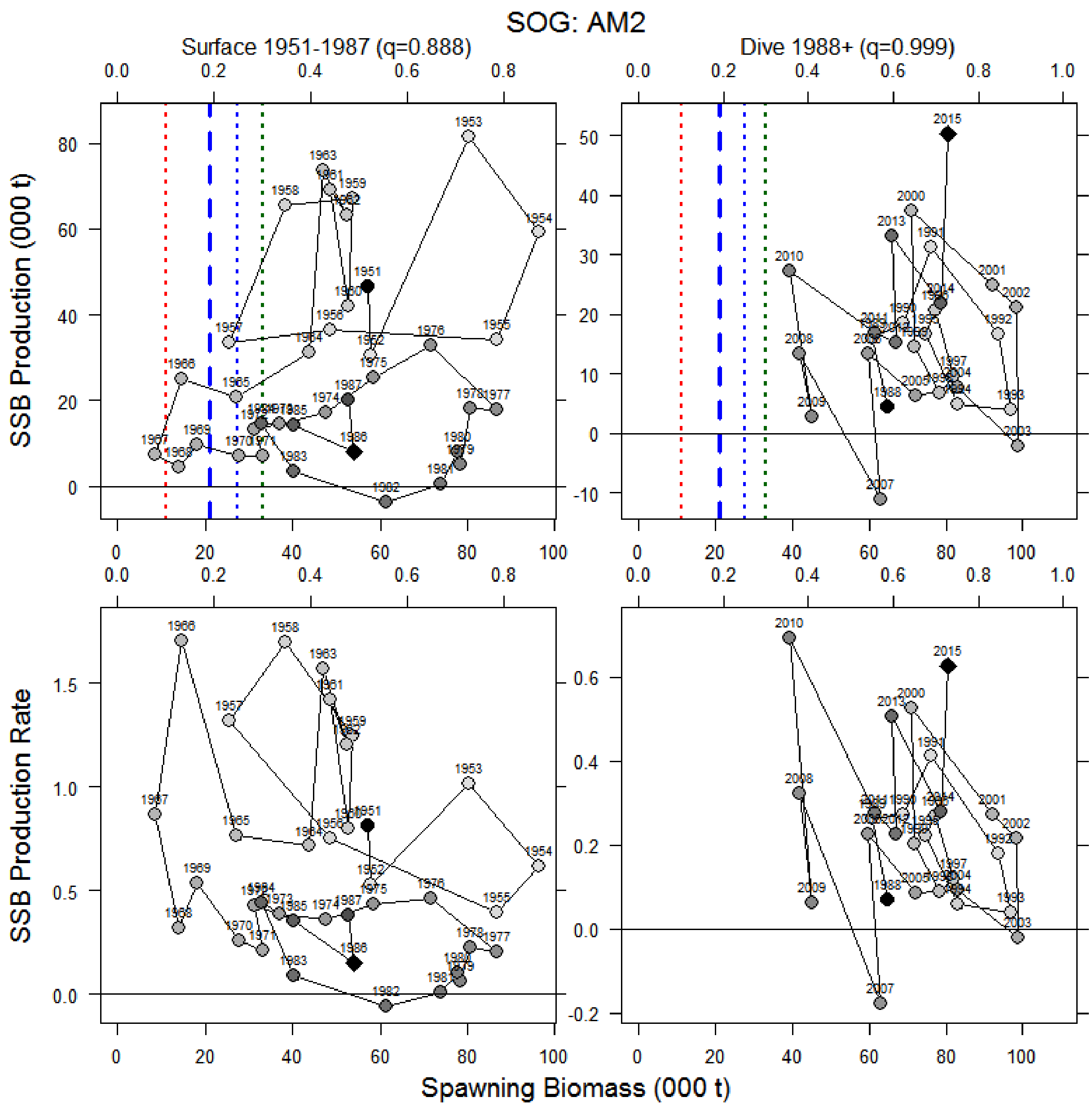

Figure 37. Phase plot of SSB production (upper panels) and SSB production rate (lower panels) against SSB for SOG Pacific Herring based on model AM2. See appendix text for description.

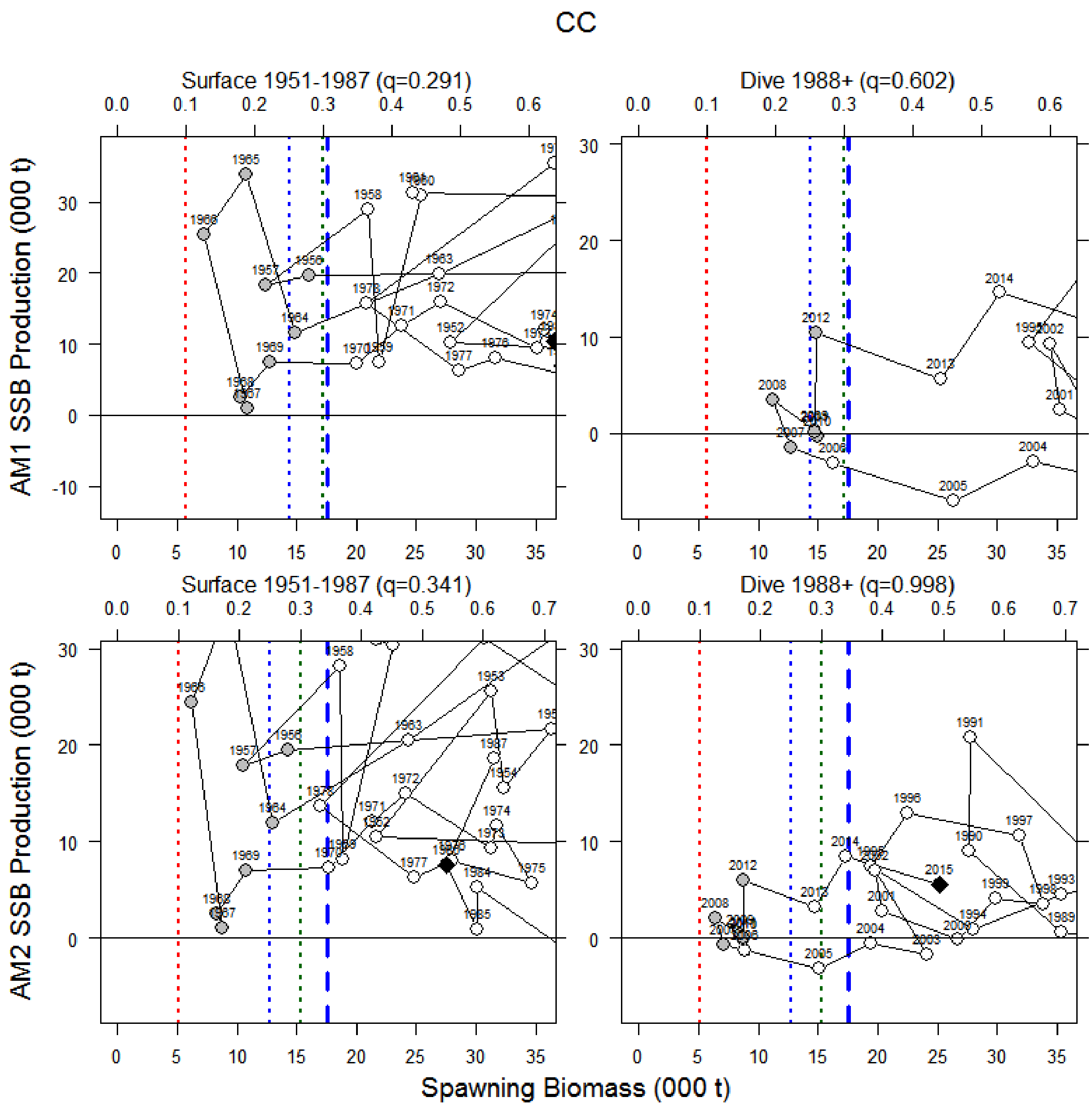

Figure 38. Phase plot of SSB production against SSB for model AM1 (upper panels) and model AM2 (lower panels) for the surface survey (left panels) and dive survey (right panel) periods for the CC management area. See appendix text for description.

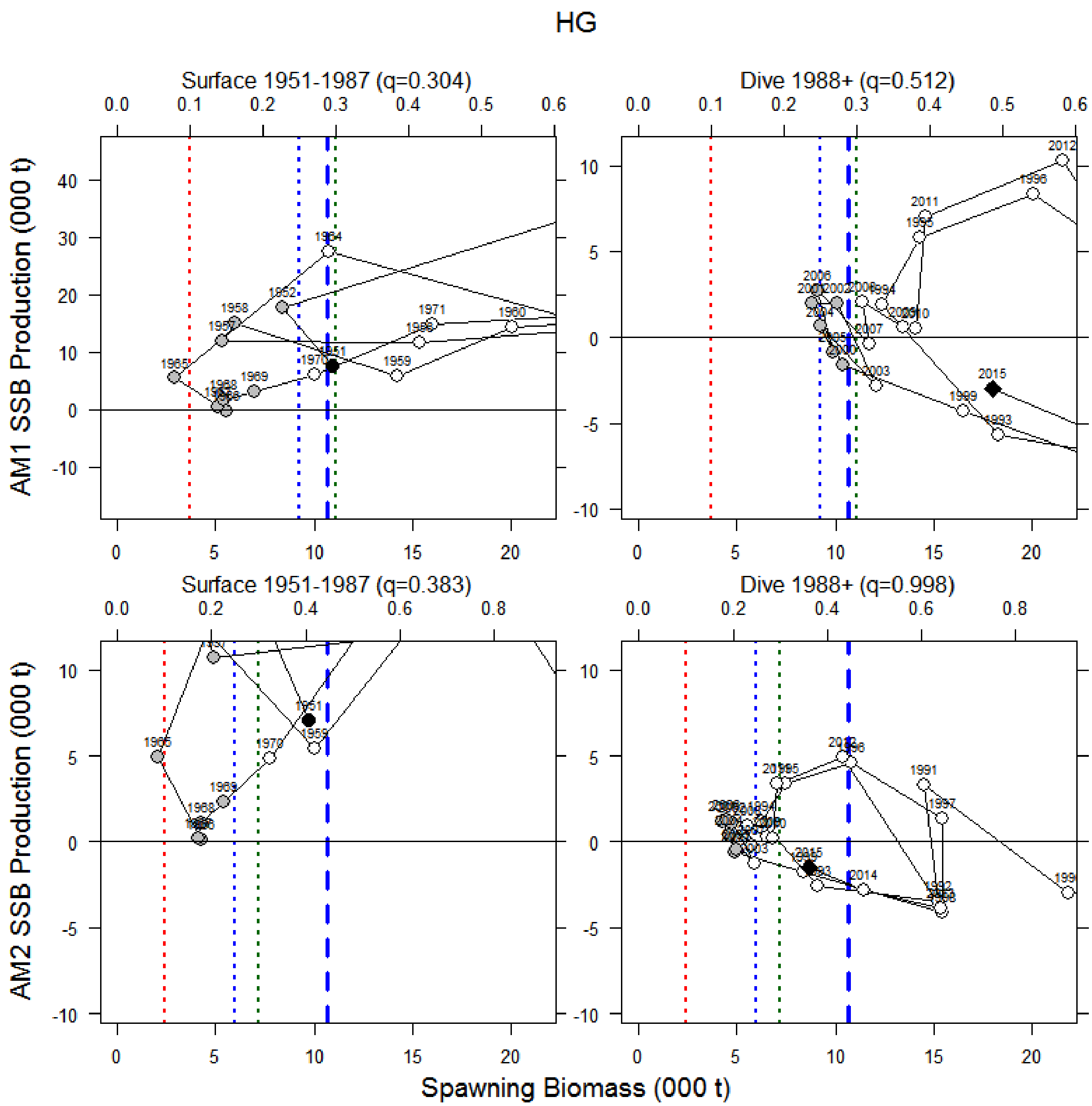

Figure 39. Phase plot of SSB production against SSB for model AM1 (upper panels) and model AM2 (lower panels) for the surface survey (left panels) and dive survey (right panel) periods for the HG management area. See appendix text for description.

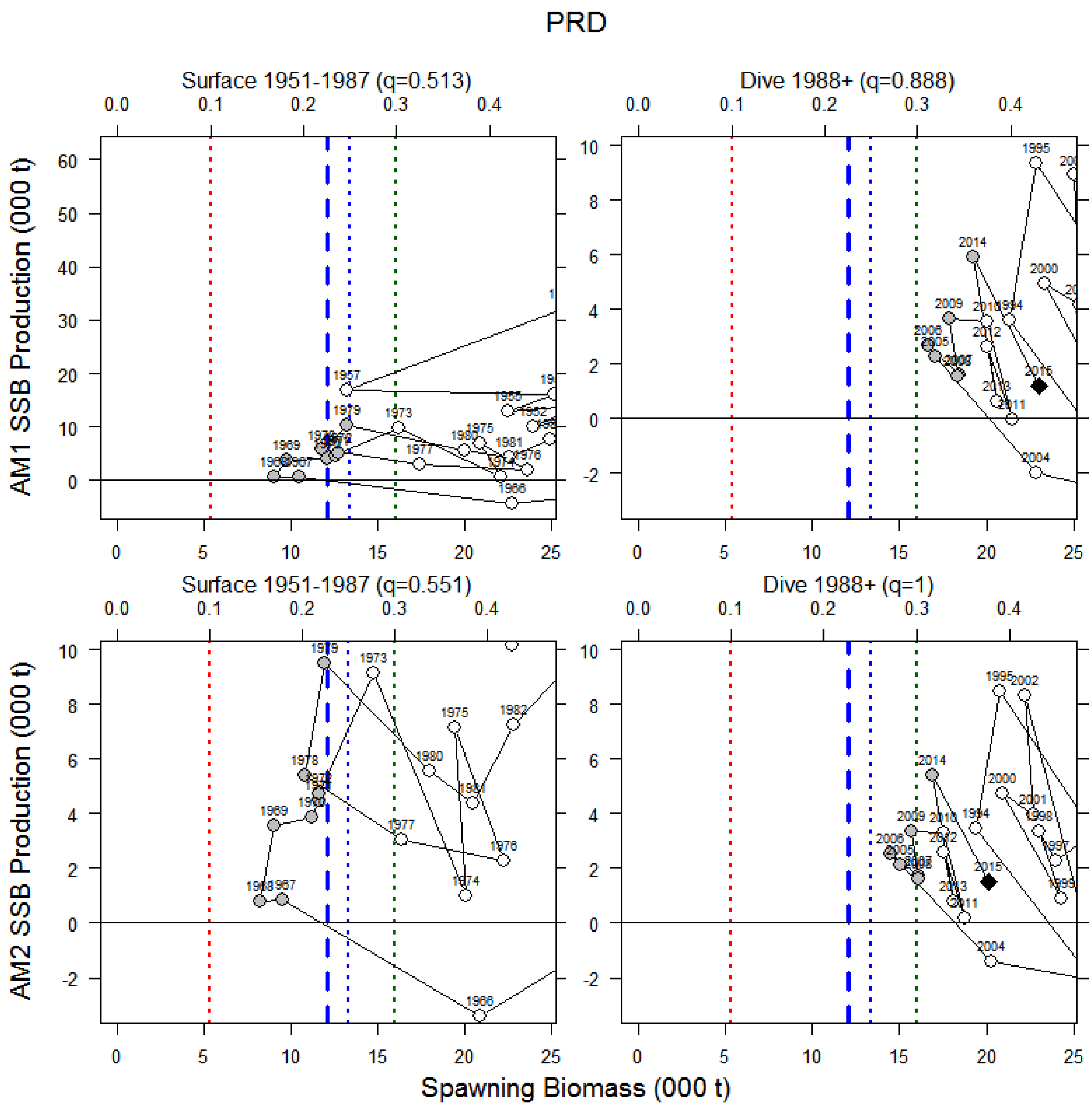

Figure 40. Phase plot of SSB production against SSB for model AM1 (upper panels) and model AM2 (lower panels) for the surface survey (left panels) and dive survey (right panel) periods for the PRD management area. See appendix text for description.

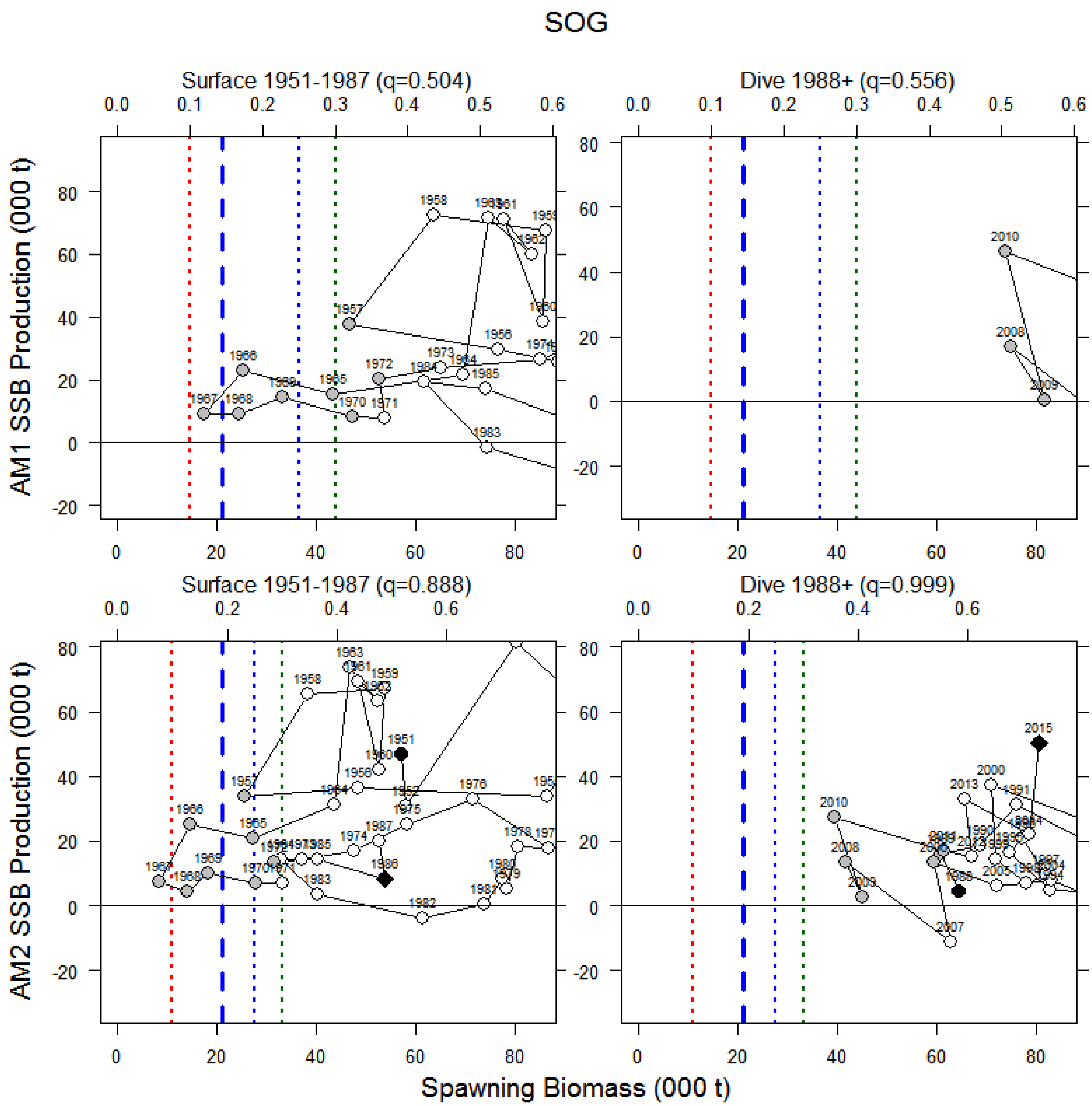

Figure 41. Phase plot of SSB production against SSB for model AM1 (upper panels) and model AM2 (lower panels) for the surface survey (left panels) and dive survey (right panel) periods for the SOG management area. See appendix text for description.

APPENDIX D – SPAWN INDICES, AGE STRUCTURE AND SAMPLE SIZE

Graphical analysis of changes in proportions at age with spawn index values for the CC, HG, PRD, and SOG management areas. Figure 42 through Figure 45 show results for models AM1 and AM2 for each stock in CC, HG, PRD, and SOG management areas. Results for the WCVI management area appear in the main body of the paper.

Each figure show the time series of annual spawn index values for the surface (1951-1987, grey squares) and dive (1988-2016, open circles) survey periods (panel a). Observed annual proportions at age for ages 2-5, and a plus group of ages 6 and older are shown in panel (b). Numbers of fish aged in each year are shown in panel (c).

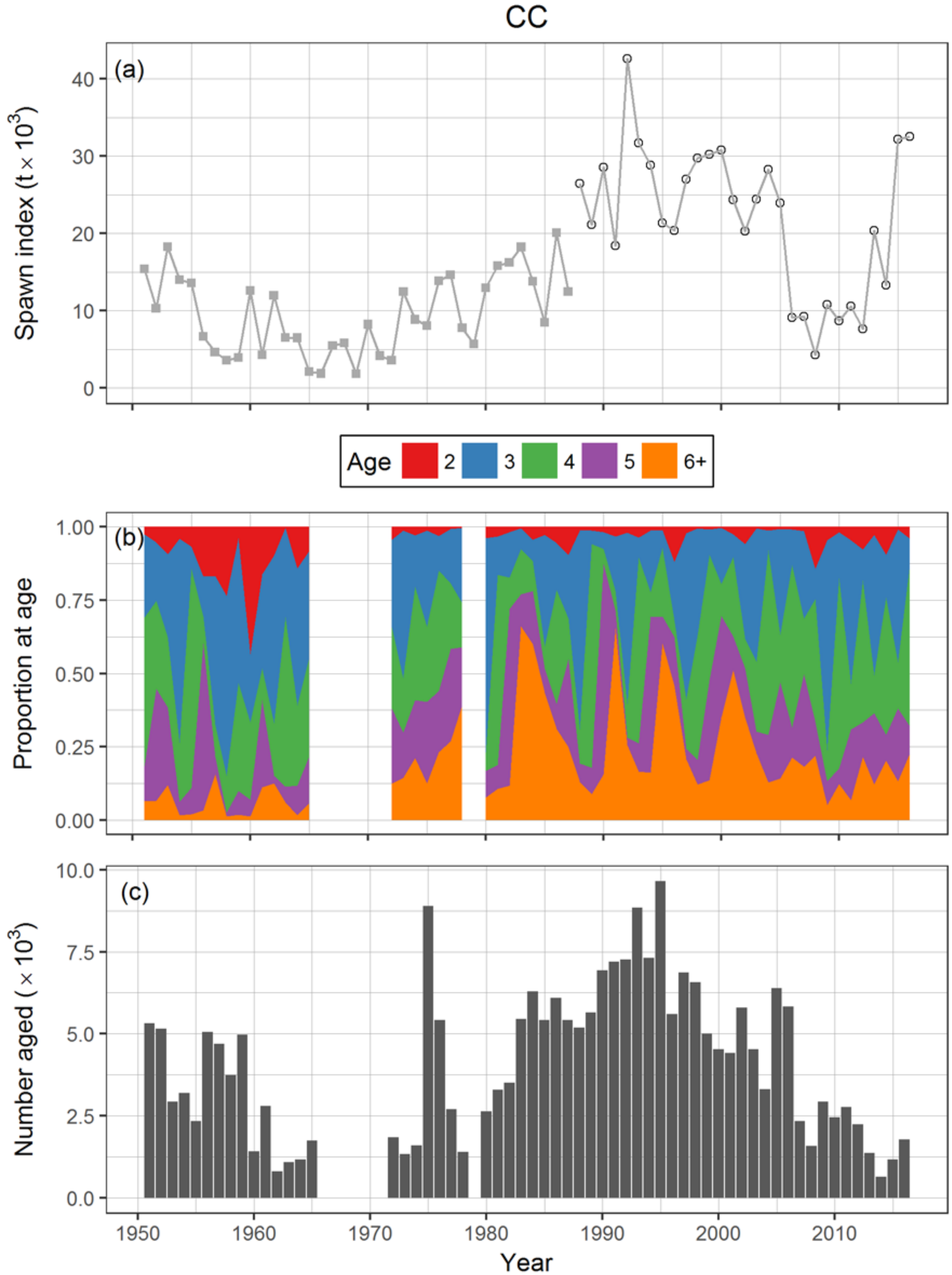

Figure 42. Spawn indices, proportions at age, and numbers of fish aged for CC Pacific Herring. See appendix text for description.

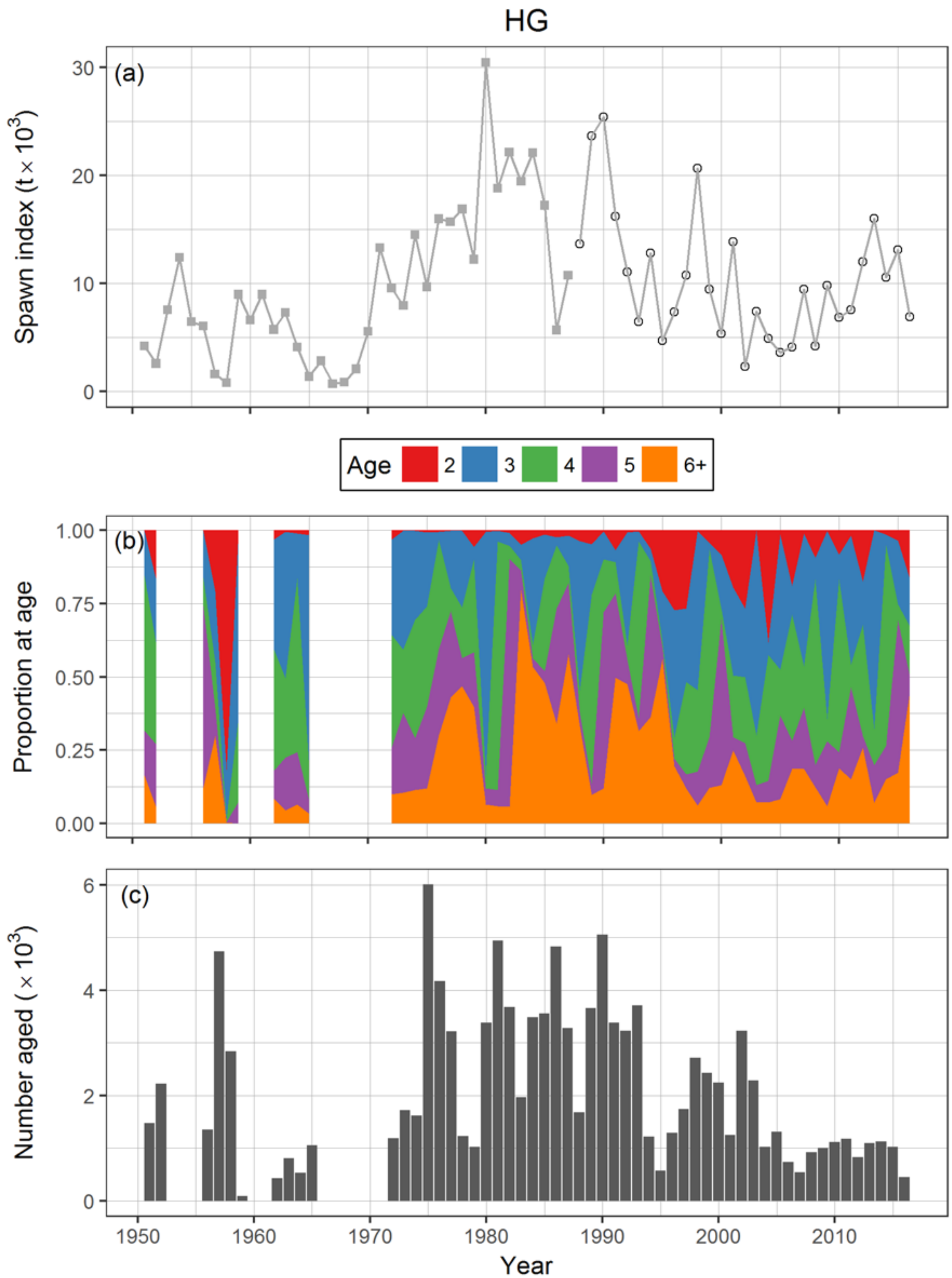

Figure 43. Spawn indices, proportions at age, and numbers of fish aged for HG Pacific Herring. See appendix text for description.

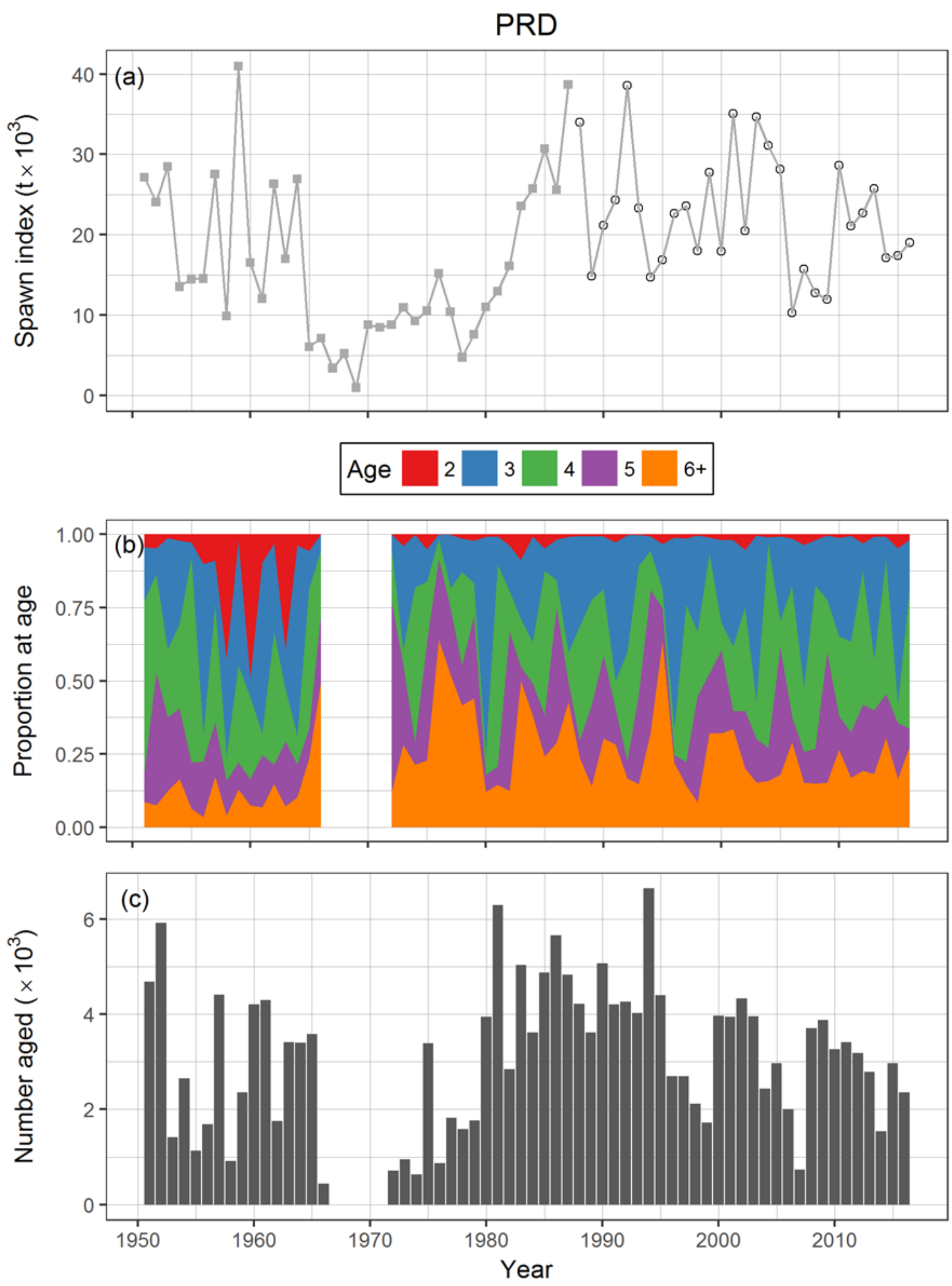

Figure 44. Spawn indices, proportions at age, and numbers of fish aged for PRD Pacific Herring. See appendix text for description.

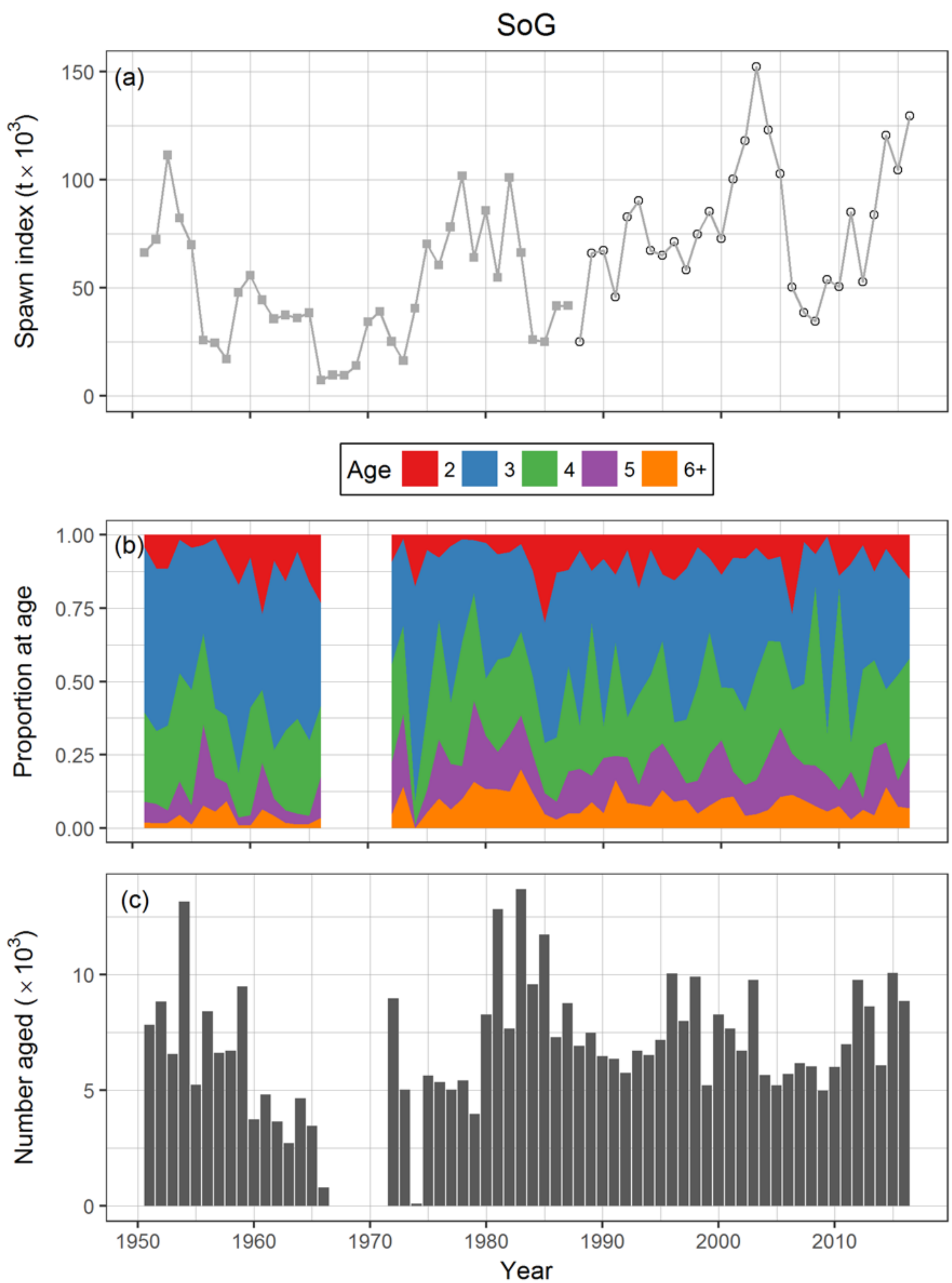

Figure 45. Spawn indices, proportions at age, and numbers of fish aged for SOG Pacific Herring. See appendix text for description.

APPENDIX E – PACIFIC HERRING RENEWAL

A multi-year process to revise the assessment and management system for BC Pacific Herring fisheries was initiated in 2015 (DFO 2015a). This process, Pacific Herring Renewal, was undertaken to address challenges to the existing system and to improve alignment with DFO policy (e.g., the DFO PA Framework). Pacific Herring Renewal consists of three components related to both science and fisheries management:

1. **Management Framework.** Includes identifying specific management objectives that incorporate reference points (e.g., LRPs) and evaluating the current management procedure against alternatives;
2. **Fisheries Management Reform**. Includes a review of licence fees, the pooling and licensing system, potential alternatives to on-grounds management and fishery monitoring; and
3. **Stock Assessment and Survey Program**. Includes a review of stock and fishery monitoring programs to determine where improvements and efficiencies might be realized.

In keeping with the DFO PA Framework, Pacific Herring Renewal will include the participation of resource users in development of a revised management system.

Practical considerations mean that there will be a tradeoff between allocating resources to maintaining the "operational stream" of activities needed to provide annual harvest advice and the development of a "strategic stream" needed to address components (1, 3). The current steps of annual data collection, stock assessment, and application of a harvest control can define a management procedure (MP) for Pacific Herring (de la Mare 1998, DFO 2016a). These steps form the basis of the operational stream that supports annual management decisions about harvests by First Nations and commercial fisheries in five major and two minor management areas (DFO 2015a). The operational stream is necessary not only to provide a mechanism for management decisions, but also to ensure that required stock and fishery monitoring data are collected. Specific steps may be varied to meet short-term tactical needs.

However, the operational stream relies on choosing a single "best" assessment model to produce harvest recommendations, rejecting all alternative hypotheses regarding stock and fishery dynamics (e.g., DFO 2016a). Currently for BC Pacific Herring, two cases of a statistical catch-at-age model are applied to each major stock. These cases differ only in an assumption about the value of the dive-based spawn survey catchability parameter (DFO 2016a). This assumption, which focuses only on parameter uncertainty, has frequently been the subject of debate since at least the late 1990s (e.g., DFO 1998, DFO 2015b). The scale of the estimated spawning biomass and estimates of key management parameters are substantially different between the two cases (see DFO 2016a; Martell et al. 2012). This is not a surprising outcome since even small differences in model assumptions, estimation methods, or the weights placed on different data sources can cause widely disparate estimates of spawning biomass, productivity and consequently BRPs (Cox et al. 2013).

The comparison of only two model cases has tended to focus attention on model choice criteria based on statistical fit and/or some subjective assessment of relative plausibility (e.g., DFO 2016a), rather than on strategic considerations related to sustainable use goals for resource users and hypotheses about structural dynamics related to time-varying processes or stock structure. The presentation of two alternative models also means that managers must either determine the allowable catch by integrating dissimilar measures of risk across the sensitivity cases, or choose a single model. Under the single "best" assessment model approach, a single set of BRPs can be postulated and all other similarly plausible models (and BRPs) must be rejected. However, the need to meet accepted scientific practice and increase compliance with element PA3 of the DFO PA Framework means that a potentially wide range of alternative hypotheses about stock and fishery dynamics must be developed as illustrated for BC Pacific Herring by Cox et al. (2015[1]). Each alternative view is expressed as a mathematical model that produces a corresponding set of BRPs such as LRPs. As new hypotheses are considered, different BRPs will result, perhaps defined similarly, but with different estimated values.

In part, the limitations posed by the single best assessment model approach and desire to strengthen policy compliance created the need for a “strategic stream” that can accommodate the complexity of BC Pacific Herring population dynamics and acknowledge the diverse and sometimes conflicting interests of resource users. The strategic component of Pacific Herring Renewal is intended to focus science activities on management-oriented outcomes by adopting a management strategy evaluation (MSE) approach (Smith 1994, de la Mare 1998, Kronlund et al. 2014b, Cox et al. 2015[1]). Under the management-oriented paradigm, effort is directed to designing management procedures that adequately meet strategic ecological, social-cultural, and economic objectives encompassed in policy and defined by resource users (Cox and Kronlund 2008; Cox et al. 2013; de la Mare 1998; Lane and Stephenson, 1995). Encompassing a range of plausible uncertainties for Pacific Herring means that a set of models is needed to represent the alternative hypotheses, with each model leading to a corresponding set of BRPs that can differ as a result of assumptions about system dynamics (e.g., changes in natural mortality or growth over time). These “operating models” are used to simulate data consistent with the assumed dynamics with observation errors typical of the stock and fishery. The simulated data, generated from a virtual population about which all processes are known, can then be used to evaluate the performance of candidate MPs relative to measurable objectives related to a given set of BRPs (see for example, Cox et al. 2015[1]).

The evaluation consists of “test driving” each proposed MP against simulated data produced by each operating model to gather performance measures related to the objectives. The performance measures are statistics that can be used to rank the management procedures in terms of a desired trade-off of management outcomes. Those management procedures that achieve satisfactory outcomes across a range of alternative views of uncertain stock and fishery dynamics represented by the operating models are said to be robust. In addition to increased robustness, a successful MSE can insulate the operational stream from piecemeal investigations of alternative views of stock and fishery dynamics, so that unnecessary disruptions to annual decision-making are minimized. A management procedure should only be changed when there is compelling evidence that uncertainty about the true underlying system dynamics means that an acceptable trade-off of management outcomes is unlikely under the current management procedure. This change in paradigm from the singe “best” model approach means that efforts are focused on comparing management outcomes instead of focusing efforts on a restricted investigation of parameter uncertainty (e.g., what survey catchability value is best) (Hilborn and Peterman 1996).

Regardless, for practical reasons there is a need to conduct the operational and strategic management streams for Pacific Herring concurrently while the MSE approach develops. Although there is a need to allocate finite resources between the operational and strategic streams, they do not work in opposition. The strategic stream extends the operational stream to address missing DFO PA Framework elements by formalizing stock and fishery objectives (element PA1), admitting a reasonable range of uncertainty (element PA3), and adding the required evaluation step to assess management performance (element PA4).

APPENDIX F – MANAGEMENT PARAMETERS FOR HERRING STOCKS

Review of the broad diversity of approaches to establishing both limit and target reference points for herring stocks worldwide illustrates the challenges posed by clupeids. We review examples from Alaska, US (Atlantic), and Atlantic Canada reporting reference points, including biological limits where available, for Atlantic and Pacific Herring stocks. We also review work conducted under the International Council for the Exploration of the Seas (ICES). In general, even when LRPs are reported for these stocks, the practice is to treat them as OCPs rather than reference levels to be avoided. This type of application is contrary to the intended use of LRPs described in the DFO PA Framework.

Alaska

Although Zheng et al. (1993) reported B_{MSY} estimates for Togiak and Prince William Sound herring stocks, biological limit reference points representing a point of irreversible or slowly-reversible harm have not been established for Pacific Herring stocks in Alaska. Most Pacific Herring stocks in Alaska are managed using a HCR that includes a minimum biomass threshold applied as an OCP, i.e., when the spawning stock biomass (SSB) is estimated to be below the minimum biomass threshold, commercial fishing is curtailed. The HCRs for a number of Alaskan stocks, including those that have historically had the largest biomasses, utilize a minimum biomass threshold level set at 25% of the average unfished biomass (B_0). Smaller stocks, for which there is generally less information, have thresholds set either at 25% of average (fished) biomass or a subjective level based on historical biomass. These choices include the expected biomass necessary to grow populations above levels observed in the 1970's and minimum levels needed to prosecute a commercial fishery. When the forecast spawning stock biomass (B) is below the minimum biomass threshold, the harvest rate for commercial fisheries is set to zero. HCRs for Pacific Herring stocks in Alaska are based on the threshold control rule implemented for BC Pacific Herring stocks and additional work specific to Alaska stocks (e.g., Carlile 1998a, 1998b, 2003; Funk and Rowell 1995; Zheng et al. 1993). For Togiak Pacific Herring, a simulation analysis using an age-structured model with a Ricker spawner-recruit relationship found the HCR combining the 20% harvest rate and a fishery threshold of 25% of the estimated B_0 rarely triggered fishery closures (Funk and Rowell 1995). For other South East Alaska stocks the harvest rate applied by the HCR varies between 10% (or 12%) and 20% of the forecast SSB when stocks are estimated to be above the threshold (Thynes et al. 2012):

1. The Prince William Sound HCR applies a variable sliding scale harvest rate between 0 and 20%;
2. The Kamishak Bay (lower Cook Inlet) includes a stepwise scale harvest rate between 0 and 15%; and
3. The eastern Bering Sea stocks use HCRs that include a 10%, 15%, or 20% maximum harvest rate, depending on the stock (i.e., with no sliding scale).

Atlantic US Stock

The Gulf of Maine/ Georges bank Atlantic Herring (*Clupea harengus*) complex is composed of several spawning aggregations, however methods to distinguish fish from each aggregation are not established and recent assessments have combined data from all areas into a single assessment of the entire complex (NFSC 2012a, 2012b). Current status for the aggregate stock is reported as relative overfished and overfishing thresholds using MSY-based reference points for the aggregate stock. A biological LRP representing a point of irreversible or slowly-reversible harm is not described. However, a policy-based limit relative to MSY is utilized. Current stock status is compared to a $B_{\mathrm{THRESHOLD}}$ level of $0.5B_{\mathrm{MSY}}$, such that the stock is considered overfished when $B < B_{\mathrm{THRESHOLD}}$ and fishing mortality rate is compared to a $F_{\mathrm{THRESHOLD}}$ level (usually F_{MSY}), such that overfishing is occurring when $F > F_{\mathrm{THRESHOLD}}$. In all circumstances, F targets are set as to avoid exceeding $F_{\mathrm{THRESHOLD}}$. In the management of US Atlantic Herring stocks, the biomass limit ($B_{\mathrm{THRESHOLD}}$) serves as an OCP and is not avoided with high probability.

Atlantic Canada Stocks

Three different approaches are used in defining LRPs for Canadian Atlantic Herring (*C. harengus*) stocks: a model-based approach is used for West coast of Newfoundland (NAFO Div 4R), an empirical approach based on a fixed time period is used for Southwest Nova Scotia/ Bay of Fundy (NAFO Div 4VWX), and low biomass values from a historical time period are used for Southern Gulf of St. Lawrence (NAFO Div 4T). In all three cases, biomass thresholds (B_{LIM}) operate as OCPs. For West coast Newfoundland, three reference points are defined for spring-spawning and autumn-spawning stocks: B_{LIM}, defined as 20% of the maximum observed historical spawning-stock biomass (a proxy for virgin stock size), B_{BUF}, (>B_{LIM}) the lowest observed historical spawning-stock biomass which produced good recruitment, and F_{BUF}, a target fishing mortality of $F_{0.1}$ (McQuinn et al. 1999). Following implementation of the DFO PA Framework, these references points are referred to as LRP (B_{LIM}) and USR (B_{BUF}), and assessment advice includes reporting of stock status relative to the LRP and USR (Gregoire and McQuinn 2010). The harvest strategy framework defined in Gregoire and McQuinn (2010) describes a hockey stick rule with exploitation rate ramping down between $F_{0.1}$, defined as a maximum limit exploitation rate for the ‘healthy zone’ (biomass > USR), and F_{med}, (spring-spawners) or F_{high} (autumn-spawners) defined as the maximum limit exploitation rate for the ‘critical zone’ (biomass > LRP). The reference points F_{med} and F_{high} are calculated from the slope of the line corresponding to the ratio between recruits and SSB where 50% (F_{med}) and 10% (F_{high}) of the stock-recruit observations are above the line.

The LRP for Southwest Nova Scotia/ Bay of Fundy herring is defined as the average survey value from the acoustic survey index from 2005 to 2010 for German Bank and Scots Bay. This empirical approach is chosen due to the absence of analytical models for these stocks. The years 2005-2010 reflect a period of stable survey acoustic estimates below which the risk of serious harm is deemed unacceptable (Clark et al. 2012). For comparisons of stock status relative to the LRP, status is calculated using a three-year running average of the acoustic estimates. The Scotia-Fundy Herring Integrated Fisheries Management Plan includes three conservation objectives, including: to continue to strive for fishing mortality at or below $F_{0.1}$ (DFO 2015c). However this objective is not included as a fishing mortality limit in the context of the DFO PA Framework, likely because a model-based approach is needed to compare realized fishing mortality rates relative to an F-target or F-limit.

For Southern Gulf of St. Lawrence herring, spring and fall components, LRPs are calculated as the average of the four lowest values of biomass during the late 1970s - a period of high recruitment - to SSB ratios. Interim USRs were defined at biomass levels to which stocks are expected to grow under average recruitment and removals reflective of the removal reference rate (DFO 2005). $F_{0.1}$ is the suggested removal reference above the USR: for the spring component $F_{0.1} = 0.35$ and for the fall component $F_{0.1} = 0.32$. There does not appear to be a removal reference rate associated with the DFO Cautious zone, as shown in the recent stock assessment (Figures 30-31 in DFO 2016b), however the implied harvest strategy is to reduce exploitation below the removal reference with the objective of increasing SSB (DFO 2005).

ICES: Norwegian spring-spawning herring and North Sea herring stocks

The Study Group on the Precautionary Approach (SGPA 97 and SGPA 98) define the ICES approach for describing stocks and fisheries within safe biological limits as:

… there should be a high probability that spawning stock biomass (SSB) is above a limit B_{lim} below which recruitment becomes impaired or the dynamics of the stock are unknown, and that fishing mortality is below a value F_{lim} that will drive the spawning stock to that biomass limit. The word 'impaired' is synonymous with the concept that on average recruitment becomes systematically reduced as biomass declines below a certain point. Because of uncertainty in the annual estimation of F and SSB, ICES defines the more conservative operational reference points, B_{pa} (higher than B_{lim}), and F_{pa} (lower than F_{lim}), where the subscript PA stands for precautionary approach. When a stock is estimated to be at B_{pa} there should be a high probability that it will be above B_{lim} and similarly if F is estimated to be at F_{pa} there should be a low probability that F is higher than F_{lim}. The reference values B_{lim} and F_{lim} are therefore estimated in order to arrive at B_{pa} and F_{pa}, the operational values that should have a high probability of ensuring that exploitation is sustainable based on the history of the fishery (ICES 2003).

Analyses for Norwegian spring-spawning herring (*C. harengus*) and North Sea herring stocks (*C. harengus*) use model-based equilibrium approaches to establish biological limits below which recruitment may become impaired, i.e., a point of serious harm, however these limits are not avoided with high probability and serve as OCPs.

For Norwegian spring-spawning (NSS) herring, a biological limit (B_{lim}) of 2.5 million tonnes is established as the minimum spawning stock biomass that would ensure adequate recruitment, based on available stock-recruitment information (Tjelmeland and Rottingen 2009). In the 1960s the NSS herring stock collapsed, following which the management objective was to rebuild the stock beyond the B_{lim} of 2.5 million tonnes. NSS fisheries are managed using a hockey-stick style HCR with OCPs set at B_{pa} and B_{lim}. When the stock falls below the B_{pa} of 5.0 million tonnes the fishing mortality rate is reduced from F=0.125 to F=0.05.

Precautionary limit reference points were established for the North Sea herring stock in 1998, including a biological limit (B_{lim}) of 800,000 tonnes that reflects a level below which recruitment may become impaired (ICES 2016). Main elements of the management plan include maintaining SSB above the B_{lim} of 800,000 tonnes, which is addressed through differential target fishing mortality rates triggered by OCPs. For example, when SSB > B_{trigger} of 1.5 million tonnes (an OCP) the target fishing mortality rate for adults and juveniles is F_{adult}=0.25 and F_{juvenile}=0.05. When SSB is estimated to fall between B_{trigger} and B_{lim} the target fishing mortality rate is reduced and set between F=0.25 and F=0.10 for adults, and is not to exceed F=0.05 for juveniles. If SSB < B_{lim} then fishing mortality is limited to no more than F=0.10 for adults and F=0.04 for juveniles (the Fish Site). With the addition of new data, the B_{lim} for North Sea herring was re-evaluated in 2007 and 2011, following which recommendations were to maintain a B_{lim} of 800,000 tonnes (ICES 2016).

APPENDIX G – CONSIDERATION OF LRPS IN SIMULATION STUDIES

Simulation studies that supported the design of the HCRs implemented in BC and Alaska were conducted at the same time as, or just after, implementation of the HCR using data and methods of the day (Haist et al. 1986; Haist et al. 1993; Hall et al. 1988). A similar approach to HCRs was introduced in Alaska supported by the work of Zheng et al. (1993). The results of Hall et al. (1988) suggested that if the harvest rate for the Strait of Georgia stock was less than U=0.3, the probability of biomass declining below a cutoff of $0.25B_0$ should be less than 5% over a 30-year period. Similarly Haist et al. (1986) concluded that a 20% harvest rate would keep average spawning stock biomass at a level well above the cutoff and result in a low percentage of years with spawning biomass below the cutoff. However, these studies were conducted following the rapid recovery of Pacific Herring stocks in BC from the historic lows in the late 1960s, and prior to the declining trend in weight-at-age and increasing trend in estimated natural mortality that began around 1990 (DFO 2016a).

Based on results from either model AM1 or model AM2 (DFO 2016a), the spawning biomass in the Central Coast (CC), Haida Gwaii (HG) and West Coast Vancouver Island (WCVI) management areas was below the estimate of $0.25B_0$ much more frequently than 5% of the time, and beginning in the mid-2000s persisted at or below that level for up to two to eight (AM1) or seven to ten (AM2) years depending on the management area (see Results section, DFO 2014). Corresponding closures of the commercial fisheries in the CC, HG, and WCVI management areas in themselves do not diagnose whether serious harm occurred, but did lead to loss of benefits to resource users over one to two Pacific Herring generations. Persistent estimated spawning stock biomasses below the cutoff are in part due to treating the cutoff values as OCPs rather than thresholds to be avoided. Uncertainty in estimating forecast spawning biomass and unfished biomass mean that the true forecast biomass after subtracting catch can have a non-negligible probability of being below the cutoff value. For example, see the probabilities of being below cutoff reported in decision tables contained in DFO (2014, 2015a, 2016a). Regardless, the goal of preserving an escapement of $0.25B_0$ in hopes of fostering a rapid increase from low spawning biomass levels has not been achieved.

Haist et al. (1993) conducted a comprehensive simulation study that included estimation of density-dependent mortality for Pacific Herring. They found that improved model fits to catch-age data were obtained by allowing natural mortality to increase at low stock abundance, i.e., by including depensatory effects consistent with the hypothesis that predators consume a higher proportion of a prey stock as prey biomass decreases. In addition, the study included a harvest simulation model that was used to estimate a minimum spawning stock biomass (MSSB) required to prevent stock collapse for a range of fixed harvest rates from 0.1 to 0.4. Stock collapse was arbitrarily defined as 2% of the estimated unfished average spawning stock biomass or less. The MSSB was set at the level where the probability of stock collapse is zero. Simulation results suggested that:

1. MSSB levels may be required at harvest rates as low as 0.1 (for the HG management area under density-dependent natural mortality scenarios);
2. The MSSB level increases as harvest rate increases;
3. Fisheries are closed on average from 0% (CC, SOG) to 8% (HG) of the time over 25 years at a harvest rate of 0.2 under density-dependent natural mortality scenarios;
4. Fisheries are closed on average from 4% (SOG) to 25% (WCVI) of the time over 25 years at a harvest rate of 0.3 under density-dependent natural mortality scenarios; and
5. The MSSB level as a percentage of average unfished biomass at a harvest rate of 0.2 under density-dependent natural mortality scenarios was 10% of unfished biomass for PRD and

WCVI, and 14% of unfished biomass for HG under density-dependent natural mortality scenarios. There was no need for a MSSB level for CC or SOG under a harvest rate of 0.2.

Pacific Herring stocks in BC were declared collapsed (Hourston 1980; Pearse 1982) in the late 1960s at estimated depletion levels higher than 2%. A collapse threshold of 2% of average unfished biomass, as used by Haist et al. (1993), is more consistent with the status of critically endangered stocks likely to face an extremely high risk of extinction (IUCN 2001) than stocks that are collapsed with the potential for recovery. Therefore, it is important to note that updating of the harvest simulation methodology used by Haist et al. (1993) with a definition of collapse aligned with historical experience for BC Pacific Herring would result in higher MSSB levels at a given harvest rate (see for example, Essington et al. (2015) who used a definition of 25% of the average population biomass as the threshold for collapse). In addition, as new operating models are introduced under the strategic stream described in Appendix E, more complicated ecological interactions may be considered so that the biomass levels used to define recruitment overfishing may not behave according to the compensatory stock-recruitment dynamics currently used in the BC Pacific Herring stock assessment model, i.e., Beverton-Holt stock-recruitment, where per-recruit production increases at low abundance (Punt 2006). Alternatively, stock-recruit dynamics may be depensatory, when per-recruit production decreases at low abundance. Nevertheless, the Haist et al. (1993) study used a sophisticated conceptual approach to estimating MSSB levels that, with modifications, would be appropriate for consideration for the strategic stream of Pacific Herring Renewal (DFO 2015a).

Recent studies of BC Pacific Herring management procedures show that the current HCR performs adequately only over a narrow range of simulated conditions that do not fully represent the plausible range of stock dynamics (e.g., Cleary et al. 2010, Cox et al. 2015[1], DFO 2015b). Cleary et al. (2010) conducted a feedback simulation exercise to compare the performance of a management procedure with a HCR that mimicked the Pacific Herring HCR and an alternative where the HCR specified control points at $0.4B_{\mathrm{MSY}}$ and $0.8B_{\mathrm{MSY}}$, i.e., the provisional HCR specified in the DFO PA Framework (DFO 2009a). Both HCRs specified a target harvest rate at U=0.2. For the purposes of their analysis they proposed a limit reference point at $0.5B_{\mathrm{MSY}}$ (to be exceeded with 95% probability) based on the suggestion of Shelton and Sinclair (2008) with respect to policy guidance in the New Zealand harvest strategy (Ministry of Fisheries 2011). The $0.5B_{\mathrm{MSY}}$ policy choice corresponds with a "soft limit" in the New Zealand Policy where a formal, time-constrained rebuilding plan is mandated. If the soft limit is breached the plan must ensure that the stock builds to the level of the soft limit with 70% probability to restore the biomass from a low state and restore the age composition from a possibly distorted state with relatively few larger, fecund fish. The *target* reference point adopted by Cleary et al. (2010) was B_{MSY}, to be exceeded with 50% probability over the simulation time horizons of 10, 20 and 30 years.

While both management procedures met objectives under high productivity scenarios over a 30 year time horizon, neither procedure met objectives under low productivity scenarios. This result was in keeping with results obtained by Schweigert et al. (1997) and suggested that the harvest rate of 0.2 may assist prevent long-term declines but might not allow rebuilding of severely depleted stocks. In other words, when the spawning biomass is increasing from low levels, particularly during low productivity periods, U=0.2 might better be regarded as a limit to be avoided than a target harvest rate.

Recently Cox et al. (2015[1]) considered candidate limit reference points for Pacific Herring in terms of their role as benchmarks against which to contrast the expected conservation performance of candidate management procedures. The performance of four different management procedures that varied only in the formulation of the harvest control rule was evaluated against candidate LRPs from each of the three categories of biomass-based limit reference points (Table 15).

Table 15. Summary of LRPs considered by Cox et al. (2015[1]). In all cases the fishing rate LRP is based on best practice (Sainsbury 2008) and the DFO PA Framework.

Case	Type	Biomass LRP	Source of Biomass LRP	Fishing Rate LRP
1	Equilibrium	$0.25B_0$	Interprets cutoff as a LRP	F_{MSY}
2	-	$0.3B_0$	Sainsbury (2008)	F_{MSY}
3	-	$0.4B_0$	Implied from Lenfest Rule (Pikitch et al. 2012)	F_{MSY}
4	-	$0.4B_{MSY}$	DFO PA Framework Policy	F_{MSY}
5	Dynamic	$0.25B^*_0$	Dynamic variant of (1) based on projection with 0 catch, i.e., 25% of $B_{unfished}$ (NSB_0 in Cox et al. 2015[1])	F_{MSY}
6	Historical (or empirical)	$\min(B_t)$, $t=1,...,T$	Minimum of estimated biomass	F_{MSY}

A limit fishing rate reference point of F_{MSY} was selected based on policy and best practice literature (e.g., Sainsbury 2008) and the DFO PA Framework policy. A feedback simulation evaluation was conducted that depended on a catch-at-age operating model parameterized by outputs from the 2014 Pacific Herring stock assessment (DFO 2014). Estimates of stock-recruitment parameters, natural mortality, growth and gear selectivity were input to the operating model so that unfished biomass and MSY-statistics could be calculated.

Based on the results of the simulation analyses, Cox et al. (2015[1]) concluded that research to identify LRPs for BC Pacific Herring fisheries should focus on fixed (equilibrium) objectives related to biomass despite the evidence for non-stationarity in observed weight-at-age and estimates of natural mortality rate. Examination of simulated outcomes against alternative LRPs for BC Pacific Herring stocks suggest that there may be little value in using LRPs that:

1. Track the dynamics of natural mortality and growth (Dynamic LRPs);
2. Utilize the lowest level of biomass from which the stock has recovered (Historical LRPs); or
3. Reference equilibrium-based F_{MSY} as a limit fishing mortality rate.

The results for dynamic ($0.25B^*_0$) and historical LRPs were criticized because they failed to indicate a breach of the LRP even in situations where risk could be significant, i.e., very low biomass levels. The key limitation of the dynamic LRP was the “ratcheting down” effect noted by Sainsbury (2008) where very low levels of abundance can occur at low productivity for a stock characterized by fluctuating productivity, as may be occurring for Pacific Herring. Essentially, the LRP is set to progressively lower levels as the stock declines, effectively lowering the conservation threshold. Cox et al. (2015[1]) did not evaluate Sainsbury’s best practice recommendation (b) to take the maximum of $0.3B^*_0$ or 0.2 of the median unfished biomass but instead used B_0 as a proxy for $B_{unfished}$ and considered multipliers of 0.25, 0.3, and

0.4. The Historical minimum biomass set the LRP to low levels such that the probability of a breach was near zero but at situations where there may be risks. The DFO PA Framework policy allows for consideration of historical estimates of biomass to be used as reference points, but suggests that the time for calculating minimum or average biomasses should be based on a fixed period of high productivity, rather than a moving period. Adopting a fixed window for historical reference points may avoid the results of Cox et al. (2015[1]), but there is no guarantee that a historical minimum is above levels associated with the possibility of serious harm.

In general the outcomes of the simulations reinforced the findings of Punt et al. (2008), who concluded that management performance can in some situations be relatively insensitive to the choice of reference points. For example, reference points based on F_{MSY} were ineffective in discriminating between management procedures because estimated values of F_{MSY} for each stock were relatively high and the annual fishing mortalities specified by the management procedures were much lower. Similarly, the progressive reduction of dynamic and historical LRPs in Cox et al. (2015[1]) meant that the management procedures failed to take conservation actions even at low levels of stock status that might pose a risk of serious harm. A LRP that is not effective in discriminating between management procedures is not useful. Simulation testing is the only way to *a priori* determine the utility of a given LRP in distinguishing management procedure choice. However, the simulation testing should be conducted in the context of a fulsome statement of limit *and* target-based objectives (which do not currently exist for BC Pacific Herring), since the components of the management system will interact in ways that cannot be predicted by evaluation of a single component (Kronlund et al. 2014b).

An alternative Historical minimum biomass LRP could be derived by fixing the time period over which the LRP is calculated to correspond to an acknowledged period of low abundance, or above where serious harm may be a possibility based on other considerations. This alternative may address the recommendation of Cox et al. (2015[1]) not to use worse-case scenarios for empirical LRPs, i.e., the minimum in the entire reconstructed biomass trajectory. For BC Pacific Herring this period could be during the late 1960s when the stocks collapsed or during the mid-2000s to mid-2010s when at least three of the five major stocks remained at low levels of abundance for an extended period. The latter period might be preferable given that the historical lows that occurred in the late 1960s were a result of overfishing and were not associated with the observed decline in weight-at-age of the last 2-3 decades. Recent assessments for Pacific Cod (Forrest et al. 2015) and Rock Sole (Holt et al. 2016) used LRPs based on the lowest estimated biomass from which the stock recovered to an above average level.

Other conclusions of Cox et al (2015[1]) were that LRPs may vary among Pacific Herring stocks because historically, the five major herring stocks in B.C. exhibit different biomass dynamics in response to fisheries, recruitment variability, trends in size-at-age, and the mean, variability, and trends in natural mortality rates. Estimated natural mortality differs substantially among Pacific Herring stocks in BC and had the strongest effect on projected stock trajectories (DFO 2016a), and hence performance of management procedures relative to LRPs.

APPENDIX H – FORAGE FISH CONSIDERATIONS

Forage fish experience dramatic fluctuations in population biomass (Pikitch et al. 2012) driven by stochastic environmental forcing (Chavez et al. 2003; Cushing 1988; Sydeman et al. 2013, see Shelton et al. 2014) and fishing (Essington et al. 2015). One challenge in defining serious harm to the forage fish population is accounting for possible deleterious effects propagated to dependent species, as noted in the DFO PA Framework (DFO 2009b). The implication is that in addition to providing biomass to support sustainable fishing on the forage fish stock, there must also be consideration given to dependent species when defining serious harm.

The need for a multi-species definition for serious harm to forage fish is defined in a broad suit of policies. Relevant international and domestic policy guidance on this topic includes:

1. *United Nations Fish Stocks Agreement (UNFSA 1995): Article 5 of the UNFSA explicitly states that nations should "adopt, where necessary, conservation and management measures for species belonging to the same ecosystem or associated with or dependent upon the target stocks, with a view to maintaining or restoring populations of such species above levels at which their reproduction may become seriously threatened".*
2. *NOAA Fisheries Ecosystem-based Fisheries Management (EBFM) Road Map (NOAA 2016) provides a national implementation strategy that includes six Guiding Principles considered actionable steps for implementing EBFM within NOAA Fisheries. Guiding Principle 5 "Incorporating ecosystem considerations into management advice" includes a requirement to Develop and monitor ecosystem-level reference points. The EBFM Road Map also acknowledges challenges for implementation: "An important challenge as we implement EBFM is to advance our understanding of processes as we discern the relative importance to fishery resources. NOAA Fisheries will work to better understand a broader suite of ecosystem processes, drivers, and threats, including [specific to forage fish]:*
3. *Trophic relationships (including predator-prey relationships and forage fish dynamics, and food demands of commercial and protected species); and*
4. *Sustainable Fisheries Framework- Policy of New Fisheries for Forage Species (PNFFS).*

In Canada, policy guidance is provided by the Sustainable Fisheries Framework (SFF) including: *Guidance for the Development of Rebuilding Plans under the Precautionary Approach Framework: Growing Stocks out of the Critical Zone and the Policy on New Fisheries for Forage Species.* The conservation-based PNFFS includes five objectives, two of which describe requirements of forage fish management that can be linked to LRPs, either through considerations for defining minimum biomass levels, e.g., Objective 2: "*maintenance of ecological relationships (e.g., predator-prey and competition) among species affected directly or indirectly by the fishery within the bounds of natural fluctuations in these relationships*", or through features linkable to definitions of serious harm, e.g., Objective 4: "*maintenance of full reproductive potential of the forage species, including genetic diversity and geographic population structure*" (DFO 2009b).

With respect to defining LRPs, the PNFFS specifically notes: "*The biomasses of forage species used as LRP in management should ensure both that future recruitment of the target species is not impaired and that food supply for predators is not depleted*." Finally, the PNFFS includes a number of management prerequisites, including: "*Consistent with the precautionary approach, there should be clearly identified conservation (limit) reference points and associated harvest control rules, for measurable properties of both the forage species (see 5.1.1) and some dependent marine predators* (see 5.1.2)". *The reference points should ensure that fisheries do not reduce the forage species population to levels where either its productivity, or the productivity of predators on it, would be reduced.*"

These international and domestic policies provide guidelines related to sustainable management of forage fish that describe requirements for ecosystem approaches for establishing biological limits. However, these policies are not prescriptive as to *how* to define limits or targets for forage fish.

Ecosystem-wide analyses of alternative harvest policies for the management of forage fish have been prominent in recent literature. Smith et al. (2011) used three different ecosystem models (Ecopath with EcoSim (EwE), OSMOSE, and Atlantis) in five areas to examine how changing harvest rates on all the modeled Low Trophic Level (LTL) species (defined as plankton eating species, including forage fish like Pacific Herring) affected the biomass of non LTL species (including other commercial species, marine mammals and seabirds). They conclude that considerable reductions in ecosystem impacts occur when LTL species are managed to a *target* biomass equivalent to 75% of the unexploited biomass and that this target could be achieved with exploitation rates of less than half MSY rates. Pikitch et al. (2012) used a combination of case studies and ecosystem modelling (Ecopath and EwE) to describe a set of robust, precautionary standards, management targets, and biomass thresholds to support the maintenance of forage fish populations within marine ecosystems. Their approach classifies forage fish into one of three "information tiers", and for each information tier they recommend management actions designed to ensure with high probability, that the "Dependent Predator Performance Criterion" is met. Pacific Herring meet the criteria of the "intermediate information tier", for which the recommended LRP, B_{lim}, is $0.4B_0$. Pikitch et al. (2012) recommend a hockey stick HCR with $B_{\mathrm{lim}} \geq 0.4B_0$ and F less than or equal to the lesser of $0.5M$ and $0.5F_{\mathrm{MSY}}$, and recommend a further increase in B_{lim} and decrease F when the ecosystem contains highly dependent predators or when precision of diet dependencies is low. This recommendation sets the lower biomass-based OCP equal to the LRP. However, see Cox et al. (2015[1]) for a simulation-based performance evaluation of this rule with respect to candidate reference points that resulted in a higher frequency of fishery closures due to the high harvest rate provided by $0.5F_{\mathrm{MSY}}$ relative to the current target harvest rate of 20% for BC Pacific Herring.

Essington et al. (2015) collated time series of biomass and fisheries catches for forage fish populations that account for nearly two-thirds of global catch of forage fish, including anchovies, capelin, herrings, mackerels, menhaden, sand eels, and sardines. They examined common and unique characteristics of population dynamics and population collapses. The authors describe forage fish population collapses as sharing these characteristics: high fishing pressure for several years before collapse, a sharp drop in natural population productivity, and a lagged response to reduced fishing pressure. They also found that lagged response to declines in natural productivity could sharply amplify the magnitude of naturally occurring population fluctuations. Using a subset of 15 populations for which it was possible to estimate fishing rate and natural population productivity, and using a "population collapse" criterion of 25% of average population biomass over the available time series, Essington et al. (2015) found average population productivity to have declined sharply beginning 2–3 y before collapse, rapidly falling to $-0.02\ y^{-1}$ (expressed as the fraction of average population biomass) immediately before collapse and rebounding shortly thereafter.

On the topic of biological reference points for forage fish fisheries, Essington et al. (2015) described the use of standard static reference points to judge stock status (e.g., unfished biomass) as having little meaning for forage fish stocks due to the lack of relationship between population production and population biomass, given large cyclical fluctuations in abundance. However, alternatives were not provided, as this was not the focal point of the research.

In an examination of fluctuations in food abundance (forage fish) on seabird breeding success across seven ecosystems, Cury et al. (2011) identified a threshold in prey abundance below which seabirds experience consistently reduced and more variable productivity. They recommend maintaining a minimum *target* forage fish biomass above one-third of the maximum observed long-term biomass, suggesting "one-third for the birds" as a guiding principle that could be applied widely to help manage forage fisheries for ecosystem resilience.

The role of Pacific Herring as a forage fish is well known, but the ecosystem service requirements of their predators are poorly understood and objectives for predator species are not generally specified. Guenette et al. (2014) provided a summary of the application of reference points and HCRs for forage fish in Canada and internationally, identifying Canada as part of the group of countries that do not account for the special situation of forage fish in terms of management. The PNFFS under the SFF requires a minimum level of information on the population dynamics and predation of forage species, however PNFFS only applies to new forage fisheries. Marine Stewardship Council (MSC) requirements for certification of LTL fisheries is for stocks to be maintained at a default target of 75% of the unexploited level, compared to the default 40% in non-LTL fisheries (Kaplan et al. 2012).

Tyrrell et al. (2011) showed that a wide variety of modelling approaches produce BRPs that are more conservative when predation mortality is explicitly incorporated in prey abundance calculations, and recommended inclusion of predation mortality in the estimation of reference points for single-species. Stock assessments for BC Pacific Herring include estimation of annual time varying natural mortality (DFO 2016a; Fu et al. 2004). While predation mortality is not directly measured for BC Pacific Herring stocks, this approach is an improvement over relying on fixed natural mortality assumptions such as M=0.2 y^{-1}. A similar approach is used for southern Gulf of St. Lawrence Atlantic cod (*Gadus morhua*) where M for the stock is modelled using independent random walks for estimates of M for the 2−4 and 5+ year age groups (Swain et al. 2012).

Until numerical relationships between predators and prey are more fully quantified, defining biological limits for forage fish in an ecosystem context will need to be based on meta-analyses and simulation studies as described in the references above. LRP recommendations for LTL species, e.g., $B_{\text{lim}} \geq 0.4B_0$ and F less than or equal to the lesser of 0.5M and 0.5F_{MSY} (Pikitch et al. 2012) could also be combined with best-practices for target and limit reference points described by Sainsbury (2008), i.e., "*In the absence of appropriate trophic models the best practice target reference point for biomass of nominated key prey species is no less than the mid-point between the unfished biomass and* B_{MSY}. However, recent work by Hilborn et al. (2017) showed that existing analyses of predator requirements that used trophic models have, in general, ignored important factors that include:

1. The high level of natural variability of forage fish;
2. The weak relationship between forage fish spawning stock size and recruitment;
3. The role of environmental productivity regimes;
4. The size distribution of forage fish, their predators and subsequent size selective predation, and
5. The changes in spatial distribution of the forage fish that affect the reproductive success of predators.

Hilborn et al. (2017) concluded that the impact on predators caused by fishing on forage species was generally less than indicated from trophic models, and that there is little evidence for strong connections between forage fish abundance and their predators for a range of US fisheries. They argued that any evaluation of harvest policies for forage fish needs to include these issues, and that models specific to individual species and ecosystems are needed to inform fisheries decision-making and policy.