

Marine Scotland Science Report



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The West of Scotland Marine Ecosystem: A Review of Scientific Knowledge

N Bailey, DM Bailey, LC Bellini, PG Fernandes, C Fox, S Heymans, S Holmes, J Howe, S Hughes, S Magill, F McIntyre, D McKee, MR Ryan, IP Smith, G Tyldsely, R Watret and WR Turrell

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Bailey N, Bailey D M⁵, Bellini L C⁵, Fernandes P G, Fox C¹, Heymans S¹, Holmes S², Howe J¹, Hughes S², Magill S¹, McIntyre F², McKee D³, Ryan M R⁵, Smith, I. P⁴, Tyldesley G², Watret R², Turrell W R²

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Marine Scotland is the directorate of the Scottish Government responsible for the integrated management of Scotland's seas. Marine Scotland Science (formerly Fisheries Research Services) provides expert scientific and technical advice on marine and fisheries issues. *Marine Scotland Science Reports* are a series of reports that publish results of research and monitoring carried out by Marine Scotland Science. These reports are not subject to formal external peer review.

Marine Scotland Science
Marine Laboratory
375 Victoria Road
Aberdeen
AB11 9DB

THE WEST OF SCOTLAND MARINE ECOSYSTEM: A REVIEW OF SCIENTIFIC KNOWLEDGE

¹Scottish Association for Marine Science, Scottish Marine Institute, Oban Argyll, PA37 1QA

²Marine Scotland Science, Marine Laboratory, 375 Victoria Road, Aberdeen, AB11 9DB

³Department of Physics, MASTS, University of Strathclyde, John Anderson Building, 107 Rottenrow, Glasgow, G4 0NG

⁴University Marine Biological Station, Millport, Isle of Cumbrae, KA28 0EG

⁵Institute of Biodiversity, Animal Health & Comparative Medicine, College of Medical, Veterinary & Life Sciences, Room 514, Graham Kerr Building, University of Glasgow, Glasgow, G12 8QQ

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1 Synthesis

The Physical and Biological Environment

Geography

1. The geomorphology of the area is complex and has been extensively influenced by its volcanic and glacial history leading to a complex coastline with raised beaches, fjords and numerous islands. Underwater there are examples of many features including areas of sandy, muddy-sand and muddy seabed, gas seeps and pock marks, exposed rock, reefs, shelves, pinnacles and seamounts. A wide range of water depths are found from the Malin and Hebrides Shelves, which are generally shallower than 250 m, to areas such as the Rockall Trough where depths can be in excess of 2500 m. In some areas there are rapid depth changes which combine with tidal flows to generate world renowned oceanographic features e.g. the Coryvreckan whirlpool. Although the overall bathymetry of the area has been charted, the seabed and habitats have not been surveyed to high resolution except over limited areas.

Oceanography

2. When coupled with ocean currents and tides the complex geomorphology of the region gives rise to a wide variety of oceanographic features including tidal jets, and gyres as well as partially isolated fjords where the waters are renewed periodically.

Oceanographic Trends - Offshore Waters

3. The currents within ICES Division VIa are strongly influenced by ocean circulation in the wider north-eastern Atlantic and in particular by the strength of the North Atlantic current (NAC). Observations in the Rockall Trough show that between the 1960s-1990s waters have tended to be cool and relatively fresh but from the mid-1990s there has been a period of warming and increasing salinity. However, this trend has recently shown signs of slowing.

Oceanographic Trends – Inshore waters

4. The general oceanographic patterns in the Little Minch and Sea of the Hebrides have been described from summer 1996 to autumn of 1998 by Gillibrand *et al.* (2003). Longer term observations are available from the Tiree passage mooring and some inshore stations. These show the mean water temperature to be around 10-11°C with clear seasonal fluctuations of around 3°C. There has been a sequence of alternate cooling and warming periods but with an overall trend for increasing temperatures at a rate of 0.57°C per decade up to 2004. However, since then, the rate of warming appears to have slowed as seen offshore.

Nutrients

5. At a regional scale, concentrations of nutrients tend to be highest in the winter due to the inflow of Atlantic water and nutrient concentrations generally decline through the spring due to biological production. Available evidence suggests that the amount of Atlantic water transported onto the shelf areas to the west of Scotland has a large influence on the availability of nutrients in the shelf and inshore waters in early spring. Terrestrial sources of nutrients are of less importance because the Scottish west coast is dominated by low-intensity agriculture and has rather low population density (except for the Clyde estuary). Long-term time-series of nutrient measurements are lacking for the west of Scotland and it is therefore difficult to rigorously test links between changes in atmospheric and oceanographic forcing and nutrient levels.

Chlorophyll and Primary Production

6. The general seasonal trends in chlorophyll standing stock for both the sea-lochs and coastal areas are known but analyses of longer-term trends suffer from a lack of data. The CPR phytoplankton colour index (PCI) is the only survey with long-term coverage (back to 1948). The PCI shows a shift in patterns of chlorophyll standing stock around the mid-1980s with the largest changes in the shelf waters. This change has previously been noted for adjacent sea areas such as the North Sea. Furthermore this change coincides with the recent warming of the surface waters which in turn appears to be linked with the amount of Atlantic water entering the area. As postulated for the North Sea, the increase in the colour index during summer may be related to changes in the composition of the post-spring bloom phytoplankton community with a shift toward increased abundance of flagellates during the summer.

7. Examples from other parts of the world have been found linking the amount of chlorophyll or primary production and haddock year-class strength. It might therefore be worth re-examining whether gadoid recruitment patterns for the west of Scotland can be linked with inter-annual changes in primary production. Although proposed mechanisms remain tentative, it is clear that potential links between environment, primary and secondary production and fish survival can only be explored when there are robust time-series available for both fish recruitment and environmental variables. Renewed effort in environmental monitoring is needed for the west of Scotland waters as environmental data coverage, both in time and spatially, is currently sparse.

Zooplankton

8. Copepods generally form the major component of the diet of larval gadoids. Other members of the zooplankton, mainly jellyfish and arrow-worms, can be important predators of fish eggs and larvae. Changes in the abundance and species composition of the zooplankton have therefore been suggested as key factors affecting the overall survival of young stages of gadoids in other areas.

9. The abundance of copepods in ICES Division VIa, as monitored by the CPR, has declined over the times-series (1948-2008). In the offshore waters changes in total abundance are mainly driven by shifts in the numbers of *Acartia* spp. whilst variability in *Para-Pseudocalanus* spp. tends to dominate in the shelf waters. In comparison to some historical periods, the abundance of *Para-Pseudocalanus* spp. seems to be particularly low at present. The abundance of *Calanus finmarchicus* has declined but not to the extent seen in the North Sea. In contrast, the abundance of *Calanus helgolandicus* in the shelf waters of ICES Division VIa has increased substantially since the mid-1990s. The strongest decline in total copepod abundance appears to have occurred later in the year with abundances in CPR samples in September now being between 10 to 100 times lower than in the 1950s. In the shelf waters this appears to be mainly due to a decline in *Para-Pseudocalanus* spp. Euphausiid abundance has also declined since the 1950s but in this case the most substantial declines occurred earlier in the year so that the peak abundance now occurs around August.

Time series for gelatinous zooplankton are almost completely lacking for waters to the west of Scotland. Circumstantial evidence suggests bloom events may be significant in the area but whether this affects gadoid year-class strength is not known.

Interactions between Zooplankton and West of Scotland Cod

10. Food availability for the early life stages of fish is generally thought to be an important factor affecting their year-class strength. The decline in *Calanus finmarchicus* and euphausiids in the North Sea is frequently cited as a reason for reduced recruitment of cod in that area. Although there have been changes in the abundance of *Calanus finmarchicus* in the shelf waters of ICES Division VIa, much larger declines in the abundance of *Para-Pseudocalanus* spp. have occurred. This may actually be of more importance for the early gadoid larvae to the west of Scotland as *Para-Pseudocalanus* copepods are known to be important prey for larval gadoids in other areas. However this evidence is circumstantial and based on limited CPR sample routes. Improved monitoring of the zooplankton in the gadoid spawning areas as well as research into the diets, feeding success, growth and condition of larval and juvenile gadoids in waters to the west of Scotland is needed.

Ocean Climate: Evidence from Long-term Monitoring

11. In-situ data suggests that coastal waters west of Scotland are currently (i.e. 1982-2008) warming at a rate twice that of North Sea coastal waters ($0.6^{\circ}\text{C decade}^{-1}$ compared to $0.3^{\circ}\text{C decade}^{-1}$).

12. North Sea warming may be greater in winter than in summer over this period, but there is no clear seasonal difference west of Scotland.

13. Large scale gridded data sets fail to show an east / west difference so clearly, hence the effect may be constrained to coastal waters.

Remote Sensing – What Can it Tell Us?

OCRS Products and Performance in Coastal Waters

14. Ocean Colour Remote Sensing directly provides information on the optical properties of the ocean. Optical data can be converted into estimates of a variety of useful products such as chlorophyll concentration which is a measure of the amount of plant material in surface waters, and sea surface temperature which is useful for identifying physical structures. OCRS products are generally more reliable for deep ocean waters. Great care has to be taken in interpreting OCRS products in shallow coastal regions where the presence of suspended sediments and coloured dissolved organic material from freshwater inputs can cause errors.

Review of Studies in Scottish West Coast Waters

15. Very little work has been published on the optical properties of Scottish west coast waters. The work that has been published has generally been restricted to sea-lochs and the Clyde Sea area. This means we have very little information on which to judge the quality of standard OCRS products for this region. It is likely that the quality of OCRS products will range from reasonably good quality in deep Atlantic waters to pretty poor in shallow turbid regions or close to major freshwater sources. Other published applications of OCRS data in Scottish waters include identification of potential Harmful Algal Blooms, the position of fronts between water masses and mapping of features on the seabed in very shallow waters.

Survey of Data Availability from Current Sensors

16. There are a number of satellite-borne ocean colour sensors currently in orbit, with new / replacement sensors scheduled for launch over the coming decade. The current continuous record of ocean colour time series data extends back to 1997. Data at various stages of processing can be obtained from space agencies (NASA and ESA) along with software for processing images. Online databases also provide access to O CRS data products that have been processed using standard settings. Much of this data can be freely accessed for academic research or non-profit activities.

Capabilities and Limitations of Ocean Colour Remote Sensing

17. O CRS is unique in remote sensing in that it provides information on biological processes as well as physical properties. Perhaps the key attraction of O CRS is the ability to map large areas of ocean rapidly with frequent repeat visits. This enables views of dynamic processes on scales that are simply unobtainable by any other means. The major limitations of the technology are: (a) inability to retrieve signal through cloud cover, (b) only surface waters are observed, (c) limited performance in coastal areas due to interference from light reflected from land and limited spatial resolution (typically ~250m max), and (d) variable data quality due to effects of suspended sediments and dissolved organic materials on standard processing routines. Potential applications include: water quality monitoring, monitoring algal blooms (including harmful blooms) and sediment plumes, and identifying water masses and fronts. This data is potentially useful for ecosystem modelling, managing areas of special interest (e.g. for aquaculture operations or protected areas) and monitoring ecosystem variability e.g. timing, magnitude and duration of the spring phytoplankton bloom. In the long term, the O CRS time series may provide unique insight into global and regional responses to changing climate patterns.

Preliminary Study of Available Data

18. A simple analysis of standard chlorophyll products retrieved from the Giovanni database reveals strong seasonal variability in monthly averaged chlorophyll concentration in the northeast Atlantic Ocean. In open shelf seas the spring bloom chlorophyll concentration peaks at ~ 2 - 3 mg m⁻³. There is some evidence of year to year variability in the timing of peak chlorophyll production with a late summer bloom in some years and not others. The Clyde Sea area presents significantly higher apparent chlorophyll concentrations, but these must be treated with caution as this is an area that is known to experience raised levels of suspended sediments and dissolved organic material in places. Overall, there does not appear to be any significant long term trend in the level of chlorophyll production during the course of the available time series. However variability in bloom timing / duration could have potential to influence productivity at higher levels in the food chain.

Conclusions and Recommendations

19. Ocean colour remote sensing enables monitoring of biological and physical processes across vast areas of ocean that cannot be matched by any other means. As such, OCRS has a crucial role to play in any future strategy for managing Scotland's extensive marine resources. Given the acknowledged limitations of the technology, OCRS does not provide a standalone solution for monitoring Scotland's seas. In order to exploit this technology to its full potential, there is an urgent need for a survey of Scottish coastal waters to identify optical characteristics that would enable assessment of data quality for standard OCRS products and development of regionally tuned variants. OCRS should form a central plank of a much broader strategy for long-term monitoring of Scottish territorial waters. Ideally this would involve the development of an integrated marine observatory that would merge data from OCRS and in situ platforms with a coupled physical / ecosystem model to provide a multilevel overview of current and changing environmental conditions.

Whitefish Stocks: Role of the Coastal Zone

The Role of the Nearshore Zone

20. In Scotland, juvenile Atlantic cod and whiting occupy the nearshore environment for most of their first year of life, and move into deeper water as age-1 fish during their first winter and second summer. Haddock are present in the nearshore only sporadically, and exhibit no consistent use of the nearshore at any point during ontogeny. Juvenile whiting and cod in the nearshore environment associate with complex habitat, including eelgrass, rocky substrate, macroalgae, and reef habitat. Loss of these habitats can negatively affect juvenile abundance of these species. The nearshore environment does not appear to exert any effect on haddock abundance or distribution.

21. Natural conditions or human impacts in the nearshore environment can have large effects on juvenile fish abundance. The resulting effects of year-classes strength can be traced through adult populations for up to six years following their settlement year.

22. In some countries, the abundance of juvenile cod and whiting in the nearshore is used as an indicator of year-class strength and recruitment into adult populations. Nearshore studies are not used for haddock monitoring. Nearshore surveys conducted at a limited number of sites with high temporal resolution are shown to provide reliable estimates of year-class strength for cod and whiting. Genetic sampling can reveal links between juvenile and adult populations.

Monitoring and Mitigation

23. Threats to juvenile cod and whiting abundance, such as habitat loss, vulnerability to capture as juveniles by *Nephrops* fishing trawls, pollution, salinity changes, and predator abundance can be monitored and, in some cases, mitigated through management schemes.

As the loss of suitable nearshore habitat appears to have negative effects on cod and whiting juvenile abundance the potential benefits of habitat restoration or artificial reef implementation should be explored further.

Possible Future Coastal Surveys

24. A programme of coastal surveys could be initiated to cover the three main coastal cod populations. The areas to be surveyed can be chosen to take in a representative sample of whiting nursery areas. The phases of such a project would be:

- Identification of nearshore areas and critical habitats for juvenile whitefish populations using a wide-scale survey to determine areas where juvenile gadids are abundant and which habitats support the greatest density of young fish. The most appropriate methods for this are described in Davies *et al* (2001) but are likely to include beach seining and SCUBA surveys.
- Further work on the identification of links between nearshore juveniles and adult populations using fin clips and population genetics.
- Ongoing annual sampling at a subset of sites representing each population. Wherever possible the survey sites chosen should support SNH's work to survey mSACs and/or SEPA's responsibilities under the Water Framework Directive.

West of Scotland Demersal Fisheries

25. The seas to the west of Scotland are considered largely as ICES area VIa and although the sea does not have a specific name it encompasses many smaller seas such as the Hebrides, the Firth of Lorn, the Sound of Jura and the Firth of Clyde. In comparison to the North Sea, the fishing grounds, especially for demersal fisheries, to the west of Scotland are of a much smaller extent due to a rougher terrain and deeper waters.

26. This section reviews earlier assessments of the state of the demersal fisheries West of Scotland. A separate report will present the most up to date assessment of the state of west coast fish stocks.

Survey Nomenclature

27. The research vessel survey for demersal fish west of Scotland has had several names applied to it in a confusing manner. This section also proposes a standardisation of survey name.

The Q1 and Q4 Scottish VIa IBTS

28. The section goes on to review results of the two large IBTS surveys of fish in area VIa, the assessment methodology used in fishery independent assessments and the validity of the data.

Application of a Survey-Based Assessment Model (SURBA)

29. The SURBA model is currently a survey based assessment which has a long history in catch-based fisheries stock assessment and was first applied to European research vessel survey data by Cook (1997). The simple basis of the model has been expanded and adapted over the years and has been recently used experimentally in ICES assessment working groups such as in the assessment of Northern Shelf Demersal Stocks (WGNSSDS now incorporated into the WG Celtic Seas - WGCS) for west of Scotland ICES Division VIa.

30. For this report, SURBA 3.0 was used to analyse the survey data from the Quarter 1 survey ScoGFSQ1 for cod, haddock and whiting in the west of Scotland. Attempts to fit the model to ScoGFSQ4 survey proved unreliable and this may be a result of the relatively short time series for this survey. It was found that SURBA could not fit an optimum model for the Q4 survey, particularly in the case of whiting. The survey data for ScoGFSQ1, however, is available from 1985-2009 for all three species, cod haddock and whiting.

31. The results of the analysis show that in general the surveys support the stock trends estimated from the ICES assessments for cod and haddock with some disagreement between the two analyses for whiting. The SURBA results for cod in Division VIa supports the perceived long term decline in the stock with some stabilization in recent years. Trends in recruitment, SSB and Z show the most similar patterns between the working group assessment and SURBA outputs for haddock and this may be attributed to the fact that commercial catch data is excluded from the assessment for haddock from 1994 onwards.

32. The whiting comparisons with the SURBA run show more conflicting results, which could be explained by the fact that the TSA model run in 2004 was found to behave in a manner difficult to explain as the information from the catch and survey data was quite divergent. Considering the outputs for whiting, SSB appears to have reached an all time low and recruitment also appears to have fallen in the more recent years.

33. The SURBA method appears to provide consistent estimates of relative trends in population parameters from research vessel survey indices. Clearly it is possible to get detailed information from survey based assessments and use them for stock assessments when catch data is thought to be unreliable.

West of Scotland Industry Survey – 2010

West Coast Surveys

34. Marine Scotland Science conducts two surveys each year of the fish in the shelf seas west of Scotland; the Quarter 1 Scottish Vla International Bottom Trawl Survey (Q1 Sco.Vla.IBTS) and the Quarter 4 Scottish Vla International Bottom Trawl Survey (Q4 Sco.Vla.IBTS). In Quarter 1 of 2010, an additional survey of the gadoid fish species was conducted as part of the West Coast Industry Science Forum; the “industry science survey”. The planning and execution of the industry science survey involved the explicit participation of the fishing industry, using two chartered trawlers to conduct the survey.

Aims of this Study

35. The aims of the analysis of the industry science survey results presented in this report is to 1) produce an absolute estimate of abundance for the principal gadoid species (rather than the index of relative abundance which is normally produced from the two IBTS surveys), and to 2) compare the estimates at age from the industry science survey with the indices at age produced from the Q1 Sco.Vla.IBTS. Abundance estimates were calculated on the basis of certain assumptions about the catchability of the net.

Abundance Estimates

36. Minimum and best estimates of the total stock and spawning stock biomass of cod were lower than those produced by the ICES assessments, whilst biomass estimates for haddock were more in line with assessment values.

	TSB (tonnes)		SSB (tonnes)	
	Industry Science Survey Best Estimate ($\pm 95\%$ CL)	2010 ICES Assessment	Industry Science Survey Best Estimate ($\pm 95\%$ CL)	2010 ICES Assessment
Cod	2833 (0-7348)	10533 (8419-12647)	2139 (0-5532)	6227 (4385-8069)
Haddock	22471 (11950-32992)	19018 (8756-29280)	19461 (12704-26218)	13377 (9065-17689)
Whiting	10944 (8004-13884)	Unknown	4704 (3698-5710)	Unknown

Estimates at Age

37. The 'best' estimates at age for all species are compared with the indices produced by the IBTS. These were very similar in the case of haddock and whiting, but less comparable in the case of cod.

Uncertainty

38. Estimates of uncertainty in the industry science survey are also provided. The estimates of cod were very uncertain with measures of relative standard error in excess of 75%. There was less uncertainty in the estimates of haddock (RSE~20%) and whiting (RSE~12%).

Outcome of the 2010 Survey

39. The results from the survey and the principles of its design have since been used to redesign the Scottish west coast IBTS surveys.

Modelling the West Coast Ecosystem – Ecopath with Ecosim

40. This investigation set out to use a particular approach to broaden our understanding of cod, haddock and whiting population dynamics on the West Coast of Scotland. The first goal was to reproduce the biomass and catch time-series from the stock assessments. Then we looked at how the model responded to various hindcast and forecast scenarios which were designed to explore hypotheses for poor gadoid performance on the West Coast. In order to do this we have parameterised an Ecopath with Ecosim model, building on previous work on an EwE model for the West Coast (Hagan and Pitcher 2005). The main difference between this investigation and the previous one was the use of time series data, again derived from surveys and stock assessments, to drive and fit the Ecosim model.

41. The default setup of an Ecopath model is for each functional group to be in equilibrium with respect to production and mortality. In Ecosim, changes in biomass for a particular functional group result from changes in mortality or prey biomass. Using this default we

were unable to reproduce some of the key features of the biomass trends, notably the initial steep decline in mature haddock biomass. In order to reproduce this decline in Ecosim it was necessary to assume that the mortality was initially greater than the production for the haddock functional group, or in the terms of Ecopath, that there was a negative Biomass Accumulation. Making this assumption for cod and whiting produces different fits without clear improvements. It should be noted that further consideration could be applied to all functional groups within the model in this regard but this is left for future work.

42. As well as some improvements in the fit to data being achieved through the use of biomass accumulation terms in the initial model setup, noticeable gains were achieved through the use of an automatic fitting routine within Ecosim which adjusts vulnerability parameters. These parameters govern how changes in functional groups' biomasses affect the interactions between them with the two extremes being described as "top down" and "bottom up" (Christensen *et al.*, 2005). The automatic adjustments to the parameters depend on how the data differs from model output and in turn on choices made about how the Ecopath model is set up (such as the inclusion of biomass accumulation terms or not).

43. In order to draw conclusions from the fitted values it is therefore necessary to have a degree of certainty, obtained from both well founded choices in the initial model set up and indication that these choices above all others lead to an improvement in model fit. While some improvement in model fit was observed it is felt that a more exhaustive exploration of justifiable set-ups and a sensitivity analysis on the model outcome is necessary before direct conclusions can be drawn from the fitted vulnerability values. As is noted in the scenario testing section the values chosen for these parameters can play a large roll in any hindcasting or forecasting.

44. The other avenue that was explored to produce a better fit to the data was the use of a driving function for primary production. Again, an automatic fitting routine within the EwE software was used to find a time series which results in the best fit to data for all functional groups. The result was a quantitative improvement in the fit but the qualitative improvements were small. A cursory comparison was made between the fitted forcing function and ocean colour index time series as well as North Atlantic Oscillation time series, but no correlation was observed. More work could be done in this area. It was unclear whether the automatic fitting procedure was just fitting to noise in the fitting dataset as opposed to more general trends observed. Perhaps an investigation into how using smoothed sets of fitting data affects the fitted forcing function could provide further insight.

45. The resulting fits to the biomass and catch data for the mature stanzas of cod, haddock and whiting show some merit. The broad trends are picked out but the amplitude in the variation, particularly for haddock and whiting, is not as high as observed in the stock assessments. Looking at the juvenile stanzas for haddock and whiting suggests a reason for this as the model seems to be unable to reproduce the annual variation in recruitment. An investigation into forcing recruitment was carried out with good results but this was not

carried forward for the scenario tests as it is not known how recruitment would have behaved in the light of the different conditions of the scenarios.

Controlling Grey Seal Population

46. The investigation into controlling the grey seal population suggests that while grey seals provide a considerable source of mortality to cod the rise in their population over recent decades has not been the reason for the decline in the cod stock. The rise in grey seal biomass from the fitted EwE model was at the lower end of that estimated by the assessment (SCOS, 2008). Recent further analysis (Holmes and Fryer, 2011) has examined assessments in which cod mortality is specifically included. This had the effect of scaling up the biomass of cod present from early years forward but does not appear to change the conclusion that the relative effects of seals and other sources of mortality have not changed. These analyses have taken the latest SMRU estimates of seal abundance which suggest a levelling off in abundance since the mid 1990s. Forcing the grey seal biomass to increase at higher trajectory might have resulted in more of the decline in the cod biomass being explained by the rise in the grey seal population. This was not investigated in the current work. Owing to not all of the ground being available for trawling and also uncertainty about the extent to which meta-populations of cod occur in this area, the interpretation of the relative importance of different sources of mortality is confounded.

Impact of the Nephrops Fleet Bycatch

47. The investigation into the impact of the *Nephrops* trawl fleet on the three gadoid stocks shows that there is insufficient bycatch by the fleet to have a large impact on the stocks. Moving to a clean *Nephrops* trawl results in very little change to the haddock and whiting biomass trajectories whilst a small increase is observed in the cod biomass trajectories.

Moving to F_{MSY}

48. The final scenario investigated involved making a forecast of biomass and catch trajectories on the basis that the stocks are fished at maximum sustainable yield (F_{MSY}). The construction of this scenario involves making a number of assumptions. In the stock assessments F_{MSY} values have been estimated for assessed stocks and here an assumption is made that these values of F_{MSY} can be achieved in the context of a mixed fishery. For the other functional groups we assumed that F remained constant at the most recent value. This is unlikely to be the case but more work is necessary to estimate the impact that changing to F_{MSY} for some stocks would have on effort and in turn on the fishing mortality on other stocks.

49. The forecasts are carried out on two model variants and for the haddock forecast in particular there are wildly different outcomes. The way the model responds depends on the fitted vulnerability parameters. Given the more plausible model set up we see cod, haddock and whiting biomasses increasing. Cod biomass eventually returns to historically high

values and the catch rises but does not achieve the previously observed levels. Haddock biomass returns more quickly to historically high values and the catch shows a limited increase. For whiting the catch shows very little recovery despite an increase in biomass.

Future Work

50. In order to test the scenarios, it was necessary to have a fitted Ecosim model. Fitting the model and implementing the scenarios have been the main goal. In doing so the construction and balancing of the Ecopath model had to be done fairly quickly. Re-visiting this part of the process would be informative and perhaps more time should be spent exploring the inevitable inconsistencies in the original data set and the variety of ways in which these can be resolved. An exploration of how the resulting variety in Ecopath models goes on to perform in Ecosim simulations would be interesting and could give a better idea about how reliable the Ecosim predictions are.

51. If multispecies models are to become a greater source of advice for fisheries management then it seems sensible to invest in making it easier for people constructing the models to quickly access and compile the data necessary. Fishbase (Froese, 2000) is a great source of biological data although further data mining is still necessary. Fishery specific data is held in more disparate databases. For assessed species this data is compiled for the assessment but for others the various discard and landings databases need to be consulted. Furthermore, for non-assessed species derivation of survey indices needs to be carried out. If this data was held centrally and openly accessible, with relevant indices calculated this job could be quicker and calculations would not have to be repeated.

2 The Physical and Biological Environment

C Fox and J Howe

2.1 Introduction

This review is focused on ICES Division VIa which covers the inshore and offshore waters to the west of Scotland out to longitude 12°W (Figure 2.1). The southern boundary lies along the parallel 54.5°N until it joins the Irish coast. Between Northern Ireland and Scotland the boundary lies in the North Channel at 55°N. The eastern boundary of Division VIa lies along the 4°W meridian apart from a small extension in the north-east corner (Figure 2.1).

ICES Division VIa lies within Region III for OSPAR purposes. Overviews of the status of the area have been included in the OSPAR (OSPAR, 2000) Quality Status Report. ICES Division VIa was also covered in the UK Charting Progress Report where it fell within areas 6 (West Scotland including the Minch) and 7 (Scottish Continental Shelf). Since then responsibility for managing Scotland's marine environment has been almost totally transferred to the Scottish Government. In response, the Scottish Government published an assessment of the state of Scotland's Seas in 2008 (Baxter *et al.*, 2008). Recently a marine atlas of Scotland's Seas, including the western waters, has been produced (Baxter *et al.*, 2011). All these reports contain useful overviews of the region to varying levels of detail.

The waters to the west of Scotland are also considered in the UK Department of Trade and Industry Strategic Environment Assessment (SEA) where they are known as SEA7 (although the SEA7 area stretches further west than the boundary of ICES Division VIa). The SEA assessment was completed in 2007 and this report and technical annexes also contain much useful background information (http://www.offshore-sea.org.uk/consultations/SEA_7/).

2.2 Geography

The geomorphology of ICES Division VIa is complex and has been extensively influenced by its volcanic and glacial history (Figure 2.2). Water depths range from the Malin and Hebrides Shelves which are generally shallower than 250 m to regions such as the Rockall Trough where depths can be in excess of 2500 m. The seabed generally consists of extensive areas of sand and muddy-sand interspersed with rocky outcrops.

North Channel – Clyde

Considering firstly the shelf area and working from south to north, Division VIa is linked to the Irish Sea via the North Channel. This channel is strongly sculpted with rock pinnacles and a series of depressions, the largest of these being Beaufort's Dyke (max depth around 290 m). Strong tidal currents have prevented the deposition of fine sediments in the North Channel. Just to the north and east the Firth of Clyde forms a large area of coastal water with lower tidal currents. This has allowed the accumulation of substantial mud deposits

coming mainly from the rivers draining into the Firth. To the north-west, beyond Islay, there is an extensive shelf (the Malin Shelf) with variable morphology.

Malin Shelf

To the west and north-west of Islay there is a wide platform with numerous ledges descending to around 140 m near the shelf edge. On the Malin Shelf itself there are raised areas of resistant granites, known as the Stanton Banks, which rise up to around 60 m below the sea surface. They are deeply fissured with the inter-connecting gullies filled with rippled coarse shell-sand. Elsewhere on the Malin Shelf sediments are predominantly sand with patches of sandy-gravel and gravelly sand. There are significant areas of rock pavement to the west of South Harris and South Uist which are kept clear of sediment by strong tidal action. Cold-water corals (*Lophelia pertusa*) are also found in this area, for example the Mingulay Reefs to the south-west of the Outer Hebrides.

Sea of the Hebrides

Moving to the north-east of the islands of Tiree and Coll the seabed takes the form of a broad, asymmetric channel descending to a maximum of 240 m. The southern part of the Sea of the Hebrides is dominated by fine-grained sediments in the form of muddy sand except in the shallows where gravelly sand and diamicton deposits occur. There is a 320 m deep depression called the Muck Deep about 20 km south-south-west of Rhum which contains mud deposits whilst a number of other deeps occur between Muck and Ardnamurchan.

Little Minch

The stretch of water between Skye and the Outer Hebrides is known as the Little Minch and the seabed deposits become progressively coarser moving northwards. The seabed here is highly irregular with bank summits as shallow as 40 m but deeps descending to 180 m. The Minch itself has a more regular seabed with a greater dominance of coarse sediments. Moving westwards from the Outer Hebrides there is a large area of exposed rock with sand and coarse sediments further offshore (Pantin, 1991).

Beyond the edge of the Malin and Hebrides Shelves the seabed has only been mapped at a lower spatial resolution, except where detailed surveys have been conducted in relation to oil exploration or areas of special conservation or scientific interest.

Rockall Trough

The Rockall Trough is a deep water basin, separating the Malin and Hebrides Shelves to the east from the Rockall Plateau to the west. At its northern limit the Trough is separated from the Faroe-Shetland Channel by the Wyville-Thomson Ridge. To the southwest the Trough widens and deepens, descending to 4000 m, before merging into the Porcupine Abyssal

Plain. Generally the floor of the Rockall Trough is smooth and relatively featureless with sediment drifts mantling the western margins whilst slope fans locally encroach, especially along the eastern margin. The Rockall Trough itself is largely covered with fine deposits but these have been re-worked around the Anthon Dohrn seamount and Rosemary Banks by the water flows around these obstacles to form a series of moats (Roberts *et al.*, 1974). The Anthon Dohrn Seamount and Rosemary Bank, as well as the Hebrides Terrace Seamount, are all relict volcanoes. Outside of ICES Division VIa, the north-western Rockall Trough is bounded by a series of relatively shallow banks e.g. George Bligh Bank and these elevated regions effectively separate the north-western trough from the deep-water regions of the Hatton-Rockall and Iceland Basins further to the west.

2.3 Oceanography

When coupled with ocean currents and tides the complex geomorphology of the region gives rise to a wide variety of oceanographic features including tidal jets, density-driven recirculation gyres and Taylor columns as well as partially isolated fjordic systems with periodic overturning (Inall and Sherwin, 2006).

Offshore

Starting offshore, flows within ICES Division VIa are strongly influenced by circulation patterns within the wider north-eastern Atlantic (Figure 2.3) and in particular by the strength of the North Atlantic current (NAC). The NAC forms an important component of the north-eastern Atlantic arm of the global thermohaline circulation and is partially responsible (the other important factor being the prevailing wind patterns and a localised ocean radiator effect caused by deep mixing in the winter) for the more temperate climate experienced in Scotland when compared with other land-masses at similar latitudes (Rahmstorf, 2003).

At one time it was thought increasing global temperatures could lead to sufficiently rapid melting of sub-Arctic ice as to cause a shut-down of the north-eastern Atlantic THC (Vellinga and Wood, 2008). Such a dramatic change in ocean circulation would have significant impacts on European terrestrial and marine ecosystems leading to a period of cooling although, in the longer-term, this might be offset by the overall warming trend (Vellinga and Wood, 2008; Kuhlbrodt *et al.*, 2009). However, the flow-rates of the different currents forming the north-eastern Atlantic THC show considerable variability and available observational time-series are probably too short to reach a consensus on whether there has already been a slow-down (Cunningham *et al.*, 2007). It has been widely recognised that longer term monitoring of the THC is needed and the NERC program RAPID-WATCH (2007-2014) was designed as the UK contribution towards this goal.

The oceanography of the Rockall Trough has been reviewed by Ellett *et al.* (1986) and, more recently by Inall and Sherwin (2006). Deep-water circulation in the Rockall Trough involves a number of water masses (Figure 2.4). The deep waters in the trough largely originate from Labrador Seawater (Lavender *et al.*, 2000) but Eastern North Atlantic Water and North

Atlantic Current Water also flow in from the south. The eastern margins of the trough are influenced by an along slope current which flows northwards until it is deflected by the Wyville-Thomson Ridge (Huthnance, 1986). At the northern end of the Rockall Trough, cold, dense, Norwegian Sea Arctic Intermediate and Norwegian Sea Deep Water from the Faroe-Shetland Channel intermittently overflows the Wyville-Thomson partially mixing with overlying warmer North Atlantic Water in the process (Sherwin and Turrell, 2005).

Evidence from seabed scouring indicates that flows of 50 cm s^{-1} are not uncommon in the valleys between the banks in the north-western Rockall Trough but bedforms north of Bill Bailey's Bank indicate that currents can locally reach 100 cm s^{-1} . Even faster flows occur in the Faroe Bank Channel (Kuijpers *et al.*, 1998). The intermediate to surface waters (down to around 700 m) generally flow towards the north-east but include several gyres, for example around the Anton Dohrn Seamount, which in turn spin-off eddies into the Rockall Trough (Mauchline, 1986b). The influence of this can be seen both in meandering drogue-buoy tracks (Ellett *et al.*, 1986) and in satellite imagery (Huthnance, 1986).

The links between the offshore flows described above and the currents over the shelf have recently been re-examined by Inall *et al.* (2009). As shown in Figures 2.3 and 2.4 there are surface and deep northward flowing currents along the eastern boundary of the Rockall Trough. At the northern end of the Trough the surface currents continue to flow northwards but the deep current is steered eastward by the Wyville-Thomson Ridge eventually mixing with overspill water originating from the Faroe-Shetland Channel. There is considerable inter-annual variability in the quantities of North Atlantic Current water and Eastern North Atlantic Current water entering the Rockall Trough and this variability has been linked to the patterns of wind stress in the wider North Atlantic.

Slope Current

The flow of water along the eastern edge of the Rockall Trough is also known as the European Slope Current (ESC) and this is a persistent feature along a track of around 1600 km from Goban Spur (~49°N 011°W, south of the Porcupine Seabight) to north of the Shetland Islands. Based on the Shelf Edge Study (SES) conducted adjacent to the Malin Shelf (~56-57°N), residual flow (that is the flow remaining after removing the tidal signal) of around 20 cm s⁻¹ was observed in the ESC down to 500 m. In addition, flows onto and off the adjacent shelf were low in summer but more significant in winter (Souza *et al.*, 2001). The Hebridean Terrace Seamount does not rise high enough to significantly affect the ESC which flows over it.

Outer Shelf

Over the outer parts of the Malin and Hebridean Shelves and where depths exceed 100 m the water is mainly of Atlantic origin (salinity > 35.0). Tongues of higher salinity water typically penetrate onto the shelf e.g. between Tiree and Barra (Figure 2.5). Further south there is a persistent front (the Islay Front) which runs from Malin Head to off the south-western tip of Mull. In winter the front is generated by salinity differences but in summer there is also a thermal component (Hill and Simpson, 1989). Atlantic water may also be periodically transferred onto the Shetland Shelf possibly as a result of instabilities in the slope current north of the Wyville Thomson Ridge.

Coastal Current

Closer inshore the Scottish Coastal Current (SCC) flows northwards carrying a mix of Irish and Clyde sea water from the North Channel. As it flows north the SCC mixes with fresh water run-off from the Scottish Highlands. Water in the SCC takes around 4-6 months to move from the North Channel to Cape Wrath (Inall *et al.*, 2009).

Based on drifters a density driven recirculation has been observed in the south Minch during early summer (Hill *et al.*, 1997). The widespread nature of these density driven summer features has recently led to a re-assessment of shelf-sea circulation patterns around the UK which suggests the existence of an almost continuous seasonal flow around the western edge of Ireland and Scotland (Figure 2.6).

Impact of the ocean currents on Temperature and Salinity

The main impacts of the inter-annual and inter-seasonal variability in ocean currents are on water temperature and salinity and on the input of nutrients from the Atlantic into the shelf and inshore waters. Long-term reconstructions of temperature are available but for ICES Division VIa these are based on relatively sparse observations. The patterns in temperature and salinity are described later in this section and in Section 3. Apart from short-term studies, such as the Shelf Edge Study (Souza *et al.*, 2001), and a year-long study reported

in Gillibrand *et al.* (2003), the only long term oceanographic observations in ICES Division VIa are the Ellett Line and the Tiree Passage Mooring (maintained by SAMS) and a number of inshore temperature recording stations.

Recent work at SAMS is exploring the use of an automated sea glider to collect data along the Ellett Line (<http://dalriada.sams.ac.uk/glider/>). Such autonomous instruments can operate almost continuously and can collect considerably more oceanographic data at much lower cost than can be achieved with ship-based studies. One limitation is that the number of parameters which can be recorded using autonomous instruments remains more limited than for ship-based observations. The glider also requires relatively deep-water for safe operation so will not replace the need to monitor shelf oceanography using ships and moorings.

Inshore - Little Minch / Sea of Hebrides

Moving onto the shelves, the Little Minch and Sea of the Hebrides was studied from summer 1996 to autumn of 1998 by Gillibrand *et al.* (2003). Temperatures were at a minimum during the spring survey and were highest during the autumn. Salinity data showed strong, though variable, intrusions of high salinity Atlantic water at depth into the Sea of the Hebrides all year round. These intrusions reached at least to 57° N. Lower salinity Irish Sea water ($S = \sim 34.4\text{--}34.8$) was also present in the shallower, eastern areas during all the surveys. At the eastern extremities of the study areas, fresher ($S < \sim 34.4$), coastal water was observed at the surface.

North Minch

Gillibrand *et al.* (2003) also reported that the North Minch appeared to experience weaker and more transient intrusions of shelf water and exhibited a relatively strong east to west gradient in salinity. This gradient was caused by the higher levels of freshwater discharge from the mainland compared with the Outer Hebridean islands. Lying between the Sea of the Hebrides to the south and the Minch proper to the north, the Little Minch, appeared to be a zone of strong vertical mixing. Stratification was relatively weak compared to that in the basins to the north and south.

The influence of the North Atlantic Oscillation

As indicated above, the strength of flows onto the shelf regions is influenced by the wider atmospheric patterns of the north-eastern Atlantic. The North Atlantic Oscillation (NAO) index is frequently used to summarise these larger-scale atmospheric conditions although the strongest links between the NAO and parameters such as sea surface temperature are observed for the winter and early spring periods. Even in winter the NAO only accounts for around 30% of variability in sea level pressure so it certainly does not act as a 'perfect index' as is sometimes implied.

This interplay of influences can be traced right into the western Scottish sea-lochs. For example, Gillibrand *et al.* (2005) showed that the different conditions of wind forcing, precipitation and river discharge in high and low NAO index years would have a significant influence on salinity in Loch Sunart. Strong persistent westerlies, typical of high index (positive) NAO periods, were seen to inhibit deep water renewal in the upper basin during the winter months, resulting in a steady winter decline in salinity due mixing with freshwater run-off. The salinity in the main basin was more stable and was dominated by stronger exchange with the coastal waters. This particular study was based on a hydrographic model and the paper emphasized the need to develop much longer and more comprehensive observational time-series to fully test its findings.

The Sea Lochs

Many of the sea lochs along the Scottish west coast are fjordic with shallow sills which restrict exchange with the shelf waters. The Firth of Clyde has a similar shallow sill and during the summer, when the waters in the Firth are thermally stratified, ingress of North Channel water is limited. During winter the system is salinity stratified and inflow is sufficiently dense to sink to the bottom thereby renewing the deep water (Jones *et al.*, 1995; Rippeth *et al.*, 1995).

2.4 Oceanographic Trends - Inshore Waters

As mentioned previously there are a number of temperature record datasets for the inshore waters of the Scottish west coast. These include recording sites at Loch Ewe, Mallaig, Millport and Loch Maddy. The Millport time series is the longest and goes back to 1953. The daily readings were originally taken using bucket and thermometer but in April 2003 a buoy with a minilogger was installed next to the Keppel pier. The other datasets only go back to 1999 and use electronic miniloggers. Miniloggers are generally set to record every 30 minutes and data are available for download as daily averages from www.marlab.ac.uk. All these temperature time-series show clear seasonal signals and for the recent overlapping period there is good agreement between signals from the different sites. The Millport data show evidence of a warming trend since the 1980s (Figure 2.8).

2.5 Oceanographic Trends – offshore waters

Rockall Trough

Between the 1960s-1990s surface, mid and deep waters across the north-east Atlantic have tended to be cool and relatively fresh but from the mid-1990s there was a period of warming and increasing salinity. However, this trend has recently shown signs of slowing. These patterns are also apparent in the Rockall Trough data from the Ellett line (Ellett *et al.*, 1986; Holliday *et al.*, 2008). The initial pattern of cooling and freshening was explicable in terms of changes in ice melt, river discharge and precipitation. The subsequent increase in temperature and salinity probably reflects a natural multi-decadal oscillation but a

component of the temperature increase may also be attributable to global warming, thought to be a result of anthropogenic induced climate change during the late 20th and early 21st centuries (Figure 2.9). At present the deep waters of the Rockall Trough remain cool and relatively fresh being driven by the pattern of deep-water convection in the Labrador Sea (ICES Working Group on Oceanic Hydrography, 2009).

Tiree Passage

Further inshore, the Tiree passage oceanographic mooring is located on a bank in 45 m of water at the narrowest part of the channel between the Islands of Coll and Tiree (Figure 2.5). Since June 1981 hourly current measurements have been recorded using an instrument moored 11 m above the seabed and from Nov 1987 an additional current meter, 22m above the seabed, was added. Since September 1997, salinity recording was added at both depths. Due to funding problems observations ceased in September 1997 but the mooring was re-deployed from June 1999 to February 2000 and re-deployed more permanently in May 2002 recording to the present. As well as gaps due to funding problems there are data gaps caused by operational failure but such problems affect the majority of oceanographic moorings.

The mean water temperature at the mooring was 10.1°C with clear seasonal fluctuations of around 3.2°C (Figure 2.10). Less obvious tidal signals can also be detected (Inall *et al.*, 2009). The temperature timeseries show a sequence of alternate cooling and warming periods but with an overall trend for increasing temperatures at a rate of 0.57°C per decade up to 2004 (a linear trend was shown in Inall *et al.* (2009) but should not be extrapolated to more recent observations as the rate of warming appears to have slowed).

As mentioned above this pattern has also been detected throughout the north-eastern Atlantic (ICES Working Group on Oceanic Hydrography, 2009). There is a strong correlation between the Tiree passage temperature data and the time-series from Port Erin in the Irish Sea and a slightly weaker, but still significant, correlation with temperatures in the Rockall Trough. The relative strength of the correlations appears to correspond with estimates of the relative contributions of Irish Sea coastal and Atlantic water to that flowing through the Tiree passage (roughly 3:1 ratio of North Channel to Atlantic water). The timing within each year when the minimum water temperatures occur does not appear to have changed over the time series but there has been an advance in the timing of maximum temperatures of around 12 days per decade.

Reliable salinity data are only available from 2002 for the Tiree mooring and so were not considered by Inall *et al.* (2009). Analysis of the current meter data indicates predominantly northward flows with a clear seasonality in current strength. There is also a significant correlation ($r = 0.59$) between winter residual current speeds at the mooring site and the NAO. This is most likely a direct result of increased south-westerly wind stress in years when the NAO index is positive.

Gridded Datasets

There are a number of gridded datasets for sea temperature which cover ICES Division VIa at a variety of spatial and temporal resolutions (see also section 3). The underlying data come from direct and satellite observations. These data sources have been blended and a variety of cut-offs and thresholds used to remove erroneous data points. In some cases this can lead to the exclusion of 'extreme' values which may in fact be genuine. Datasets are also either interpolated or un-interpolated. In the interpolated datasets areas lacking observations have been estimated using various smoothing techniques. Due to differences in the trimming and interpolation routines used there can be differences between each of these data series although they are all correlated (being often based on the same, or at least an over-lapping subset of underlying data). The number of underlying observations decreases rapidly moving back in time so there will be more uncertainty on earlier values (Figure 2.11).

For this study we examined monthly data corresponding as closely as possible to the ICES Division IVa boundaries from NOAA OIv2 (1-degree interpolated), ERSSTv3 (2-degree interpolated) and Hadley ISST (1-degree interpolated) datasets (note the term 1 or 2-degree refers to the spatial resolution and not to temperature range or temperature resolution). Monthly data were extracted back to 1900, or as far back as available, and examined for temporal and spatial patterns.

The patterns between the annual mean temperatures in the gridded datasets are in general agreement apart from the period 1932-1938 when the ERSST suggests slightly warmer temperatures than the NOAA OI dataset (Figure 2.12). The smoothed trend suggests there was a period of cooling between 1900 to 1910 followed by relatively stable period until around 1920 after which there was quite strong warming up to around 1934. This warm period persisted until 1960 after which there was another period of cooling with temperatures declining to close to the minima observed in the early 1900s. Since the mid 1980s sea surface temperatures have again been warming although this increase may now be showing signs of slowing.

The overall trends are also apparent in each quarter (where Q1 = Jan-Mar inclusive, Q2 = Apr = Jun inclusive etc.). The recent warming trend is most pronounced in Q2 and Q4 but appears to have slowed most rapidly in the winter months (Figure 2.13).

2.6 Availability of Oceanographic Models

Computer models covering ICES Division VIa have to cope with a variety of technical problems including complex geomorphology which necessitates the use of nested or variable geometry grids, steep bathymetric gradients which can lead to problems with standard hydrodynamic equations and mixing across the shelf edge which is rather poorly understood. Although quite detailed circulation models have been developed for the west of

Scotland these have tended to focus on small areas of specific interest e.g. circulation in some of the sea lochs in relation to the dispersal of sea lice (Murray and Gillibrand, 2006).

At a wider scale, Gillibrand *et al.* (2003) reported results from an implementation of the Princeton Ocean Model which was developed as part of a PhD project but results from this have not been published further and the model is not run in an operational mode. Waters out to the shelf edge are covered by the POLCOMS model run by NOC but further offshore areas which lie within ICES Division VIa are not covered. At present there are plans to develop a regional scale model for the west of Scotland at SAMS although current resources do not allow it to be run in an operational mode.

In conclusion, the main patterns of ocean circulation to the west of Scotland had been described by the 1980s in papers such as Ellett (1979) and Ellett and Edwards (1983) although many of the features described remained speculative at that time. More recent work has largely confirmed the patterns proposed in these earlier papers but added further detail about mechanisms operating in certain areas e.g. density driven recirculation patterns (Hill *et al.*, 1997).

Nearly all of the papers and reports cited above have emphasised the lack of regular long-term oceanographic observational series (excepting the Ellett line and Tiree mooring) to the west of Scotland, particularly in comparison with data availability for adjacent areas such as the Irish and North Seas. Thus although the general oceanographic patterns are well characterised there is a much weaker understanding of variability, particularly over inter-annual and longer timescales. According to the ICES hydrographic database, less than 1% of their CTD record holdings come from ICES Division VIa and there are virtually no records prior to 1980. There is also a lack of regional-scale operational oceanographic modelling although smaller-scale models have been developed.

Summary

Available data suggest episodes of warming and cooling in the waters to the west of Scotland during the last century. Between 1900 to 1910 waters cooled followed by a relatively stable period until around 1920 after which there was quite strong warming up to around 1934. This warm period persisted until 1960 after which there was another period of cooling with temperatures declining to close to the minima observed in the early 1900s. Since the mid 1980s sea surface temperatures have again been warming although this increase may now be showing signs of slowing. Since these patterns are also apparent in salinity data they appear to be mainly related to the amount of Atlantic water flowing onto the shelf seas in Division VIa. However, localised warming, potentially associated with the global anthropogenic signal may also be occurring.

Several studies have pointed out that cod recruitment appears to be linked with the NAO, especially when stocks are depleted (Brander and Mohn, 2004; Brander, 2005; Stige *et al.*, 2006). The mechanisms underlying this link are poorly understood but temperature may be

a significant factor with stronger year-classes of cod being associated with cooler temperatures (Planque and Fox, 1998; Planque and Frédou, 1999; O'Brien *et al.*, 2000). An examination of how this correlation varies with the time of year has indicated that cod recruitment success may be most strongly linked with temperatures during the winter and early spring when cod eggs and larvae are produced..

These conclusions remain tentative as such analyses are complicated by the auto-correlated nature of both environmental and recruitment time-series. Statistical analyses are also difficult to perform with relatively short time-series which exhibit these characteristics. Nevertheless it may be worthwhile re-examining these relationships for the west of Scotland given the temperature changes described above. For ICES Division VIa comparing mean temperature data for the different periods of the year shows generally similar trends i.e. cool summers and autumns tend to be associated with cool winters and springs (Figure 2.13). Thus although Planque and Fox (1998) and Planque and Frédou (1999) showed that correlations between cod recruitment and temperature were generally strongest for Quarter 1, implying an effect of the egg, larval and/or early juvenile stages, environmental interactions at later stages might also be important.

As far as we are aware such direct relationships of year-class strength with environmental signals have not been established for other gadoids such as haddock and whiting.

2.7 Nutrients

Offshore

Offshore nutrients have been measured on some of the Ellett line cruises since 2001 but temporal coverage is patchy and the data are not analysed further here.

Shelf Waters

Outside of the sea-lochs, the general seasonal patterns in nutrients over the shelf waters have been described by Gillibrand *et al.*, (2003) based on a series of cruises conducted between July 1996 and September 1998. Dissolved surface nitrate, phosphate and silicate concentrations were highest during the winter although silicate levels throughout the water column were slightly lower to the south-east of the Outer Hebrides. During the spring survey, nutrient concentrations were much lower adjacent to the Scottish coast reflecting increased primary production in the coastal waters. By the summer, nutrient concentrations were reduced throughout most of the area except at depth where a tongue of elevated nutrient water was detected in the south-west corner of the Sea of the Hebrides and in patches in the Minch and off Rhum.

By the autumn, surface nutrient concentrations in the Little Minch were 2-8 times higher than those in either the North Minch or the Sea of the Hebrides and reflected surface temperature gradients. These nutrient distributions suggest either a much lower rate of primary

productivity or an enhanced nutrient supply in the Little Minch compared to the two adjacent basins. At depth, high nutrient levels were again observed in a tongue from the south-west associated with the intrusion of Atlantic water into the Sea of the Hebrides. Levels of surface dissolved ammonia were generally low throughout the region and throughout the seasonal cycle. Deepwater ammonia concentrations were generally around $0.1 \mu\text{mol l}^{-1}$ except in the summer in the North Minch where the intrusion of Atlantic water caused increased ammonia concentrations to about $3.5 \mu\text{mol l}^{-1}$.

The nutrient distributions observed during the autumn cruise suggested that the strong vertical mixing in the Little Minch was lifting deeper water nutrients to the surface from where they were being transported northward into the nutrient-depleted surface waters of the North Minch. It was noted that this process could have potential implications for the productivity of the Minch basins and deserves further investigation. The current data and modelling revealed a strong, persistent, southerly flow at depth through the Little Minch channel. Whilst previous studies had hinted at such a feature, these results demonstrated that the southerly flow was persistent over periods of several months (Figure 2.7).

Sea Lochs

Nutrients have been measured in various sea-lochs on the west of Scotland covering the spring bloom and in some cases, with less frequent sampling, other seasons e.g. (Solórzano and Grantham, 1975; Solórzano and Ehrlich, 1977a; Solórzano and Ehrlich, 1977b; Solórzano and Ehrlich, 1979). In terms of fuller coverage of the complete annual cycle, Fisheries Research Services (now Marine Scotland Science) and collaborators undertook monthly nutrient sampling in Loch Linnhe and the upper Firth of Lorne over the whole of 1991.

Dissolved nitrate levels at all sites were highest in winter at around $6\text{--}9 \mu\text{mol l}^{-1}$ declining in the summer to around $1 \mu\text{mol l}^{-1}$. Following the decline in dissolved nitrate caused by the spring-bloom, internally recycled ammonium became an important source of nitrogen for phytoplankton growth (Rees *et al.*, 1995; Watts *et al.*, 1998). More recently SAMS have measured summer nutrient levels in the Firth of Lorne as part of a program on harmful algal blooms. Seasonal data are available for three years and show the typical pattern of high winter levels with a decline in the late spring to summer (Fehling *et al.*, 2006). Since 2003, monitoring of nutrient and chlorophyll levels during the summer months has continued, but the data are not yet published.

Research has also been conducted on the degree to which aquaculture in the inshore areas and sea-lochs might elevate nutrient levels e.g. Laurent *et al.*, (2006). The general conclusions from these studies are that nutrient inputs from aquaculture are low relative to those due to the influx of oceanic water into the shelf areas and that any impacts would be localised to the sea-lochs where the farms are located. It is not expected that aquaculture activities would have any measurable impact on nutrient levels across the wider region (Heath *et al.*, 2005).

Clyde

This general pattern of low anthropogenic impact on nutrients does not however apply to the Clyde Estuary. Its bathymetry restricts exchange with the waters of the North Channel and it is adjacent to several major population centres, including Glasgow. Because of these factors the Clyde estuary is considered to be nutrient enriched (Simpson and Rippeth, 1993; Tett *et al.*, 2003). Rippeth and Jones (1997) and Kennington and Johns (2007) noted that an earlier post-bloom (September/October) recharge of dissolved nitrate occurs in the surface waters of the Clyde estuary compared with offshore waters.

A strong inverse relationship had previously been demonstrated between salinity and nitrate concentration in the Clyde suggesting that the main source of nitrates accumulating in the surface waters in the autumn was from terrestrial run-off. This was supported by the fact that river-flow in the Clyde estuary catchment increases in September/October and does not decline until the following February. In the deeper waters of the Clyde Estuary, Jones *et al.*, (1995) and Rippeth and Jones (1997) showed a steady increase in dissolved nitrate during the year reaching a peak around October. Since the origin of the deep-water is inflow of higher salinity North Channel water over the sill and this inflow is limited during the summer months due to thermal stratification, the accumulation of deep-water nitrate during the year appears to be due to regeneration.

With regard to dissolved inorganic phosphate and silicate, winter concentrations in the Clyde are generally higher than for the offshore areas, again resulting from terrestrial run-off. In winter-time the exchange of water between the North Channel and the outer Firth increases due to the breakdown of thermal stratification which allows stronger density driven flows (inwards at depth over the sill and outwards near the surface). The Clyde estuary may therefore act as a source of nutrients to the adjacent coastal waters during the winter.

Summary

In summary, at a regional scale, concentrations of nutrients tend to be highest in the winter due to the inflow of Atlantic water and nutrient concentrations generally decline through the spring due to biological production (Colebrook, 1986). In the sea-lochs, terrestrial sources of silicate may act to prolong the spring-diatom bloom. A secondary increase in surface nutrients may occur in the autumn as a result of the increased mixing of the water column. Longer term trends in nutrients are likely to be correlated with large-scale forcing such as the atmospheric and oceanic circulation patterns. Available evidence suggests that the amount of Atlantic water transported onto the shelf areas to the west of Scotland has a large influence on the availability of nutrients in the shelf and inshore waters in early spring. Terrestrial sources of nutrients are of less importance because the Scottish west coast is dominated by low-intensity agriculture and has rather low population density (except for the Clyde estuary). An exception is the supply of silicate to the sea-lochs where terrestrial sources may act to prolong the spring diatom bloom.

However, long-term time-series of nutrient measurements are lacking for the offshore waters of the west of Scotland and, even for inshore waters, most available time-series are relatively short or fragmented. It is therefore difficult to rigorously test links between atmospheric forcing, oceanographic transport, nutrient levels and primary production. Except for the Clyde Estuary, anthropogenic input of nutrients appears to be a relatively minor problem in the waters of ICES Division VIa. Nutrient enrichment resulting from feeds used in the marine aquaculture industry is likely to be very localised.

2.8 Chlorophyll and Primary Production

2.8.1 Available Methods for Measuring Primary Production

Primary Productivity

Primary productivity is strictly only measured using incubation experiments (Marra, 2009). Because such measurements require dedicated research ships with facilities to allow experiments using radio-labelled compounds, estimates of the amount of photosynthetic pigments in seawater are often used as a proxy for productivity. However, it must be remembered that pigment concentration is a measure of the standing stock of phytoplankton. One can estimate production by applying biomass to production conversions but if loss rates are high, due to grazing, sedimentation or other factors, actual production will be higher than estimated.

Chlorophyll Concentrations

Measurements of chlorophyll concentration at a particular location have also usually only been taken weekly or monthly. While this is adequate to show general seasonal patterns, recent high-frequency observations from moorings in the North Sea, where chlorophyll concentration is measured every few hours, have shown that concentrations can peak and decline extremely rapidly e.g. over a period of 2-3 days. Even with sampling rates as frequent as weekly, such events are likely to be missed.

There are several methods for estimating chlorophyll concentrations in seawater. The standard method is based on spectrophotometry (measuring light absorption) and is rapid and relatively easy to perform. An acidification step is normally included to remove that part of the signal due to phaeopigments.

An alternative approach is fluorometry where the water sample is stimulated with light at a certain wavelength and the amount of re-emitted light at a longer wavelength is measured. Different pigments will respond and re-transmit at specific wavelengths. Because it avoids wet-chemistry, fluorometers have been developed which can be deployed *in situ*, either on moorings or on towed bodies. Calibration of fluorometers to allow conversion of the signal to an estimate of chlorophyll-*a* concentration remains a critical step and given the problems associated with analytically measuring chlorophyll-*a*, this is not trivial. Gowen *et al.* (1983b)

also point out that measurements of chlorophyll-*a* using fluorometry may overestimate true chlorophyll-*a* as other breakdown products, for example chlorophyllide-*a*, also contribute to the signal.

It is now widely accepted that high-performance liquid chromatography (HPLC) is the most accurate approach for quantifying photosynthetically active pigments in sea water but HPLC is more expensive and time consuming compared with other methods. It is not currently available as an instrument which can be deployed on a mooring or towed body.

CPR Greenness Index

The Continuous Plankton Recorder (CPR) also provides a measure of phytoplankton concentrations (Batten *et al.*, 2003). The CPR is towed behind ships of opportunity and collects plankton onto a silk which are subsequently analysed at the Sir Alister Hardy Foundation for Ocean Science (Plymouth, UK). In the north-eastern Atlantic the CPR has been operated since the 1940s and it now comprises the largest marine ecological monitoring dataset in existence.

Silks from the CPR are visually scored for colour into 4 'greenness' categories (the CPR phytoplankton colour index). The categories used at marking are 0,1,2,3 but it has been shown that in terms of colour intensity this corresponds to multiplying factors of 0, 1, 2 and 6.5 times pigment intensity (Hays and Lindley, 1994). These latter values are stored as the corrected-PCI values and are thus on a linear scale and can be arithmetically averaged over multiple samples.

Although this seems quite a crude approach, inter-calibrations with measured chlorophyll (from underway fluorometry) and from phytoplankton cell counts show that the PCI has a linear relationship with chlorophyll concentration as long as the chlorophyll concentration is less than 5.5 mg m⁻³ and the number of phytoplankton cells on the silks is low (for comparison levels of up to 16 mg m⁻³ are observed in Division VIa coastal waters, Gillibrand *et al.*, 2003).

The relationship between PCI and chlorophyll is also affected by the relative taxa mix in the area (Hays and Lindley, 1994). A recent inter-comparison of chlorophyll trends in the PCI and SeaWiFS satellite data from the North Sea (including coastal waters) showed general agreement in trends (McQuatters-Gollop *et al.*, 2007). Thus the PCI can reveal chlorophyll patterns with an upper bounded limit of around 5-6 mg m⁻³.

A further complication is that the CPR is towed at a fixed depth of around 10 m but, in summer stratified waters, significant amounts of phytoplankton are found just above the thermocline (Reid *et al.*, 1990; Weston *et al.*, 2005). This layer of phytoplankton is sustained by diffusion of nutrients across the thermocline and persists throughout the summer even though nutrient depletion in the surface water layer as a whole has led to a decline in surface chlorophyll.

Remote Sensing

Finally, remote sensing of ocean colour from satellites is a powerful method which allows coverage of large sea-areas. There are a number of algorithms for converting ocean colour to chlorophyll concentration and for subsequently using this information to estimate primary production (see Section 4 for a more detailed discussion). Estimates of net primary production are derived from chlorophyll concentration using a temperature-dependent description of chlorophyll-specific photosynthetic efficiency (net primary production is the rate of photosynthetically fixed carbon minus losses due to respiration and is usually reported in units of $\text{mg C m}^{-2} \text{ day}^{-1}$). At a global scale overall patterns in production between different predictive models show good agreement but at regional scales larger differences can be apparent (Carr *et al.*, 2006). The application of such algorithms to open ocean (Case I) waters is well established but there are problems in coastal (Case II) waters where other pigments (mainly from terrestrial sources) and turbidity due to sediments cause interference (Tilstone *et al.*, 2005).

In terms of evaluating long-term changes, the system of earth observation satellites is a relatively recent development. The Coastal Zone Color Scanner (or CZCS) became operational in November 1978. It was designed to operate for one year (as a proof-of-concept), but in fact remained in service until June 1986. Readily available time-series for satellite derived ocean colour data generally start around 1997 with the deployment of SeaWiFS (Sea-viewing Wide Field-of-view Sensor) on the OrbView-2 satellite. In addition, data from MODIS (MODerate resolution Imaging Spectroradiometer) carried on-board the Terra and Aqua Earth Observation satellites became available from 1999 and 2002 respectively.

Earth observation data for chlorophyll and primary production estimation can therefore only be used for evaluating relatively recent trends in primary productivity. For the west of Scotland satellite coverage is also often limited by cloud-cover (Nickell *et al.*, 2010). Satellites also only measure the colour of the water at the surface but, as mentioned above, significant amounts of phytoplankton are found just above the thermocline in summer stratified waters (Reid *et al.*, 1990; Weston *et al.*, 2005). In-situ instrumentation or collection and analysis of depth discrete water samples therefore remain the only methods for estimating levels of sub surface chlorophyll and primary production.

FRRF

Recently fast repetition rate fluorometry (FRRF) has been proposed as a new method for measuring primary production using *in situ* instrumentation. FRRF works by tracking the changes in Chl a fluorescence that are induced by a series of pulsed flashes from an array of blue LEDs. However FRRF really measures electron transfer rate and although it provides useful data on community photosynthetic activity, converting this to measures of carbon fixation is challenging (Suggett *et al.*, 2009).

2.8.2 Seasonal Patterns in Chlorophyll and Primary Production

Sea Lochs

In terms of full seasonal descriptions, Grantham (1981) measured chlorophyll levels monthly in Loch Eil and the upper Firth of Lorne as part of a program to investigate the impacts of a pulp mill at Annat Point from November 1975 - October 1976.

Chlorophyll levels throughout the year were higher in Loch Eil, reaching a maximum of 5.5 mg m^{-3} , compared to a control site in the Firth of Lorne (where the maximum recorded was 2.6 mg m^{-3}).

Gowen *et al.* (1983a) measured chlorophyll levels in Loch Ardbhair between January 1981 and July 1982. Chlorophyll concentrations peaked at around $2\text{-}3 \text{ mg m}^{-3}$ in April/May and declined by September to $<0.5 \text{ mg m}^{-3}$. Much higher peak concentrations had previously been reported for other sea lochs such as Loch Etive (Solórzano and Ehrlich, 1977a; Gowen *et al.*, 1983b).

Chlorophyll levels have also been measured monthly for the whole of 1991 in Loch Linnhe and the upper Firth of Lorne (Rees *et al.*, 1995; Watts *et al.*, 1998). Surface concentrations were highest in May, reaching $5\text{-}6 \text{ mg m}^{-3}$ in the outer basin, followed by a decline to around $1\text{-}2 \text{ mg m}^{-3}$ in the summer. The variability between different sea lochs shows that phytoplankton growth is not only controlled by the availability of nutrients and light, but by the amount of freshwater run off and flushing time.

Shelf Waters

Further offshore, the general seasonal patterns in chlorophyll standing stock in the Sea of the Hebrides and the Minch have been described by Gillibrand *et al.* (2003). Chlorophyll-*a* concentrations were measured using a combination of in-situ fluorometry and analysis of discrete water samples on a series of cruises from July 1996 to September 1998. In the spring, elevated levels of chlorophyll were observed in coastal waters (up to 16 mg m^{-3}) compared with offshore waters. Patches of chlorophyll were also observed in the central parts of the Sea of the Hebrides and the Little Minch. These appeared to be associated with areas of warmer, more saline Atlantic water. In areas of elevated chlorophyll-*a*, dissolved nitrate and phosphate concentrations were markedly reduced. By summer, chlorophyll-*a* throughout the surface waters had declined to a maximum of around 2 mg m^{-3} .

During the summer slightly higher levels were observed along the northwest mainland coast. By the autumn chlorophyll-*a* levels had increased again, particularly along the Scottish mainland coast and to the south-west of the Outer Hebrides. During the winter, the amount of chlorophyll-*a* in the water was negligible throughout the region whilst deep-water chlorophyll-*a* levels were low throughout the study (around 0.2 mg m^{-3}).

Chlorophyll levels have also been measured on some of the Ellett line cruises. An example shown in Kennington and Johns (2007) from May 2001 shows the typical situation with higher chlorophyll levels close to the coast and some elevation offshore in the surface waters.

More detailed analysis of the spatial patterns in surface chlorophyll can now be undertaken using data from the SeaWiFS and Aqua-MODIS remote sensors. Such images clearly show the fine structuring in phytoplankton blooms occurring off the shelf edge. However, even in this image there is considerable cloud obscuring the waters to the West of Scotland (Figure 2.14).

Phytoplankton Species

Seasonal changes in the species mix of the phytoplankton have been described in a number of studies. The spring bloom in the shelf seas and sea lochs is generally diatom dominated and in the sea-lochs addition of silicate from terrestrial run-off can prolong the bloom. During the summer the phytoplankton communities of the coastal waters are typically dominated by dinoflagellates. Smaller nanoplankton can also be important members of the communities at all times of the year. They may be of importance as food for early stage cod larvae. Beyond the shelf break the phytoplankton communities are composed of more oceanic species. Again the spring blooms offshore tend to be dominated by diatoms but the blooms tend to be terminated by silica limitation. Coccolithophorids often become more important in the summer as part of a mixed community also including diatoms and dinoflagellates (Kennington and Johns, 2007).

2.8.3 Longer Term Trends

As mentioned previously, for analysis of long-term trends, the CPR remains the only really useful dataset. Figures 2.15a-d show the locations of analysed CPR samples to the west of Scotland from 1940 to 2008 in 20 year blocks. Figure 2.16 shows the trend in the numbers of samples analysed over time. One sample corresponds to a tow of approximately 10 nautical miles (Batten *et al.*, 2003). It can be seen that coverage is generally sparse, especially in the central areas of Division VIa. This sparseness means it is not recommended to apply spatial interpolation techniques to these data. Instead temporal trends have been analysed over the time series for all the available data within the ICES Division VIa boundary. It must be remembered that the resulting averages will be biased towards conditions in the southern and northern extremes of the region.

From 1946 until the early 1970s the average number of CPR samples analysed per month for the whole of ICES Division VIa was around 20 but this declined rapidly in the 1970s to a minimum average of around 10 samples per month (Figure 2.16). The decline in coverage is stronger for the shelf waters (east of 9°W) although for the open ocean and in recent years there are more months when no samples were analysed. Since 2005 there has been

a slight increase in sampling intensity but not to the levels achieved during the early parts of the survey.

Seasonal Blooms

Using CPR data, Colebrook (1986) described the patterns in phytoplankton abundance in the offshore waters to the west of Scotland as a classical spring bloom followed by a summer decline and subsequent autumn bloom before declining to the winter minimum. Over the shelf waters, the seasonal patterns in overall standing stocks were similar, but of lower magnitude, than for the open ocean and the peak of the spring bloom also occurred about one month earlier.

Updating this account using the CPR Colour Index there does not appear to have been any noticeable shift in the timing of the spring bloom in ICES Division VIa as a whole over the last 60 years with the bloom usually occurring in April or May over the shelf waters (Figure 2.17b) and May or June in the deeper waters (Figure 2.17c).

In many years a second autumnal peak is evident, usually between August and October. Care should be taken not to over interpret Figures 17a-17c as some of the inter-annual variability will be the result of differences in sampling intensity. There is, however, some evidence that the decline in near surface chlorophyll levels during summer has weakened over the shelf waters since the mid 1980s so that the spring and autumn blooms are not now so clearly differentiated. This change has also been noted for the North Sea and Central North Atlantic using data up to 1995 (Reid *et al.*, 1998).

The extra 13 years of data shown in this report indicate that this change has persisted. The timing of the change in the mid 1980s also corresponds with the onset of recent surface warming (Figure 2.12). This suggests that it may be linked to the levels of inflow of Atlantic water (Figure 2.5) and thus to regional climate forcing.

Considering the central North Sea, Edwards and Richardson (2004) showed that whilst timing of the diatom blooms was relatively insensitive to sea temperature, and had remained relatively constant over time, the timing of dinoflagellate blooms was linked with temperature. It was suggested that the increase in surface greenness index in the North Sea might be related to an increase in flagellates during the summer months. Further analysis of CPR data from Division VIa are required to show if similar changes have occurred.

Based on SeaWiFS.R2009 remote sensing data for the whole of Division VIa (Figure 2.17d), the spring bloom is much less clearly defined and the peak of chlorophyll concentration seems to occur about one month later than suggested by the CPR data. There is also little evidence for an autumnal bloom. However, as mentioned previously, the remote sensed time-series is short and because of the amount of cloud cover to the west of Scotland, remote sensing data will tend to be biased towards the summer months although coverage between May-September appears to be reasonably good (Figure 2.18).

Primary Production

Estimates of primary production based on estimated chlorophyll from remote sensing are available from the ocean productivity group at Oregon University (<http://www.science.oregonstate.edu/ocean.productivity/>). However, these estimates only go back to 1997 (SeaWiFS) or 2002 (MODIS). Furthermore application of different models for converting chlorophyll to primary production produce rather differing results (Figure 2.19).

The Vertically Generalized Production Model (VGPM) follows Behrenfeld and Falkowski (1997). The VGPM is a "chlorophyll-based" model that estimates net primary production from chlorophyll using a temperature-dependent description of chlorophyll-specific photosynthetic efficiency.

The depth to which sufficient light penetrates to support photosynthesis (the euphotic depth) is calculated using the Morel and Berthon (1989) Case I (open ocean) model. This model estimates the euphotic depth from surface chlorophyll concentrations and is based on empirical equations fitted to field data.

In practice, total water column chlorophyll concentration is calculated from satellite surface chlorophyll using a formula that distinguishes between lower and higher chlorophyll waters. Then, given the amount of total chlorophyll, the euphotic depth is estimated, again using separate equations for lower and higher total chlorophyll conditions.

The only difference between the Standard VGPM and the Eppley-VGPM is in the temperature-dependent description of photosynthetic efficiencies. As the name implies, the Eppley-VGPM uses an exponential function that is traceable to a paper by Eppley (1972). In that paper, Eppley evaluated changes in phytoplankton growth rates over a range of temperatures for a wide variety of species. What he noticed was that the maximum specific growth rate achieved could be related to temperature and that this caps growth rates across the measured temperature range. For this reason, the estimates of net primary production from the Eppley-VGPM tend to be markedly lower than from the standard VGPM (Figure 2.18).

In 2005, the Ocean Productivity Group introduced a different approach to productivity modelling that uses remote sensing retrievals of particulate scattering coefficients to estimate phytoplankton carbon concentration, thus replacing chlorophyll as the metric of biomass. This new approach was made possible by two recent developments: (1) the observation that total particulate carbon concentration or phytoplankton biomass co-varied with light scattering properties, and (2) the construction and application of spectral matching algorithms to satellite data for simultaneously retrieving information on particulate backscattering scattering coefficients, phytoplankton pigment absorption, and coloured dissolved organic carbon absorption. The importance of these two developments is that they allow phytoplankton carbon biomass to be estimated from backscattering coefficients and phytoplankton growth rates to be estimated from chlorophyll to carbon ratios - thus net

primary production is described as the product of carbon biomass and growth rate, rather than the traditional product of chlorophyll and photosynthetic efficiencies. The model is referred to as the Carbon-based production model (CbPM).

All the algorithms described above apply to open ocean waters. In addition a large range of algorithms have been developed for application in coastal waters to try and correct for interference from turbidity and non-photosynthetic pigments. Unfortunately sufficient ground-truthing data to rigorously test the results are only available for a limited number of sea areas (Tilstone *et al.*, 2005).

Because of the relatively short time-series and the widely differing model results it was not considered worthwhile analyzing the temporal changes in primary production estimates from remote sensing further at this stage.

Summary

In conclusion, the general seasonal trends in chlorophyll standing stock for both the sea-lochs and coastal areas in ICES Division VIa are known based on studies covering the spring bloom up to full seasonal coverage. Analyses of longer-term trends, however, suffer from a lack of data. The only dataset allowing examination of long-term trends is the CPR colour index.

Although spatial coverage of Division VIa by the CPR is not good, the time-series does go back to 1948 and the methods used for assessing the colour of the sampling silks have remained consistent over this period. The CPR colour index indicates a shift in patterns of chlorophyll standing stock around the mid-1980s with the largest changes in the shelf waters. This change has previously been noted for adjacent sea areas such as the North Sea. Furthermore this change coincides with the recent warming of the surface waters which in turn appears to be linked with the amount of Atlantic water entering the area. As postulated for the North Sea, the increase in the colour index during summer may be related to changes in the composition of the post-spring bloom phytoplankton community with a shift toward increased abundance of flagellates during the summer.

Since 1998, remote sensing data have become available via SeaWiFS and MODIS. Estimates of surface chlorophyll from SeaWiFS indicate a slightly different pattern compared to the CPR; the spring maximum seems to occur around a month later according to the remote sensing data. This difference could be artefactual or related to different spatial coverage by the CPR and remote sensing or due to differences in the water depth sampled. Examination of this would require more research, in particular matching up CPR sample locations with SeaWiFS and MODIS grid cells, as has been done for the North Sea.

Unfortunately the relatively sparse spatial coverage of the CPR to the west of Scotland, combined with the problems of cloud cover affecting remote sensing, may mean that insufficient match-ups are obtained to reach firm conclusions. Primary production estimates

are available for recent years based on remote sensing data. However, the use of different models for converting estimated chlorophyll to production leads to a wide range of estimates. Further research is needed to evaluate the reliability of algorithms for estimating primary production for waters to the west of Scotland, as has been undertaken for the Irish Sea. Such research would need access to more ground-truthing data than are currently available (see section 4 for further recommendations for future work on this topic).

There are examples where relationships between primary production and gadoid year-class strength have been found. Thus for Nova Scotian haddock the timing of the spring bloom appears to correlate with the survival of juvenile haddock (Platt *et al.*, 2003). The timing of the bloom was established using data from CZCS and SeaWiFS satellites. On George's Bank (USA), Friedland *et al.* (2008) suggested a link between the strength of the autumnal bloom and haddock year-class strength in the following year. It was proposed that this affected the food available to the maturing fish which in turn affected the amount and quality of eggs produced the following spring. Although all these proposed mechanisms remain tentative, it is clear that potential links between environment, primary and secondary production and fish survival can only be explored when there are robust time-series available for both fish recruitment and environmental variables.

2.9 Zooplankton

2.9.1 Seasonal Patterns

Offshore Waters

Zooplankton have been sampled periodically in ICES Div VIa. Early studies were mainly from deep water. Mauchline produced a series of papers on the biology and ecology of the plankton of the Rockall Trough (Mauchline 1984, 1985, 1986a, 1986b) based on samples collected between 1973-1979. Mauchline suggested that the residence time for organisms in the Rockall Channel would be 3-6 months and that the channel might act as a reservoir for oceanic species which would periodically be carried to the north-east towards the Norwegian Sea. The Mauchline sample archive and associated papers are currently being catalogued and digitised by SAMS.

Sea Lochs

A number of studies have focused on the plankton of the sea lochs. For example in 1991, Fisheries Research Services (now Marine Scotland Science) undertook monthly zooplankton sampling in Loch Linnhe over a whole year (Heath, 1995). Despite the semi-enclosed nature of the loch, it was found that exchanges with the open sea were an important factor affecting the population dynamics of the plankton. Over the whole year the loch acted as a net sink for small zooplankton and a net source of larger size classes.

2.9.2 Longer Term Trends

As for longer-term studies on phytoplankton, the CPR remains the only source of long-term zooplankton data covering the shelf and offshore areas. As noted by (Kennington and Johns (2007) there tend to be distinct differences between the zooplankton communities in the deep waters to the west of the shelf edge and in the shelf and coastal areas.

Offshore Waters

Based on CPR data the deeper waters tend to be dominated by copepods such as *Acartia* spp. and *Calanus* spp. *Metridia lucens*, *Oithona* spp. and members of the *Para-Pseudocalanus* complex are also important. Euphausiids are common in the deeper waters but are not sampled very efficiently by the CPR because of its relatively small mouth opening. The thecosomatan *Limancina retroversa* (also known as the sea butterfly) is very common.

Shelf Waters

In relation to the early life stages of cod, haddock and whiting, zooplankton over the shelf areas will be of more importance since the spawning grounds for these fish do not occur in very deep water such as the Rockall Trough (Coull *et al.*, 1998; Wright, 2005). Numerically the zooplankton of the shelf waters are dominated by *Para Pseudocalanus* spp. with *Calanus* spp. and *Acartia* spp. also being important.

Historically Reported Trends in Zooplankton

Colebrook (1986) reviewed trends over the period 1950-1982 comparing samples from the Rockall Channel with those collected over the shelf to the west of Scotland. Further comparisons were made with other areas around the UK, for example the North Sea.

In the northern North Sea the copepod *Calanus finmarchicus* is one of the most common zooplankton species and is important as a prey item for developing gadoids, particularly cod (Heath and Lough, 2007). Colebrook's (1986) analysis found that although *C. finmarchicus* occurred to the west of Scotland, it was at much lower abundances compared with its congener, *Calanus helgolandicus*. This may be significant in relation to food sources for larval gadoids as the timing of nauplii production differs between these two species and *Calanus helgolandicus* adults have lower lipid reserves compared with *Calanus finmarchicus*. This is discussed further later in this section.

Colebrook (1986) noted that the seasonal cycle in zooplankton abundance appeared rather similar for the ocean and shelf areas with maxima occurring between July-September. The increase in abundance during the spring period was slightly greater for the shelf waters and seemed to reflect higher over-wintering levels.

Inter-annual changes in abundance showed similar patterns for the open ocean and shelf waters and there was a clear downward trend in the abundance of both phytoplankton and zooplankton taxa (data covering 1950-1982). Colebrook (1986) hypothesised that this trend was related to the sizes of the overwintering stocks of both phytoplankton and zooplankton in the area.

Latest CPR Data

In this report we update these analyses using CPR data covering 1948-2008 for the main species identified on the west of Scotland. Over the whole time-series the sampling coverage is as shown in Figures 2.15-2.16. as noted previously sampling is biased towards the northern and southern parts of ICES Division VIa with a decline in sampling intensity around the mid-1970s. All available CPR zooplankton data for ICES Division VIa were extracted and examined for major trends. Samples were separated into shelf and ocean using the 9°W meridian as a separator.

Figures 2.20a-2.20c show the long-term trends in the abundance of the copepod taxa which are numerically dominant in the waters to the west of Scotland. The plots are roughly ordered in terms of overall numerical dominance of each taxa. Annual means were calculated based on monthly means in order to reduce bias towards months with more CPR samples. Because winter months tend to have lower sampling intensity, basing annual means only on the raw data tends to lead to overestimation of the mean annual abundance. In some years there were months with no CPR samples; values for missing months were first estimated by linearly interpolating between adjacent months before computing the annual means. Comparison with plots where missing data were not interpolated (not shown) showed that in most years this only makes a small difference to the result.

Although other prey may be eaten by young gadoids, it is generally considered that copepods form the most important items in the diet of the larvae with an increasing importance of euphausiids for the post-larval stages (Rowlands *et al.*, 2006; Heath and Lough, 2007; Rowlands *et al.*, 2008). Total copepod abundance has fluctuated on an inter-annual basis throughout the time-series (Figure 2.20a) but there is also some evidence for lower frequency underlying oscillations.

Trends in the shelf waters reflect rather closely the patterns in the deeper waters although with a somewhat reduced variability. In both areas overall copepod abundance appeared to be highest in the early 1950s followed by a decline until the late 1970s. Since then there was some recovery up to 1985 before another decline until the present.

Examining trends in individual taxa it can be seen that these overall patterns are driven by slightly different dynamics in the deep and shelf areas. The pattern in total copepod abundance in the off-shelf area is mainly driven by changes in *Acartia* spp. In contrast changes in the abundance of *Para-Pseudocalanus* spp. dominate the patterns in the shelf waters. The abundance of *Para-Pseudocalanus* spp. in the shelf waters was generally

highest in the 1950s, mid 1960s and mid 1980s. Since 1992 abundance appears to have been rather low.

The other species often cited as being of particular importance for cod larval feeding is *Calanus finmarchicus*. Numbers of total *Calanus* spp. have fluctuated quite strongly on an inter-annual basis with some decline in overall abundance over time, the decline being greater for the shelf waters. However, both *Calanus finmarchicus* and *Calanus helgolandicus* occur in ICES Division VIa and as they have somewhat different dynamics should be considered separately. These two species began to be separated in the CPR samples from 1958 and are considered in more depth later on.

Numbers of *Oithona* spp. fluctuated strongly but did not show much trend up to about 1995 (Figure 2.20b). Since then, however, their abundance appears to have collapsed. This species has been recorded in the diet of larval cod from the Irish Sea where it formed around 20% of gut contents by numerical abundance (Thompson and Harrop, 1991).

The abundance of *Centropages typicus* and *Metridia* appears to have fluctuated over time with no obvious trends. The numbers of *Temora longicornis* recorded were generally much higher in the shelf waters than in the deeper waters. In the shelf waters their abundance appeared to fluctuate strongly on an inter-annual basis but without any obvious long-term trends.

Trends in Calanus spp.

As mentioned previously trends in *Calanus* spp. have received considerable attention since *Calanus finmarchicus* is often the dominant copepod in northern waters. The ratio of *Cal finmarchicus*:*Cal helgolandicus* has also been proposed as an index of climate change. The abundance of *C. finmarchicus*, both on the shelf and deeper waters of ICES Division VIa has shown some decline since the 1960s although 1979 appeared to be an exceptionally positive year. The overall decline in abundance of *C. finmarchicus* has not been as severe as in the northern North Sea.

In contrast there have been large changes in the abundance of *Calanus helgolandicus* which has increased strongly since the mid-1990s. *C. helgolandicus* is associated with warmer waters than *C. finmarchicus*. It also tends to produce young later in the year compared with *C. finmarchicus* and it has been suggested that this may lead to a mismatch between larval fish and their prey (Bonnet *et al.*, 2005).

Euphausiids

As mentioned previously euphausiids become important in the diet of cod around the time of metamorphosis. Euphausiids were most abundant in CPR samples from the shelf waters in the early 1950s. Since then abundance has fluctuated about a rather lower level. It must be

noted however that the CPR does not sample euphausiids very efficiently as they are adept at escaping from the path of plankton samplers.

Comparison of the Annual and Monthly Trends in Abundance of Key Zooplankton

Copepod Nauplii

The patterns in nauplii (figure 2.21) do not exactly mirror the decline in total copepod abundance. Nauplii abundance increased around the mid 1970s before declining again around the mid-1990s for both shelf and deeper waters. The nauplii data prior to 1960 seem unusual and are probably incomplete although this is not mentioned in Batten *et al.* (2003). The apparent increase in nauplii abundance during the 1970s may reflect copepod reproductive rates which are related, at least in part, to water temperatures (Pitois and Fox, 2008). However, due to their small size the CPR does not sample copepod nauplii very efficiently so conclusions concerning the nauplii data can only be tentative.

Total Copepod Abundance

The annual (Figure 2.20.a) and monthly patterns in total copepod abundance (Figure 2.21) for both the shelf and deeper waters both show the decline in abundance over time. In shelf waters there is some evidence that the timing of peak abundance has become earlier in the year. This is more obvious in the deeper waters where the peak used to occur in August but appears to have shifted to June/July since the mid 1980s.

The annual and monthly patterns for total copepod abundance from the oceanic waters are closely reflected in the pattern of abundance of *Acartia* spp. whilst the patterns from the shelf seas more closely resemble *Para-Pseudocalanus* spp. abundance (Figure 2.22). This confirms the conclusion drawn from the analysis of the annual averages. The peak abundance for *Acartia* spp. tends to be around June-July and for *Para-Pseudocalanus* spp. in June. There is little evidence that the timing of the peak has shifted substantially.

Changes in the abundance of *Calanus* spp. in the shelf waters are shown in Figure . There appears to have been an increase in abundance of copepodite stages I-IV in recent years with the peak occurring in May/June. Because copepodite stages I-IV of *C. finmarchicus* and *C. helgolandicus* cannot be distinguished without using molecular methods it is unclear which of the two species are contributing to the recent increase. However, *C. helgolandicus* is normally thought to reproduce somewhat later in the year (Bonnet *et al.*, 2005; Helaouët and Beaugrand, 2007).

For *Calanus finmarchicus*, there is evidence of a slight overall decline as previously observed in the annual analysis (Figure 2.20c) but there have been years of increased abundance throughout the time series. There is little evidence that the timing of peak *C. finmarchicus* abundance has changed over time. The recent increase in *C. helgolandicus* is obvious with the peak abundance occurring around June.

Summary

In summary, the abundance of copepods in ICES Division VIa, as monitored by the CPR, shows evidence for a decline over the times-series (1948-2008). There may also be underlying multi-decadal cycles which are probably related to oceanographic variability across the wider north-east Atlantic. Changes in total copepod abundance in the off-shelf waters are mostly caused by variability in the abundance of *Acartia* spp. whilst variability in *Para-Pseudocalanus* spp. numbers tends to dominate overall patterns in the shelf waters. The abundance of *Para-Pseudocalanus* spp. seems to be particularly low at present in comparison to some historical periods. This could be significant for gadoid larval growth as these species are known to be important prey for the larvae of at least one gadoid species in other areas (Heath and Lough, 2007).

The abundance of *Calanus finmarchicus* has declined over time but not to the extent seen in the northern North Sea. The CPR atlas suggests that abundance of *Calanus finmarchicus* in waters to the west of Scotland has always been relatively less than in the northern North Sea (Barnard *et al.*, 2004).

In contrast, the abundance of *Calanus helgolandicus* in the shelf waters of ICES Division VIa has increased substantially since the mid 1990s. Whether this copepod forms a significant part of larval and post-larval gadoid diets is unclear although other studies suggest a minor role as its reproductive peak tends to occur later in the year (Helaouët and Beaugrand, 2007).

Regarding the abundance of copepod nauplii and copepodites in the shelf waters during the expected peak of gadoid larval abundance (April-May), there is little obvious evidence of a substantial decline and there may even have been an increase in copepod juvenile stages, possibly linked to increased water temperature raising reproductive rates.

The strongest decline in total copepod abundance appears to have occurred later in the year with abundances in CPR samples in September now being between 10 to 100 times lower than in the 1950s.

Euphausiid abundance has also declined since the 1950s but in this case the most substantial declines occurred earlier in the year so that the peak abundance now occurs around August. These changes may be significant for juvenile gadoid growth since euphausiid larvae begin to be increasingly selected by juvenile haddock (> about 25 mm in length) and larger juvenile cod (> around 35 mm in length) (Gaard and Reinert, 2002).

The analysis presented above demonstrates that significant changes have occurred in the abundance of the dominant taxa of plankton in the shelf waters of ICES division VIa since the 1950s. However, we have only considered abundance data in this report and because prey biomass varies with taxa it would be necessary to convert abundance to biomass to more fully investigate the impacts of these changes in the plankton (Heath, 2007).

It is widely recognised that there are considerable problems in using CPR data to investigate changes in zooplankton biomass (Pitois and Fox, 2006; Pitois and Fox, 2008). As with all net based plankton sampling systems, the CPR does not sample all taxa or developmental stages equally. Smaller taxa and young developmental stages are generally under-sampled due to the mesh size of the CPR whilst larger taxa, e.g. euphausiids, actively avoid the opening of the sampler.

In addition, the sampler is towed at a fixed depth so that vertical migration can bias abundance estimates (Batten *et al.*, 2003). CPR data are therefore most suited to the production of large-scale distribution maps with relative seasonal and inter-annual trends also being well captured for many taxa (Clark *et al.*, 2001; John *et al.*, 2001).

However, because the CPR provides the only long-term broad spatial survey for plankton, both Batten *et al.* (2003) and Pitois and Fox (2006) have attempted to develop correction factors to allow CPR zooplankton data to be raised to estimate true abundance. Applying such correction factors to CPR data for the west of Scotland, Pitois and Fox (2006) found that the trend for an overall decline in annual biomass over time was considerably steepened. This decline in plankton biomass appeared to be driven mainly by the drop in the abundance of medium sized zooplankton such as *Pseudo* and *Para calanus* spp.

Pitois and Fox (2008) then used the corrected CPR data to investigate potential links between prey availability and cod larval growth (up to 40 days post-hatch) around the UK using a bio-energetic model. The larval growth model did not show clear trends over time and did not support the concept that there is a simple relationship between cod larval growth, temperature and the decline in the abundance of larger copepod species, such as *Calanus finmarchicus*. Rather the model suggested that, as temperature increases, there are complex trade-offs between changes in the bio-energetic demands of the cod larvae, the composition of the copepod communities, and copepod reproductive rates.

Although the study had to make a large number of assumptions and the results were far from conclusive, the underlying message was that the reported correlations between cod recruitment, temperature and the abundance of *C. finmarchicus* and euphausiids in the North Sea (Beaugrand *et al.*, 2003) may not be explicable through a relatively simple link of effects on cod larval growth rates.

2.9.3 Gelatinous Zooplankton

The previous sections have related to zooplankton as prey for the early life stages of fish but the zooplankton also includes a number of taxa which may be significant consumers of fish eggs and larvae (Heath, 1992; Bunn *et al.*, 2000).

In terms of abundance of non-copepod zooplankton, the thecosomatan, *Limnocalanus macrurus*, is very common in Division VIa although its abundance in CPR samples has declined since the mid 1980s. This sea butterfly is known to predate on small copepods,

tintinnids and other small plankton but there does not seem to be any data on whether it takes fish eggs or larvae.

The most important group of planktonic predators on fish eggs and larvae are thought to be gelatinous zooplankton including chaetognaths, ctenophores and medusae. Jellyfish blooms have been reported to have caused problems for the aquaculture industry on the west of Scotland and Nickell *et al.* (2010) recently reviewed the available information. Blooms of the moon jellyfish (*Aurelia aurita*) are commonly reported in the inshore lochs of the western Scottish coast whilst ctenophores can also be abundant in the coastal waters with *Pleurobrachia pileus* and *Bolinopsis infandibulum* being the most commonly recorded.

As well as common inshore species there have been periodic influxes of oceanic jellyfish such as *Pelagia noctiluca*. However, data on the distribution and occurrence of gelatinous predators offshore is almost completely lacking. This is principally because gelatinous predators tend to be damaged by plankton nets.

Data from the continuous plankton recorder (CPR) gives some indication of coelenterate abundance by analysis of the % of samples with remnant coelenterate material on the CPR silks. These data show that the highest coelenterate index scores are recorded from the Clyde region (Kennington and Johns, 2007). Offshore the distribution is patchy.

Not mentioned above are the siphonophores which comprise colonies of integrated polymorphic hydroid and medusoid individuals, the Portuguese Man-o'-War (*Physalia physalis*) is one example. Relatively few species are found in British waters although *Veella veella* (By-the-Wind Sailor) has been recorded in large numbers in the Clyde. Both these species are native to warmer waters and it is likely that their distribution will spread to more northern waters if sea temperatures rise.

As with other members of the zooplankton community, jellyfish and their relatives are prone to climatic shifts and it has recently been shown that the abundances of two species (*A. aurita* and *Cyanea capitella*) recorded over a 15 year period in waters to the north of Scotland were positively correlated with the North Atlantic Oscillation (Lynam *et al.*, 2005).

A recent development is the application of molecular markers to identify the species of remnant tissue on CPR silks. Using this technique Licandro *et al.* (2010) were able to identify the warm-water scyphomedusa *P. noctiluca* as being a particularly important contributor to the recent (post-2002) increase in occurrence of cnidarian tissue on CPR samples from the north-eastern Atlantic. Despite their undoubted importance in marine food webs, as predators and prey, gelatinous zooplankton are probably the least studied component of pelagic communities.

2.9.4 Interactions between Zooplankton and West of Scotland gadoids

Food availability for the early life stages of fish is generally thought to be an important factor affecting fish year-class strength. Developing gadoid larvae feed mainly on copepods, gradually increasing the size of the prey selected as the larvae develop. At the onset of feeding the main prey items are copepod nauplii and sometimes phytoplankton (Marak, 1960). As larvae grow beyond around 8 mm in length they show a gradual switch from copepod nauplii to increasing proportions of copepodite and adult copepods in the diet.

Cod and whiting show a similar transition of prey species preference as they develop with a clear preference for *Calanus* spp. around the time of metamorphosis. This occurs even in *Calanus* poor waters and probably reflects the better energy content to energy expended ratio in capturing these larger copepods (Rowlands *et al.*, 2006; Rowlands *et al.*, 2008).

There are some differences between cod and whiting, however, as the latter are able to take adult *Para/Pseudocalanus* at a smaller larval size compared with cod. In general, whiting may be more opportunistic than cod (Shaw, 2008). Haddock larvae are even more generalist as they grow and copepod nauplii continue as a component of the diet (Rowlands *et al.*, 2006; Rowlands *et al.*, 2008).

In all three species there is a gradual switch as the fish grow to include even larger prey such as euphausiids and fish in the diets (this transition occurs around 35 mm fish length in cod and in haddock at around 25 mm). When expressed in terms of the number of organisms consumed, the diet in the pre-settlement stages of all three gadoids is usually dominated by copepods and other crustacea but when expressed in terms of wet weight, fish are seen to be the main component of the diet of larger pre-settlement cod and whiting. Piscivory is less pronounced in haddock (Bromley, 1997). Cannibalism has also been recorded in juvenile cod (Uzars, 2000) and whiting (Bromley, 1997; Lebour, 1918) and may be a significant factor contributing to density-dependent controls on the population sizes of these species.

Because fish larvae feed mainly on zooplankton, many studies have looked for links between plankton based indices and subsequent year-class strength. In particular the importance of *Calanus finmarchicus* in the diet of larval cod has led to attempts to link the abundance of this copepod to cod recruitment success across many north-eastern Atlantic stocks (Sundby, 2000). One of the most cited examples is from the North Sea where there appears to be a good correlation between cod recruitment and a multi-factor index which includes the abundance *C. finmarchicus* (Beaugrand *et al.*, 2003). However, a universal role for *Calanus finmarchicus* is questionable as cod stocks in the south of the species range feed predominantly on other taxa such as *Pseudo-* and *Para-calanus* spp. (Heath and Lough, 2007; Shaw *et al.*, 2008).

Although the abundance of *Calanus finmarchicus* has declined on the west of Scotland since the 1950s, the decline does not appear to have been as dramatic as in the northern North

Sea. Indeed in recent years the abundance of *Calanus* copepodites (un-specified) during May/June appears to have increased, possibly as a result of increased reproductive rates per copepod as waters have warmed.

The abundance of *Calanus helgolandicus* has also increased significantly since the mid-1990s. *C. helgolandicus* is characteristic of warmer waters and typically increases in abundance in the summer rather than the spring (Helaouët and Beaugrand, 2007). This difference in timing of reproduction has given rise to the idea that a change from a *C. finmarchicus* to a *C. helgolandicus* dominated system will result in a mis-match between the feeding demands of cod larvae and the availability of suitable prey (Richardson, 2008).

As far as we know, the bulk of cod spawning in Division VIa takes place in March so the abundance of small larvae probably peaks during late March or early April (Wright, 2005). At first the larvae will feed on copepod nauplii and copepodites but by mid-June they should be large enough to prey on larger copepods and even begin the transition to including prey such as euphausiids. The peak in abundance of *C. helgolandicus* in June/July might therefore occur early enough to coincide with the feeding requirements of the larger larvae whilst the abundance of *Calanus* copepodites (un-specified) appears to have actually increased during May/June and should provide a suitable food for the slightly smaller larvae.

However, there has been a substantial decline in the abundance of *Para-Pseudocalanus* spp. in March/April in the shelf waters of the west of Scotland since the 1950s (Figure 2.20b). This may actually be of more importance for the early gadoid larvae than changes in the abundance of *Calanus* spp. The decline in *Para-Pseudocalanus* (Figure 2.21) coincides with the time of year when early stage gadoid larvae should be abundant. If the timing of spawning has also advanced as waters have warmed this would exacerbate any mismatch between the production of the larvae and the availability of *Para-* and *Pseudo-calanus* as prey for the early stage larvae.

Fish larvae are also preyed on by other members of the zooplankton. It is known from studies in other parts of the world that impacts on fish eggs and larvae, particularly by gelatinous zooplankton (including jellyfish), can be significant. However there are virtually no time-series data or ecological studies for ICES Division VIa with which to examine this issue. A recent review on interactions between gelatinous zooplankton and marine aquaculture operations on the west of Scotland concluded that there are significant economic impacts of jellyfish blooms, particularly of the smaller hydromedusae, on the salmon farming industry. This suggests that there are significant populations and outbreaks of gelatinous zooplankton in the area, but that there is insufficient data to evaluate whether the jellyfish populations or the frequency of such events are really increasing (Nickell *et al.*, 2010). Jellyfish have been implicated as important predators on fish eggs and larvae in other parts of the world but there are virtually no data for the west of Scotland to evaluate if gelatinous predators are a significant controlling factor for gadoid year-class strength.

2.9.5 Trends in the Distribution of Fish Larvae from the CPR

The CPR does not sample fish larvae very efficiently due to its small opening. Nevertheless Coombs (1975 and 1980) was able to use CPR data to examine changes in the timing and spatial distribution of some larval taxa for the period 1948-1972 (Coombs, 1975).

Identification of the fish larvae in CPR samples ceased in 1972 but Defra have recently funded a project to re-examine samples from the waters around the UK in order to update the time-series. Although CPR samples from the west of Scotland show reasonable numbers of positive records for clupeoid larvae, there are insufficient positive records for cod, haddock or whiting to allow analysis of shifts in timing or location of spawning.

2.10 Conclusions Concerning Data Availability for Establishing Fish Recruitment to Environment links

There have been a number of short to medium-term studies where nutrient levels, pigments, phytoplankton and zooplankton have been measured in waters to the west of Scotland. From these studies the general seasonal trends are relatively well understood for the sea-lochs, shelf waters and to some extent offshore.

However, for analysis of trends over multi-decadal timescales the CPR remains the only source of data. The number of CPR samples available for analysis is heavily dependent on the ships of opportunity available to tow the CPR recorders and on funding of the survey. Declines in shipping routes inevitably lead to degradation of the survey.

However, given the number of ferries operating between the mainland and the outer islands it is surprising that coverage is so poor to the West of Scotland. Given the fact that the CPR is a relatively cheap method of monitoring plankton, that it is one of the few techniques which can cover large spatial areas and that changes in the plankton are being included as indices for evaluating ecosystem health (which will be needed to demonstrate that Good Ecological Status is being maintained under the European Marine Strategy Directive), every effort should be made to increase the coverage of waters to the west of Scotland using the CPR.

In addition, some fixed plankton sampling stations in the shelf waters (in addition to the Loch Ewe monitoring site) should be established in order to build time-series which will allow the development of methods for converting CPR abundance data to true abundance. The use of novel plankton sampling methods e.g. video plankton recorders or semi-automated plankton collectors (Kilburn *et al.*, 2010) should also be explored.

Further research is also needed on the development and testing of algorithms for converting remote sensing data to chlorophyll and primary productivity estimates for the region. Time-series and ecological studies on gelatinous predators are lacking even though increases in the frequency of blooms have been predicted in response to changes in climate.

Finally, research on the growth and survival of young gadoids in Division VIa is urgently needed. Published data on gadoid early life ecology from the West of Scotland stocks is almost totally lacking hampering efforts to assess the probable impacts of observed changes in primary productivity and in zooplankton community structure on recruitment to these stocks.

2.11 Acknowledgements

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2.12 References

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2.13 Figures

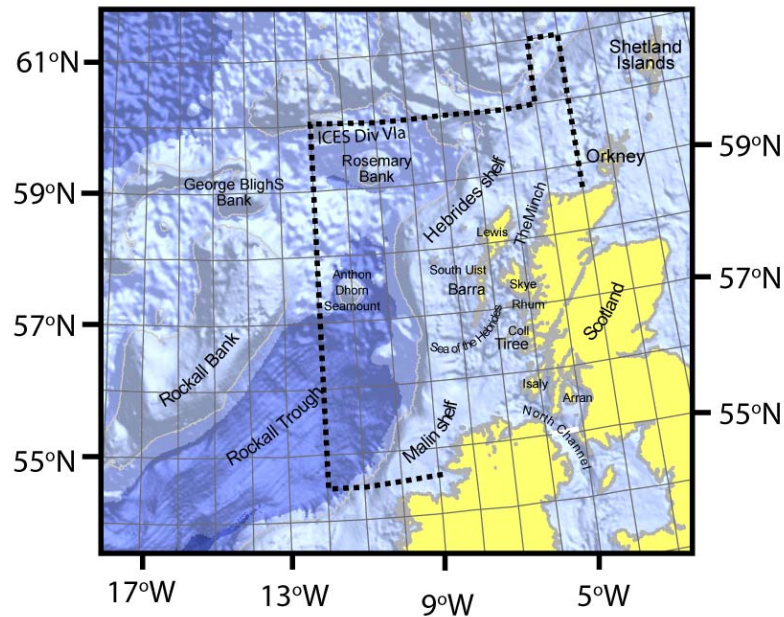


Figure 2.1 General features of the area covered in this review - note that some of the apparent features such as the north-west to north-east striations in the Rockall Trough are artefacts resulting from the underlying GEBCO bathymetry used to prepare the figure (Marks and Smith, 2006).

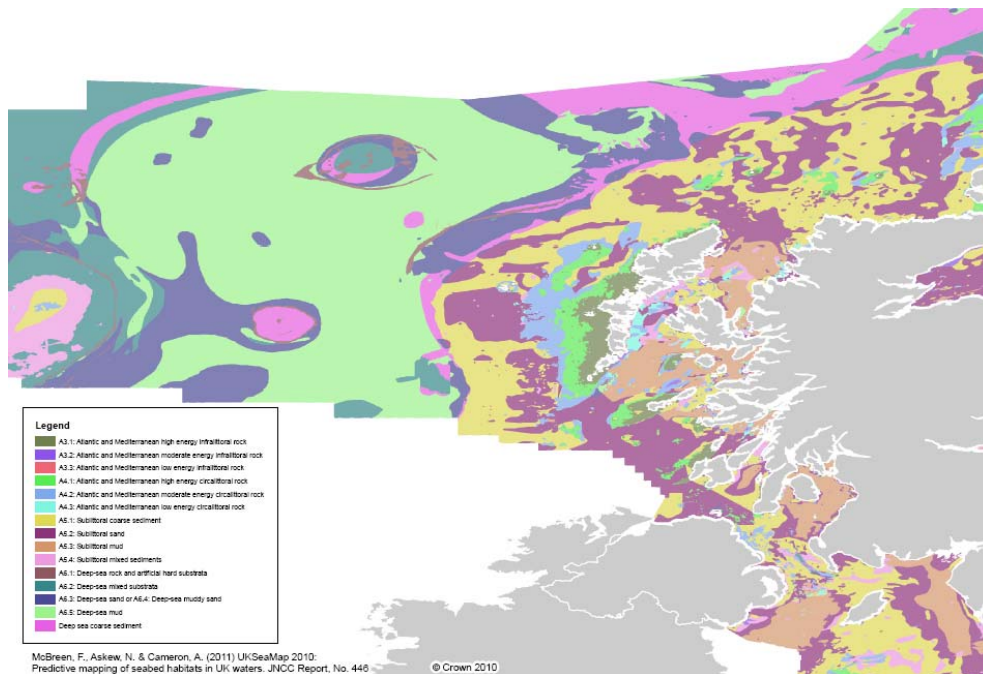


Figure 2.2 Sub-sea sediments in Division VIa (from McBreen, F., Askew, N. & Cameron, A. (2010). Predictive mapping of seabed habitats in UK waters. UKSeamap. JNCC 2 Report, No. 446).

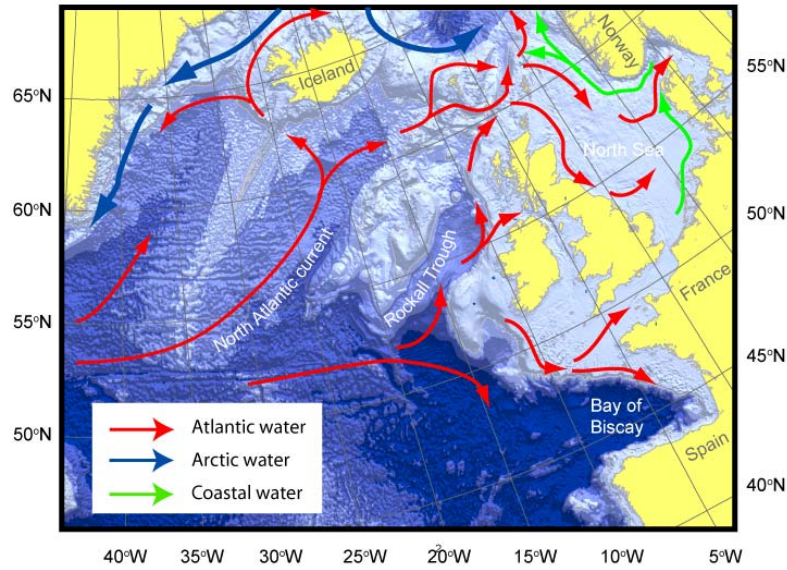


Figure 2.3 Generalised near-surface oceanographic flows in the north-eastern Atlantic (modified from Fox *et al.*, 2009).

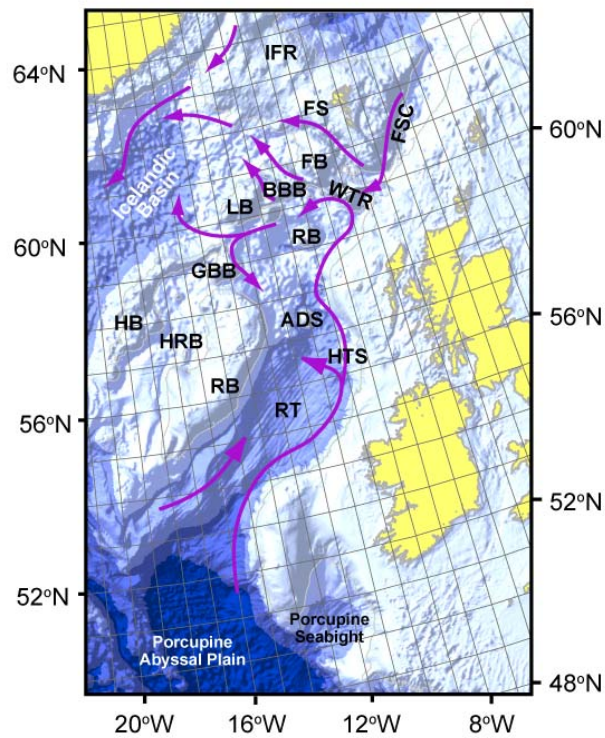


Figure 2.4 The bottom flows in the Rockall Trough and adjacent areas. Bathymetric features labelled; IFR Iceland Faroe Rise; FS Faroe Shelf; FSC Faroe-Shetland Channel; FB Faroe Bank; WTR Wyville-Thomson Ridge; BBB Bill Bailey's Bank; LB Lousy Bank; RB Rosemary Bank; GBB George Bligh Bank; HB Hatton Bank; HRB Hatton-Rockall Basin; ADS Anthon Dohrn Seamount; HTS Hebrides Terrace Seamount; RB Rockall Bank; RT Rockall Trough (modified from Kuijpers, 1998 and Howe *et al.* 2001).

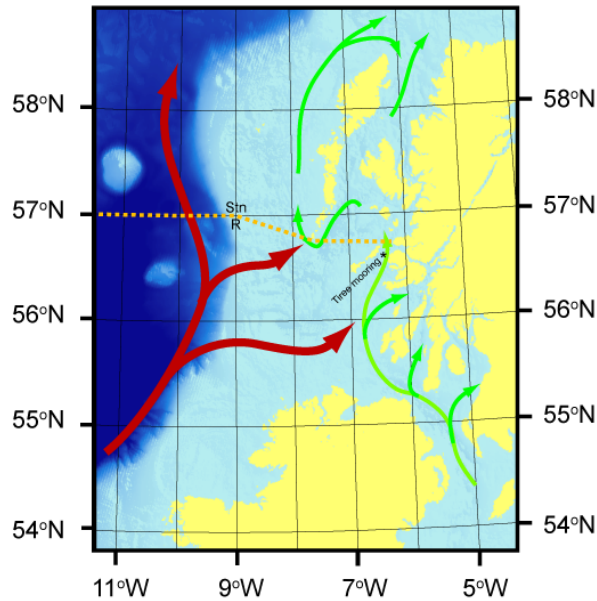


Figure 2.5 Generalised upper ocean flows onto the shelf areas to the west of Scotland. The location of the Ellett Line oceanographic section (dashed line) and location of the Tiree passage mooring are also shown (modified from Inall *et al.*, 2009).

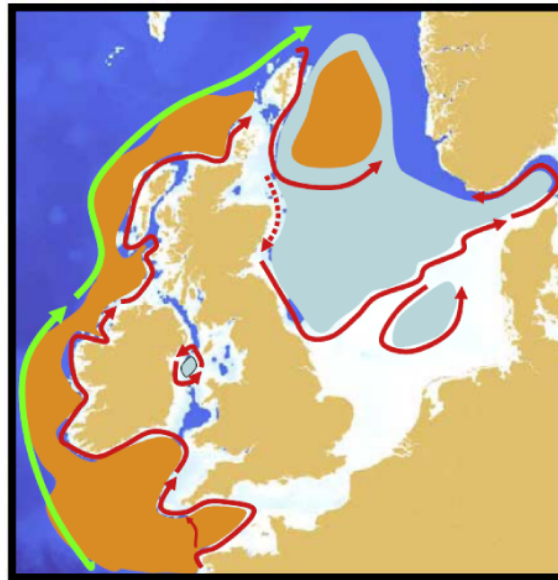


Figure 2.6 Schematic map of principal summer thermohaline transport pathways on the north western European shelf and the cold and salt pools that drive them. Orange shaded areas, regions where seasonally formed bottom dense pools are influenced by both cool winter temperatures and salty oceanic water which has penetrated the outer shelf. Light blue shaded areas, regions where only temperature is responsible for the density of dense water trapped below the seasonal thermocline. Green arrow, European slope current. Red arrows, frontal jets associated with bottom fronts at boundaries of dense cold and salt pools (Hill *et al.*, 2008).

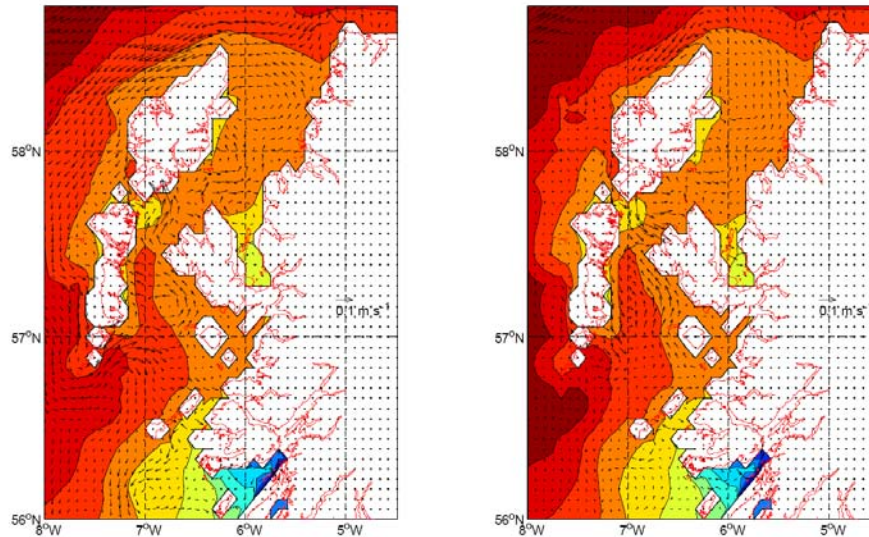


Figure 2.7 Results of oceanographic modelling of flows over the shelf regions (Gillibrand *et al.*, 2003). Each panel shows modelled mean current vectors and mean salinity distributions over the simulation period April - July 1997. The left-hand panel shows near-surface (layer 6 in the model) salinity and circulation, the right-hand panel shows near-bottom (layer 16) salinity and circulation. The colour scale for salinity ranges from 34.4 (dark blue) to 35.2 (brown) in increments of 0.1. Of note are the recirculation patterns apparent in the sea of the Hebrides and in the Little Minch.

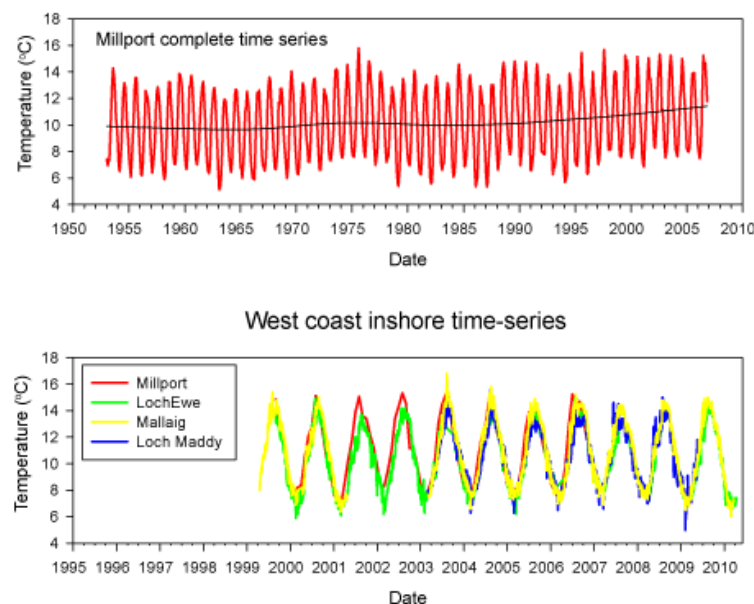


Figure 2.8 Patterns in the West of Scotland inshore temperature time series. Upper panel, the longest running time-series from Millport; Lower panel, more recent time series compared with records from Millport for the overlap period. All temperature time-series show clear seasonal signals and for the recent overlapping period there is good agreement between signals from the different sites. Bottom axis major tick marks indicate January of each year. The Millport data show evidence of a warming trend since the 1980s.

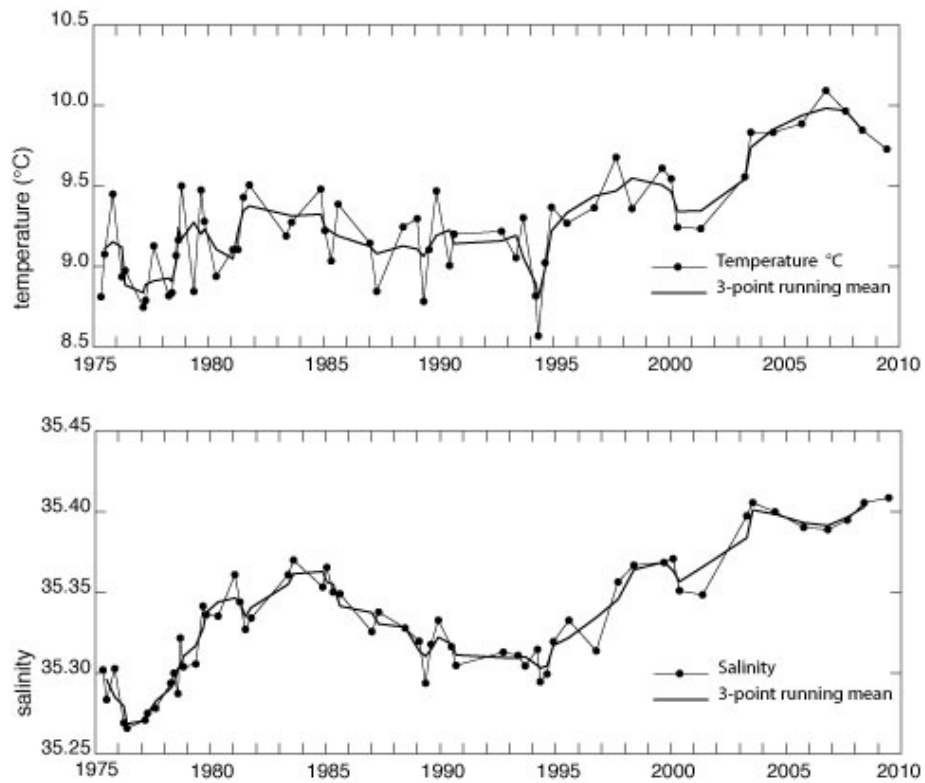


Figure 2.9 Recent trends in the surface waters (0-800 m depth) of the Rockall Trough as measured along the Ellett line. <http://www.noc.soton.ac.uk/obe/PROJECTS/EEL/index.php>.

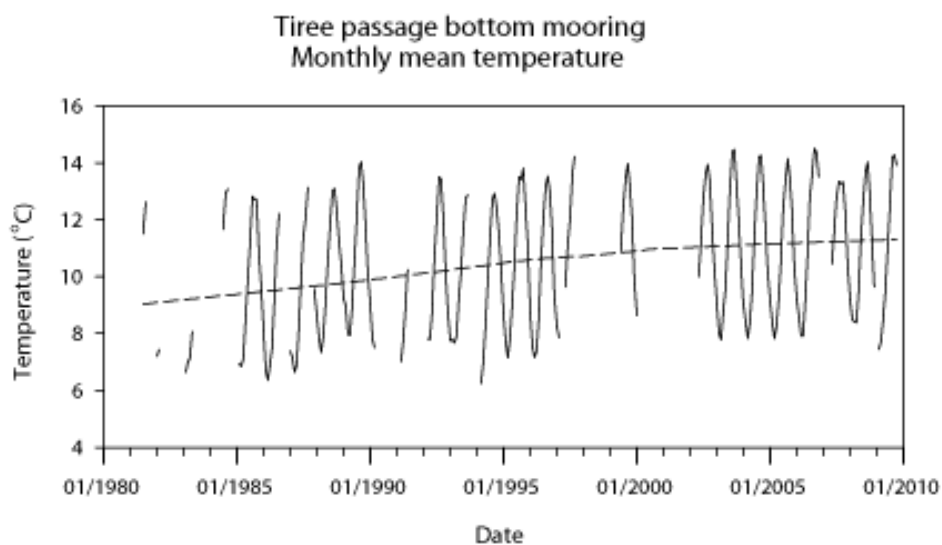


Figure 2.10 Tiree passage bottom mooring monthly averaged temperatures. The dashed line is a loess smooth. Plot includes data since 2004 which is not given in Inall *et al.* (2009).

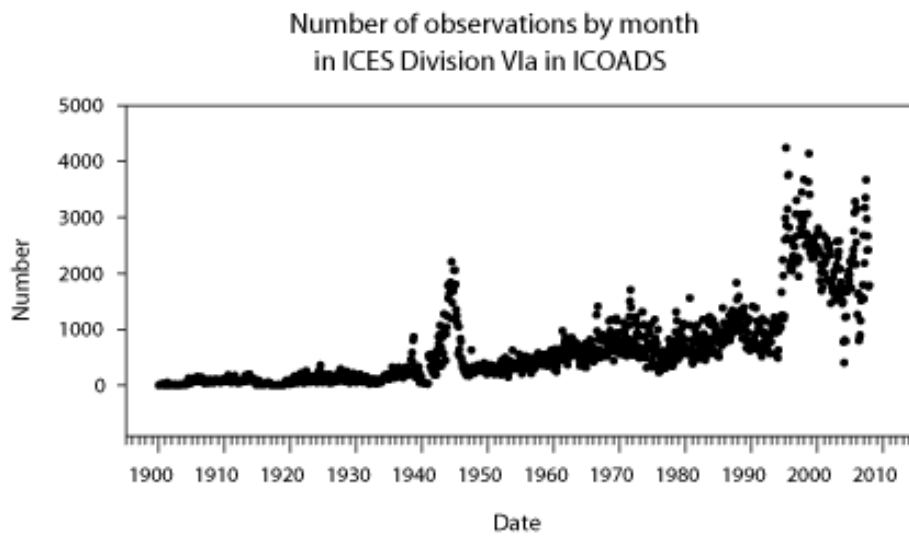


Figure 2.11 Number of sea temperature observations recorded in the ICOADS dataset for ICES Division VIa by month. The step-change which occurred in the late 1990s is related to the increase in satellite data.

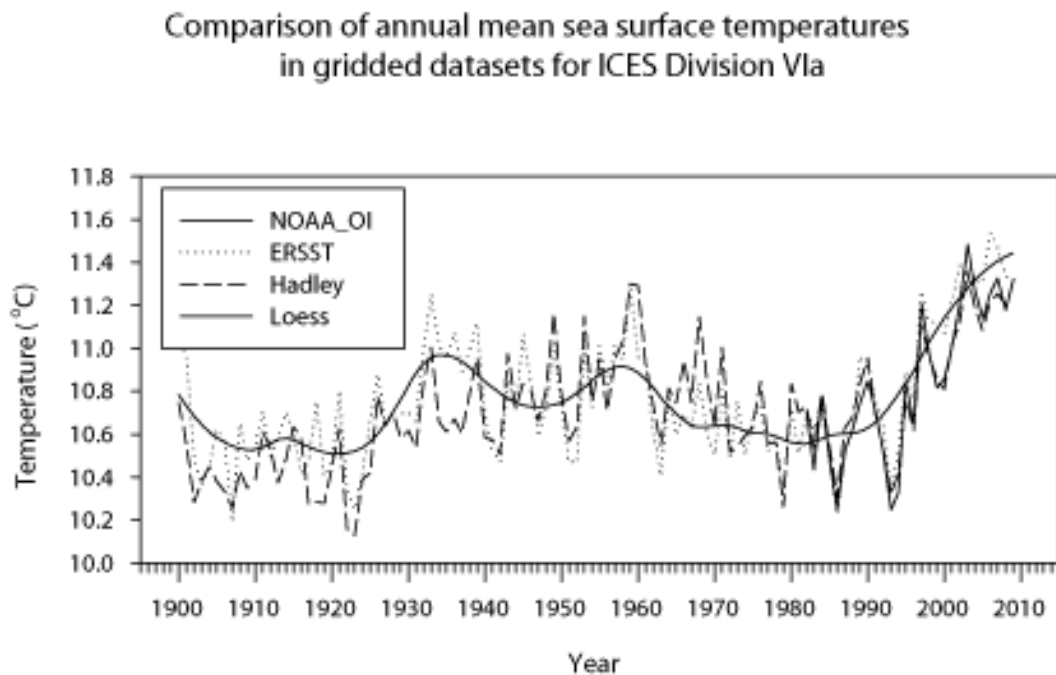


Figure 2.12 Trends in annual mean sea surface temperature from gridded datasets (smoothed line is Loess fit to the ERSST data).

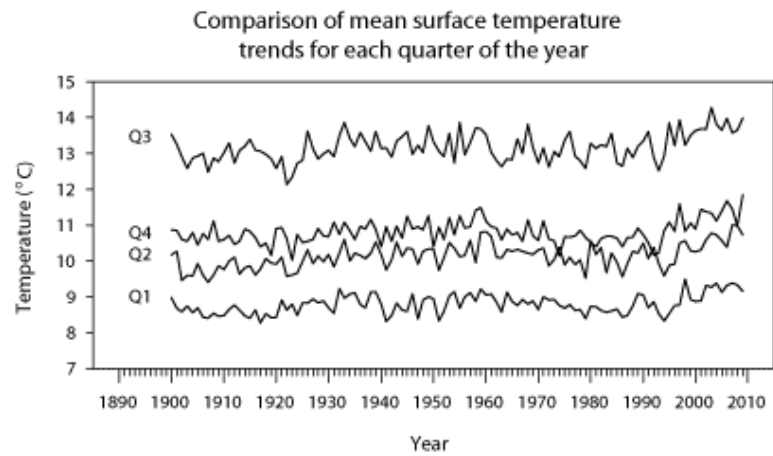


Figure 2.13 ICES Division VIa sea surface temperature data comparing quarterly averages, data is the average of ERSST and NOAA OI.

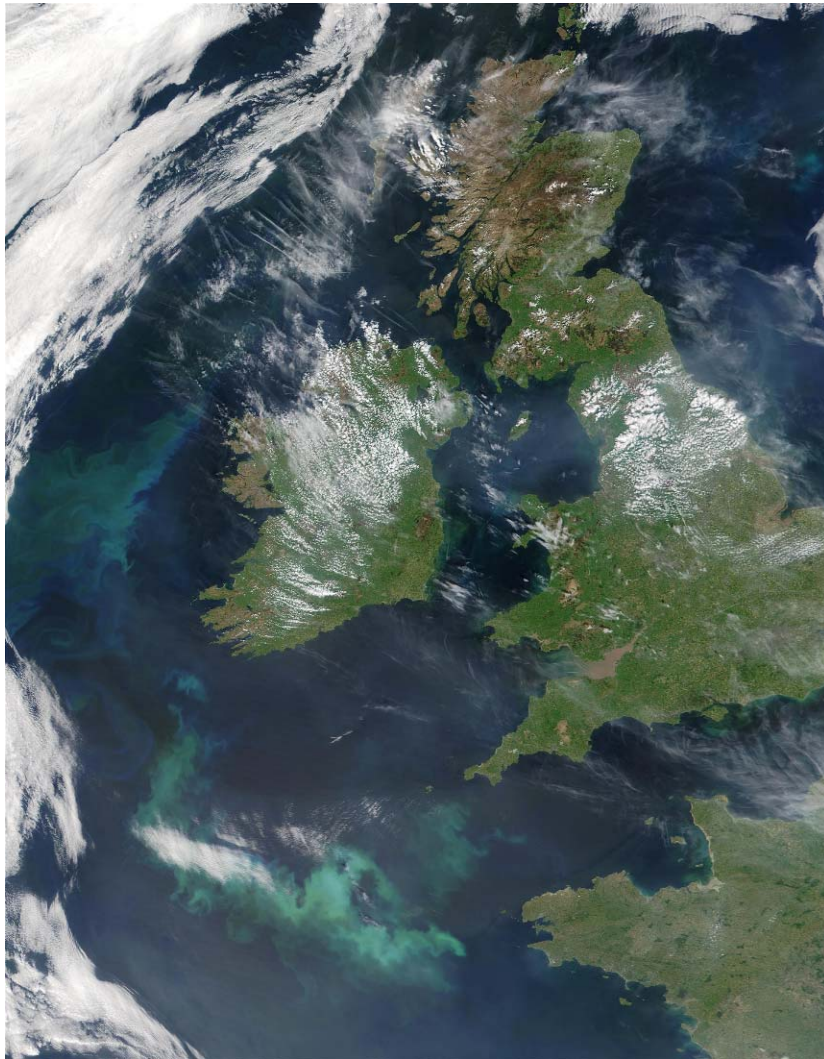


Figure 2.14 Image showing waters to the west of the UK from the Terra/MODIS sensor taken on 22 May 2001.

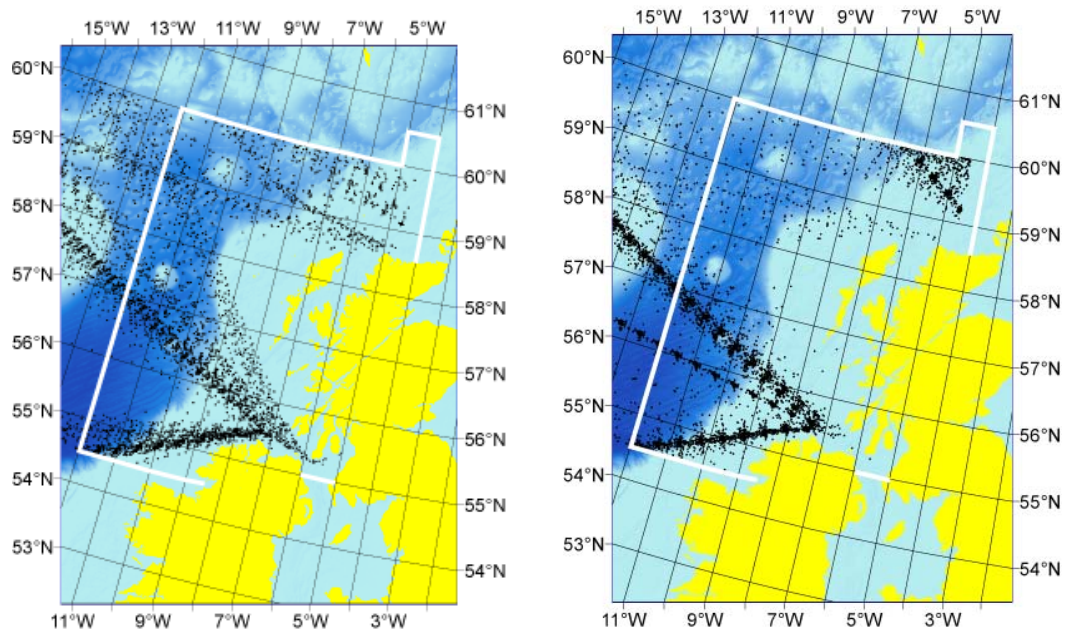


Figure 2.15 a (left) - Locations of CPR samples to the west of Scotland from 1940-1960. **b (right)** - from 1961-1980.

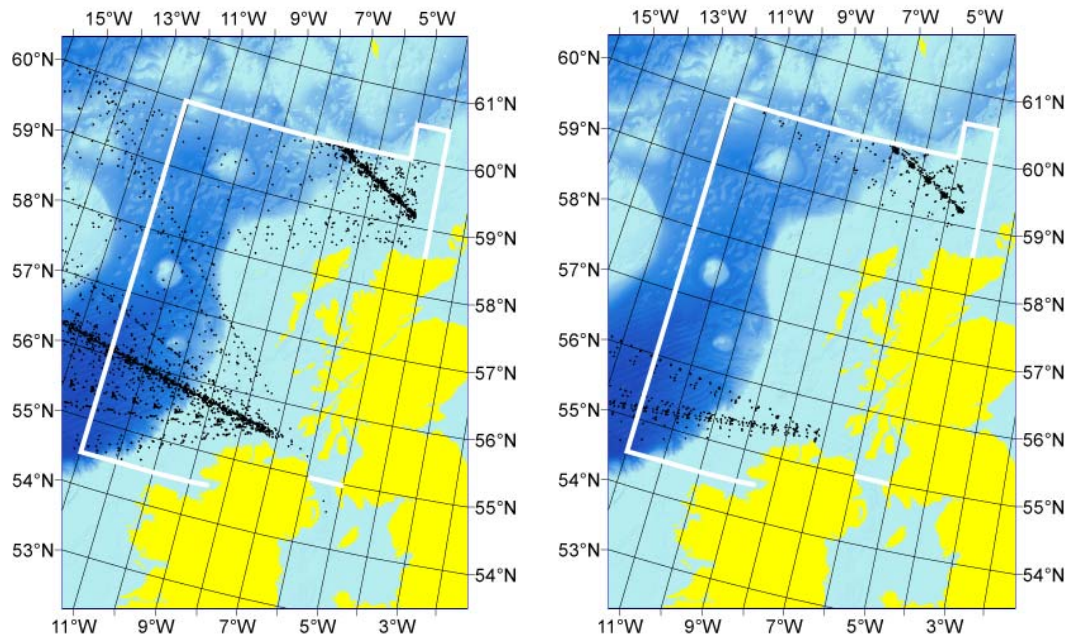


Figure 2.15 c (left) - Locations of CPR samples to the west of Scotland from 1981-2000. **d (right)** - from 2001-2008.

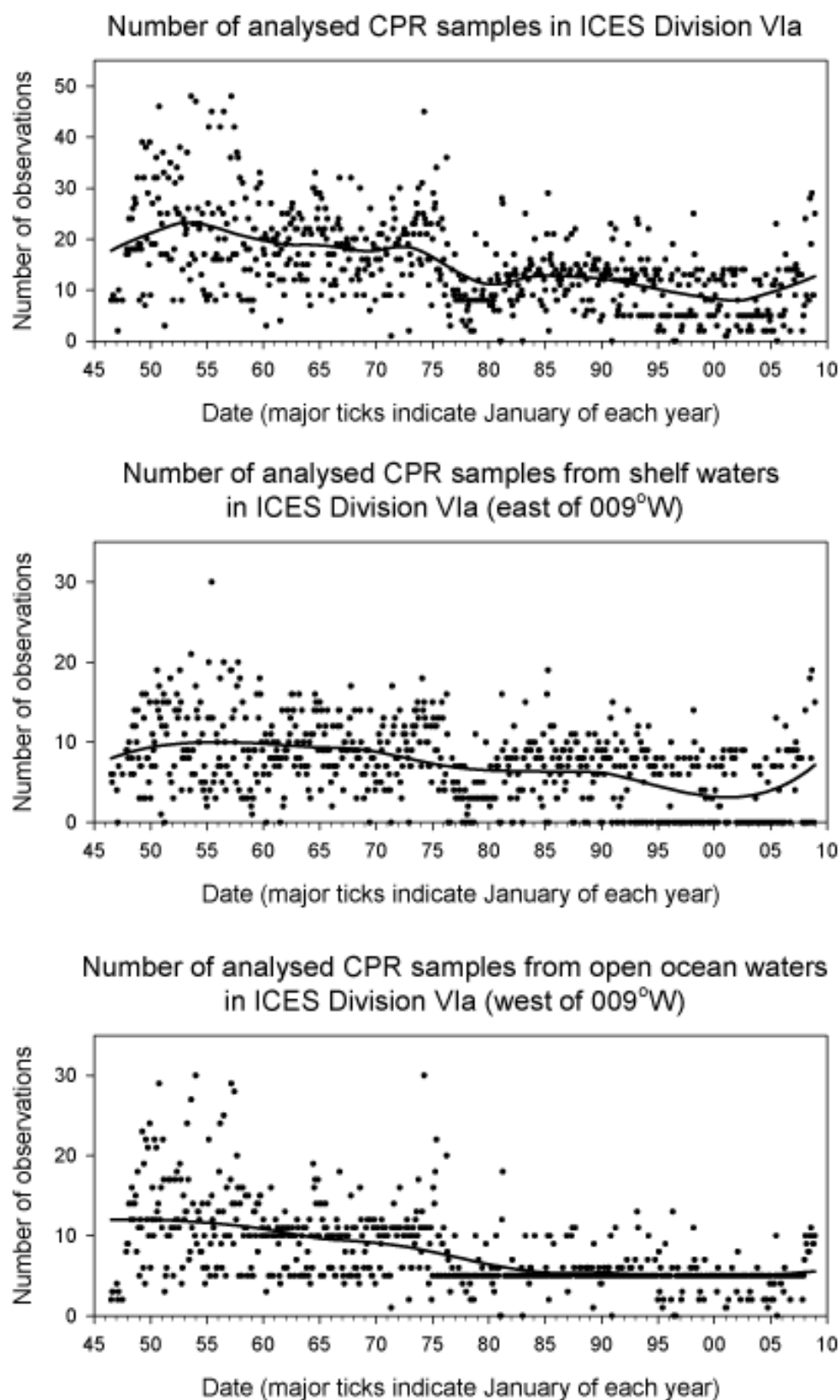


Figure 2.16 Numbers of CPR samples analysed each month from Division VIa (smoother is a Loess fit).

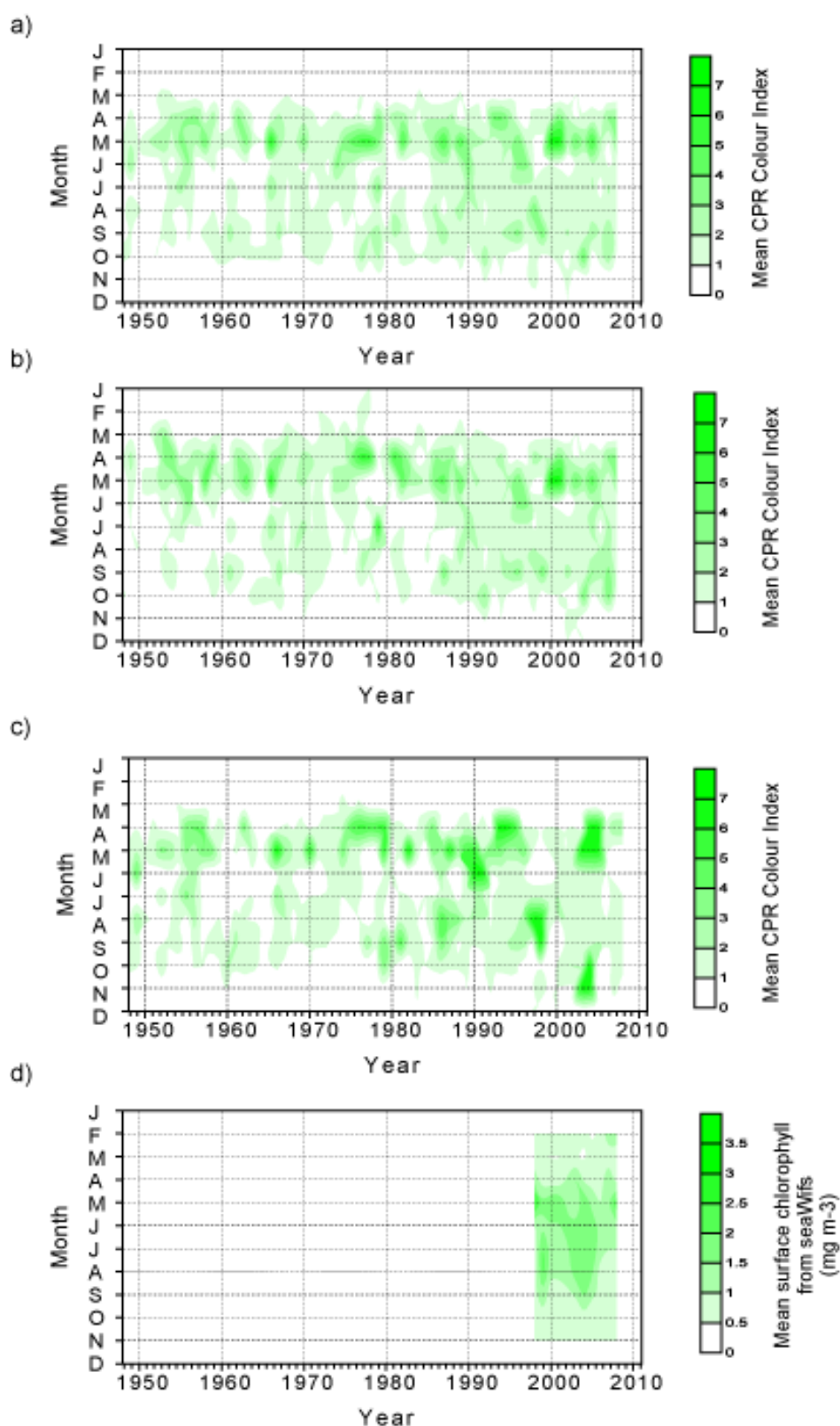


Figure 2.17 CPR Phytoplankton Colour index by month and year averaged for (a) all samples from ICES Division Via, (b) shelf samples (east of 009°W), (c) oceanic samples (west of 09°W), (d) available estimates of surface chlorophyll from remote sensing (SeaWiFS) for the whole region.

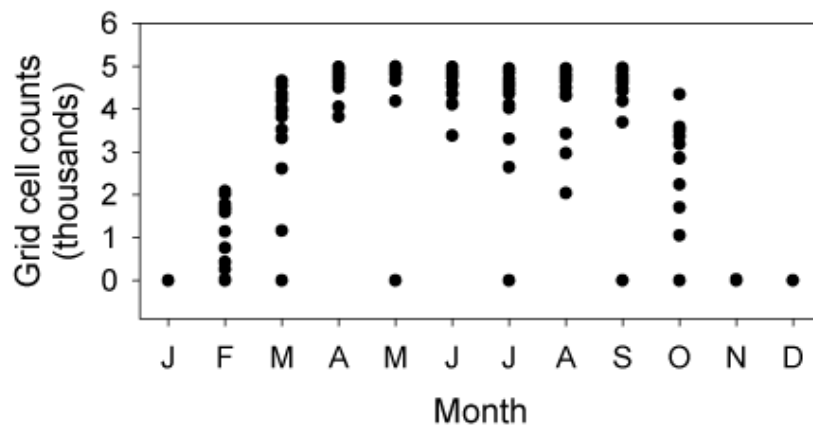


Figure 2.18 Grid cell counts by month for all available SeaWiFS data from ICES Division VIa. Dots show the grid cell count as the number of registered image cells used in deriving the area monthly mean chlorophyll values for each month by year.

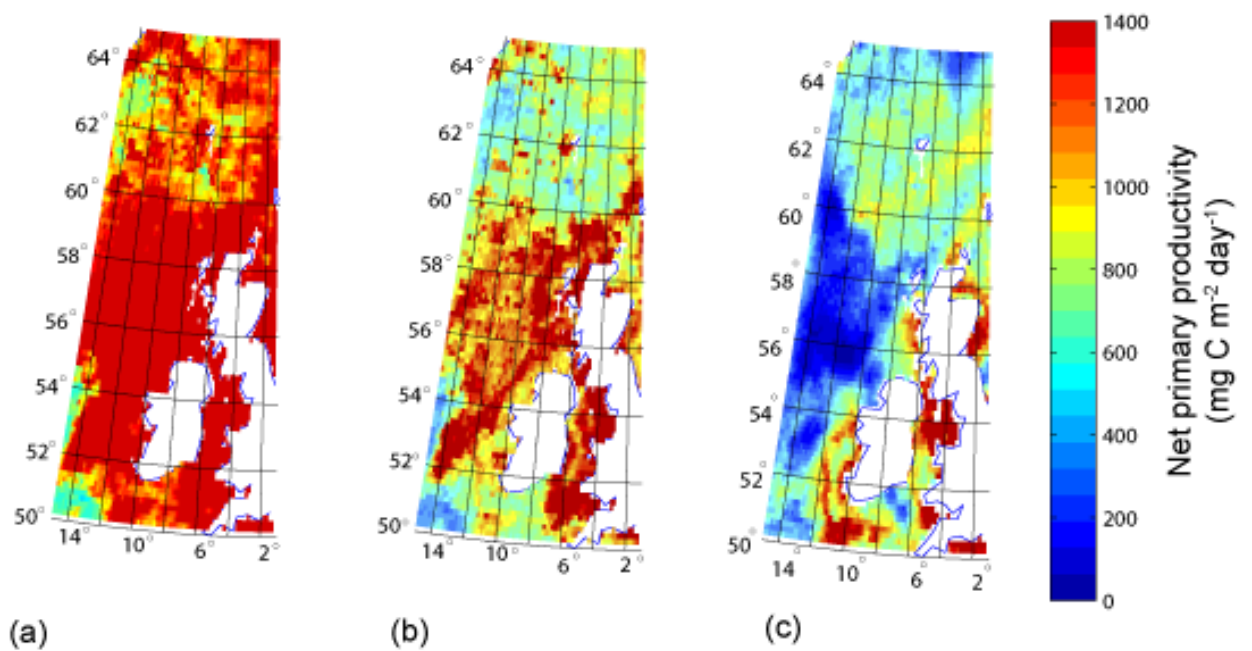


Figure 2.19 Comparisons of estimated net primary productivity for May 2004 using (a) the VGPM, (b) Eppley-VGPM, and (c) the CbPM. Satellite data used were SeaWiFS.

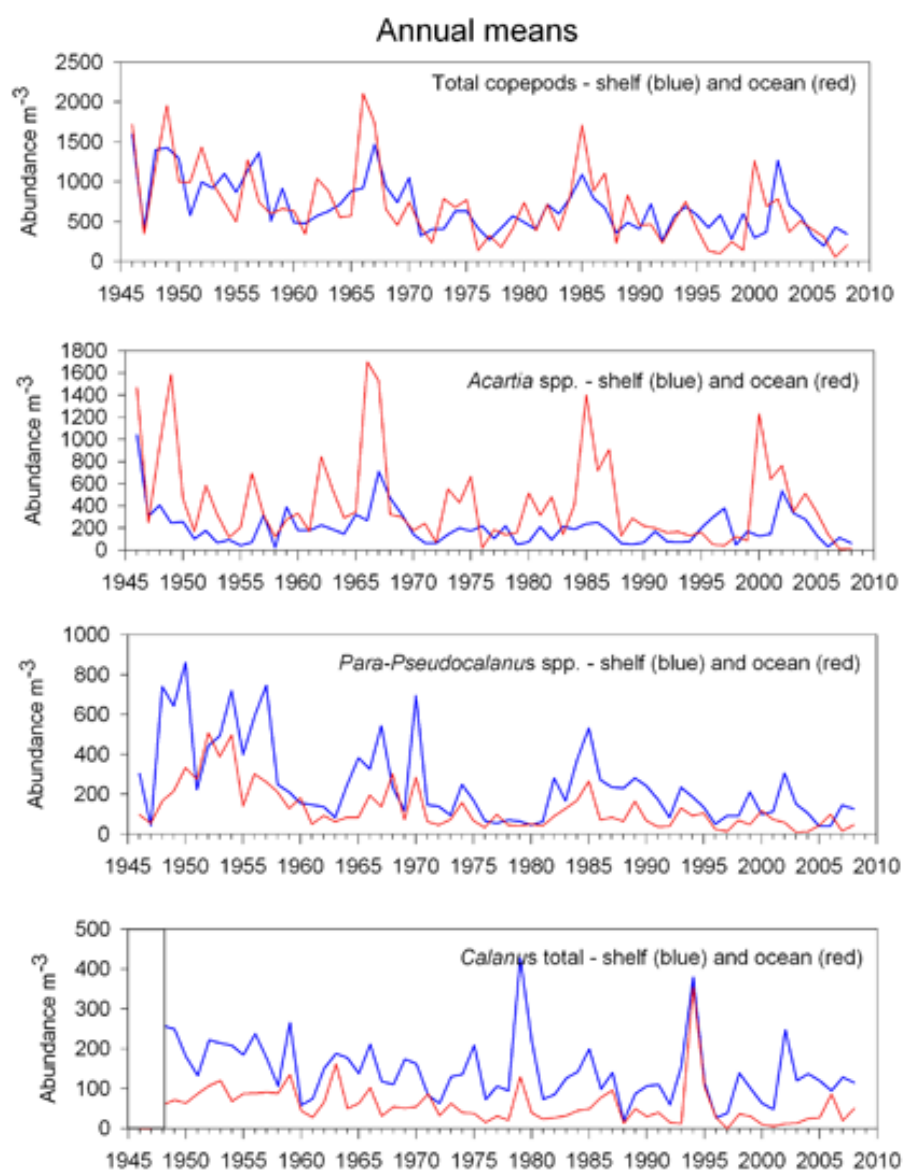


Figure 2.20a Long term trends in the annual abundance of the major copepod taxa for ICES Division VIIa.

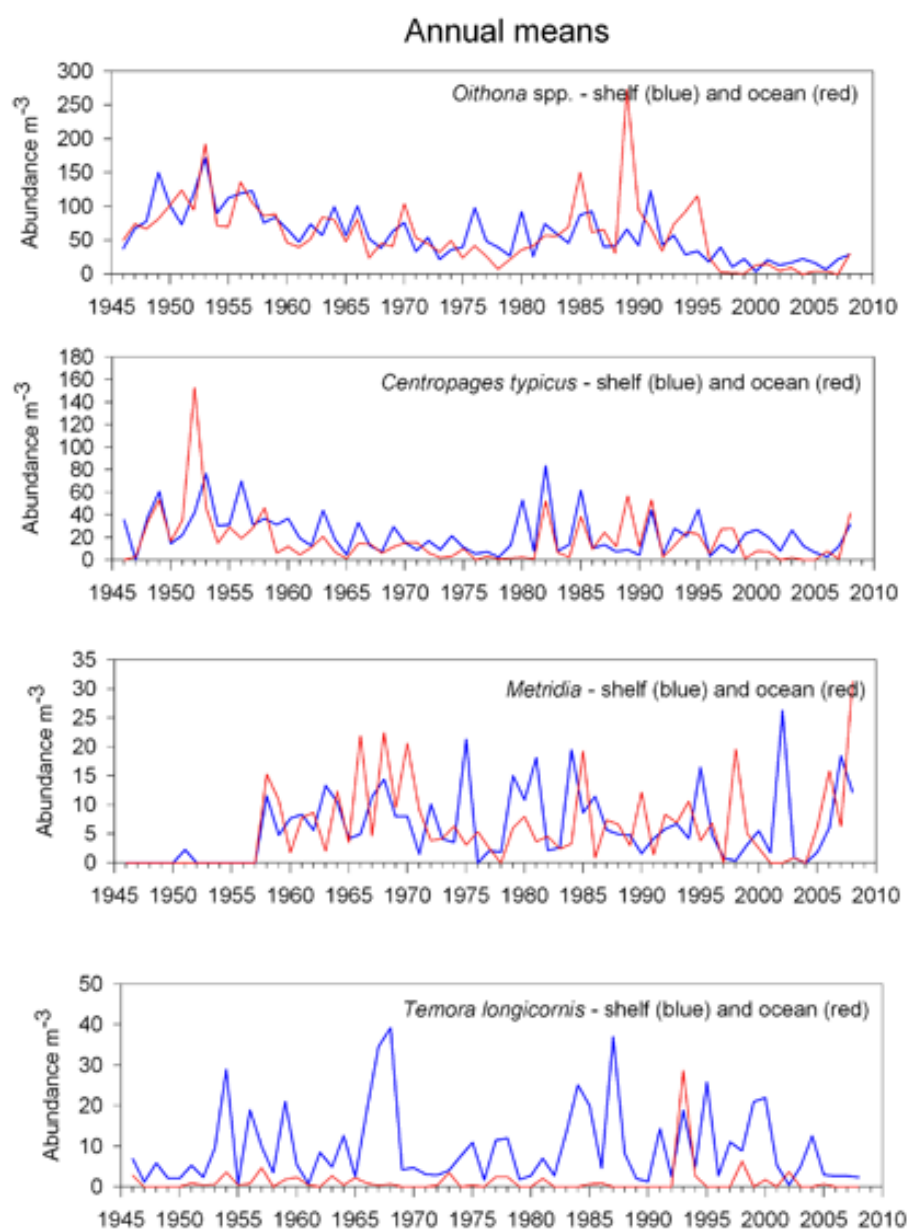


Figure 2.20b Long term trends in the annual abundance of the major copepod taxa for ICES Division Via.

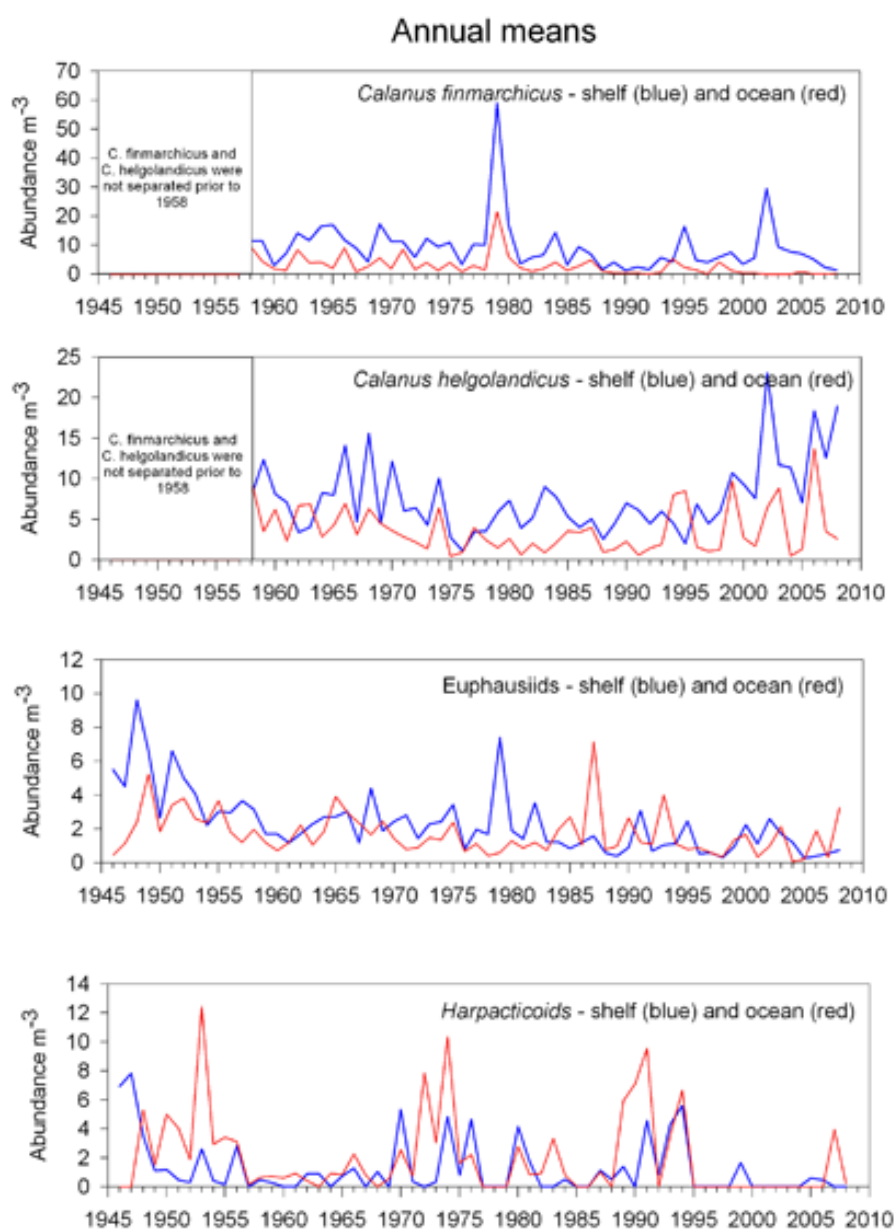


Figure 2.20c Long term trends in the annual abundance of the major copepod taxa for ICES Division VIIa.

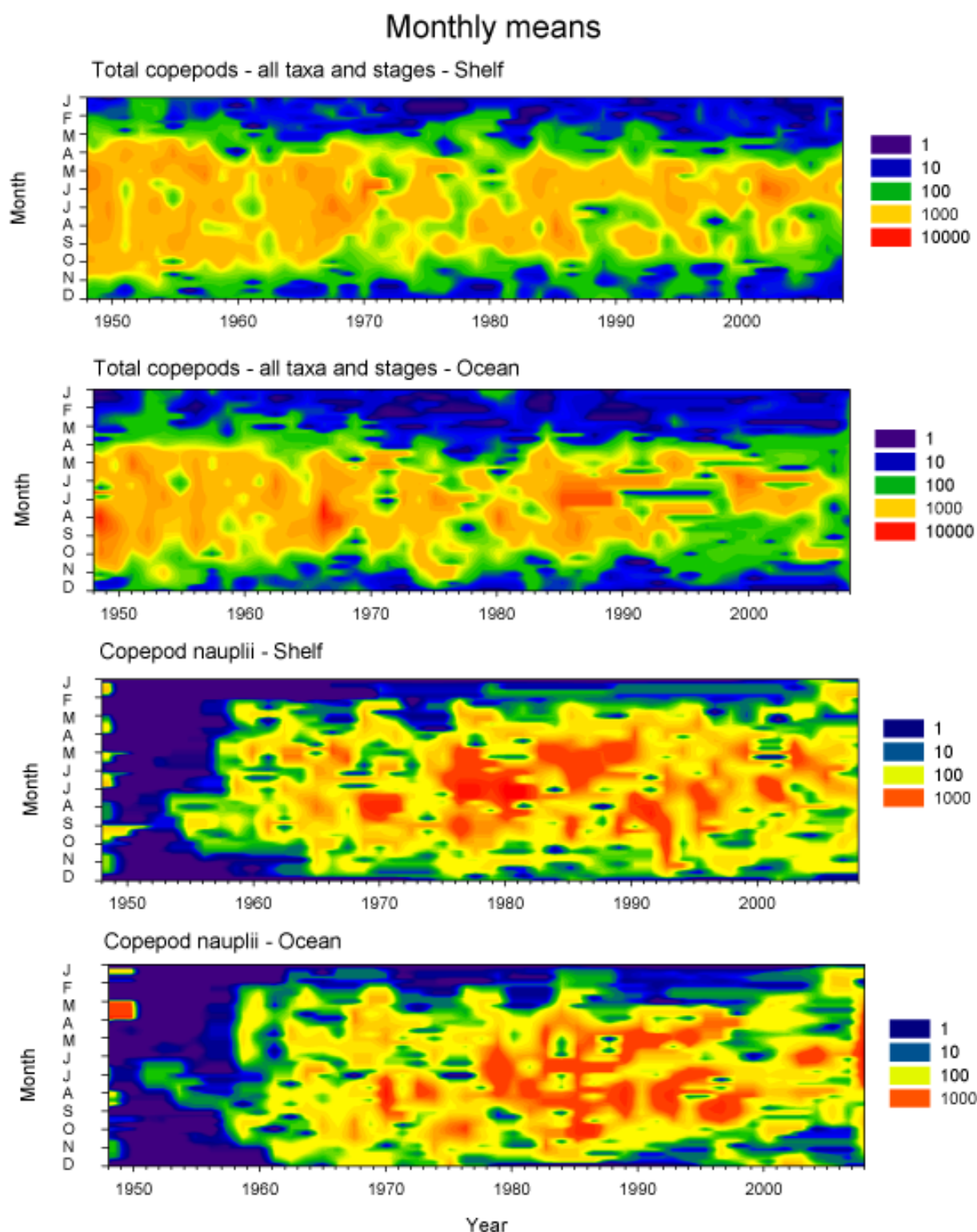


Figure 2.21 Annual and monthly patterns in abundance and timing of total copepods in ICES Division VIa (note scaling is logarithmic). Copepod nauplii do not appear to have been systematically recorded in the CPR prior to around 1960.

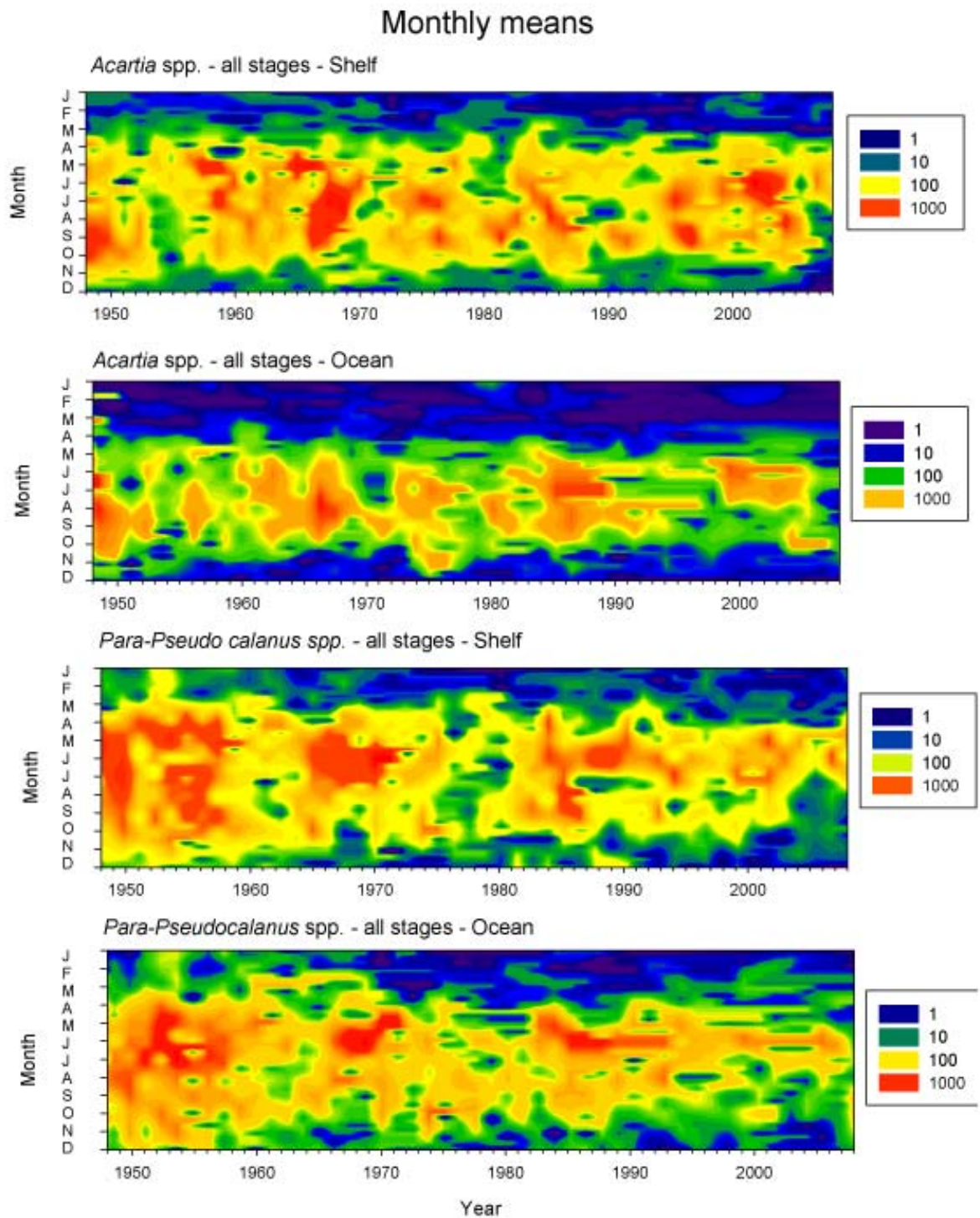


Figure 2.22: Annual and monthly patterns in abundance of *Acartia* spp. and *Para-Pseudocalanus* spp. copepods in ICES Division VIIa.

Monthly means - shelf waters

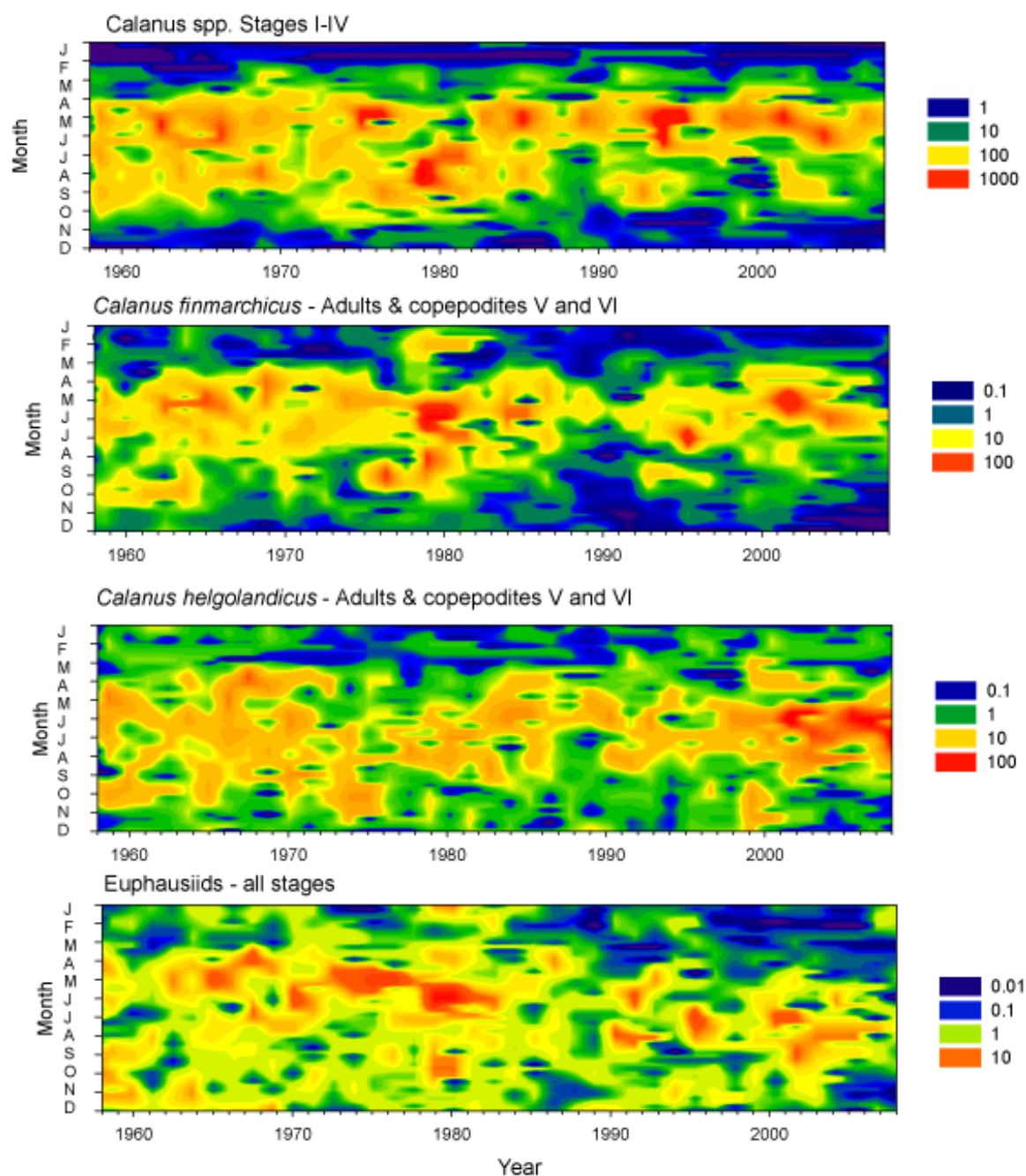


Figure 2.23 Annual and monthly patterns in the abundance of *Calanus* spp. copepods and euphausiids.

3 Ocean Climate: Evidence from Long-term Monitoring

S Hughes and W R Turrell

3.1 Introduction

This section briefly attempts to address the simple basic questions:

- is the west of Scotland warming faster than similar marine ecosystems at the same latitude, i.e. the North Sea?
- are winters warming faster than summers to the west of Scotland?

The first question is relevant to the current study as there is at least some anecdotal information which infers that some environmental difference exists, at least in the recent past, between west of Scotland waters and the North Sea. Stock recovery measures appear to have less effect to the west of Scotland than in the North Sea. This is a gross oversimplification, but it is from this line of thinking that a very basic comparison between temperature trends in the North Sea and west of Scotland waters has been undertaken.

The second question extends the first into a consideration of possible seasonal differences which would affect fish at different stages of their life history.

In-situ Time Series

The data used for this study comes from in-situ measurements performed as part of the Long Term Monitoring scheme run by Marine Scotland Science, as well as from long term monitoring performed at Millport by the University Marine Biological Station Millport and at the Tiree mooring by the Scottish Association of Marine Science.

Gridded Data Sets

The in-situ time-series are representative of shallow, coastal locations and it is therefore possible that they show a stronger variability and response to atmospheric forcing than they do to oceanic forcing. For comparison gridded datasets have been included in the analysis, covering 1° grid squares around Scotland and including extensive offshore areas.

3.2 Results

In-Situ Time Series 1982-2008

Figure 3.1 shows long-term temperature anomalies (i.e. seasonal cycle removed) plotted against year over the longest common time period available (1982-2008) for monitoring

stations representing west of Scotland coastal waters (Millport and Tiree) and the North Sea (Fair Isle and Peterhead – Figure 3.3).

While there is a good deal of variation in each of the time series, there are clear underlying warming trends in all four time series over this time period. Fitted linear regressions over the 26 year period indicate rates of warming of $0.55^{\circ}\text{C decade}^{-1}$ and $0.51^{\circ}\text{C decade}^{-1}$ at Millport and Tiree respectively, while in the North Sea the overall warming rate is 0.27° at both Fair Isle and Peterhead.

Hence we can say, at least for the time period 1982-2008, the coastal waters of Division VIa have warmed at about twice the rate of North Sea coastal waters.

In-Situ Time Series – Full Length Records

Table 3.1 shows the same warming rates for linear regressions fitted over the entire length of available records at each of the four long term monitoring sites; Millport, Tiree, Fair Isle and Peterhead. The greater warming of west of Scotland coastal waters is no longer evident, hence the difference between the west coast waters and the North Sea has only occurred in the last 25 years.

Table 3.1: Warming rates ($^{\circ}\text{C decade}^{-1}$) from linear regressions fitted to long term in-situ temperature time series.

Millport	Tiree	Fair Isle	Peterhead
1953-2009	1976-2007	1982-2008	1979-2009
0.25	0.25	0.51	0.34

In-situ Seasonal Warming Trends

Table 3.2 presents the fitted warming trends for the spring (Apr, May, Jun), summer (July, Aug, Sep), autumn (Oct, Nov, Dec) and winter (Jan, Feb, Mar) seasons for the common time period 1982-2007. Again for the long-term warming trend over this entire period the difference between the west coast and North Sea stations is evident. However, west of Scotland there seems little difference between seasonal warming rates, whereas in the North Sea autumn and winter temperatures appear to have warmed faster than spring and summer temperatures over this period.

Table 3.2: Warming rates ($^{\circ}\text{C decade}^{-1}$) from linear regressions fitted to long term in-situ temperature time series for the common time period 1982-2007.

	Millport	Tiree	Fair Isle	Peterhead
Long-term	0.60	0.55	0.30	0.27
Winter	0.18	0.51	0.55	0.43
Spring	0.25	0.63	0.11	-0.10
Summer	0.31	0.51	0.25	0.15
Autumn	0.19	0.42	0.48	0.48

Hence in terms of seasonal patterns, the waters west of Scotland show little difference between seasons in terms of long term warming rates, whereas in the North Sea winters have become warmer at a faster rate than summers, reducing the annual temperature range. These observations only apply to the last 25 years, i.e. 1982-2007.

Gridded Data Sets

Figure 3.3 shows the location of selected grid squares and Figure 3.4 shows how time series extracted from the nearest relevant grid box of the gridded datasets compare to the in-situ time series.

We can see clearly from these plots that the warming seen in the in-situ data at Millport is much stronger than seen in the gridded datasets but the overall variability seems similar. This implies that the gridded data sets do not fully resolve the detail of temperature variability in coastal waters. This should be expected as the gridded data sets undergo a degree of data interpolation and spatial averaging in their creation.

We can also see that the variability at Peterhead from the in-situ data is quite different to that from the gridded data, particularly in the period 1982-1983 and 1990-2000, the reason for these strong differences is not known and some further investigation will be necessary. The higher anomalies at the start of the analysis period are the main reason why the warming trend at Peterhead is lower than at other stations.

However, on the whole the gridded data sets agree with the in-situ observations and hence can be used to extend the analysis above to offshore waters.

Large Scale Spatial Patterns

Finally, Figure 3.5 uses the gridded data set OISST to examine how the pattern of warming may look over the entire west of Scotland / North Sea region by season. This figure suggests that rather than a west / east gradient in warming rate there exists a north / south gradient, i.e. southern areas of the region are warming at a faster rate than northern areas. While this overall pattern may be correct on a large spatial scale, it fails to explain the observed west / east difference described above from the in situ data in coastal waters. As

the in-situ data has been derived from carefully calibrated independent observation schemes there is no reason to doubt these.

3.3 Conclusions

In-situ data suggests that coastal waters west of Scotland are currently (i.e. 1982-2008) warming at a rate twice that of North Sea coastal waters ($0.6^{\circ}\text{C decade}^{-1}$ compared to $0.3^{\circ}\text{C decade}^{-1}$). North Sea warming may be greater in winter than in summer over this period, but there is no clear seasonal difference west of Scotland. Large scale gridded data sets fail to show an east / west difference so clearly, hence the effect may be constrained to coastal waters.

3.4 Figures

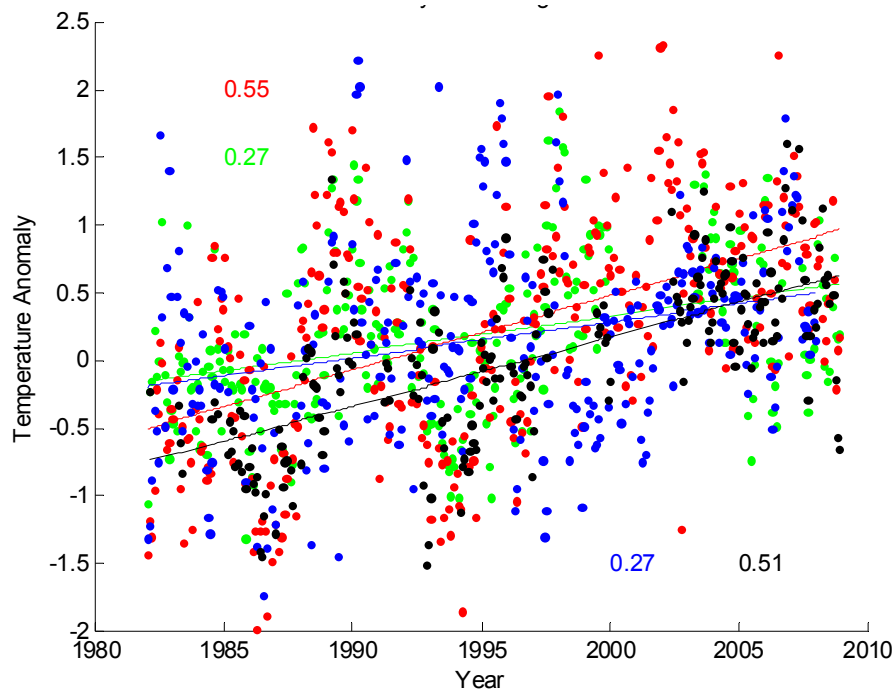


Figure 3.1: Long-term temperature anomalies plotted against year over common time period (1982-2008): Red: Millport, Green: Fair Isle, Blue: Peterhead, Black: Tiree

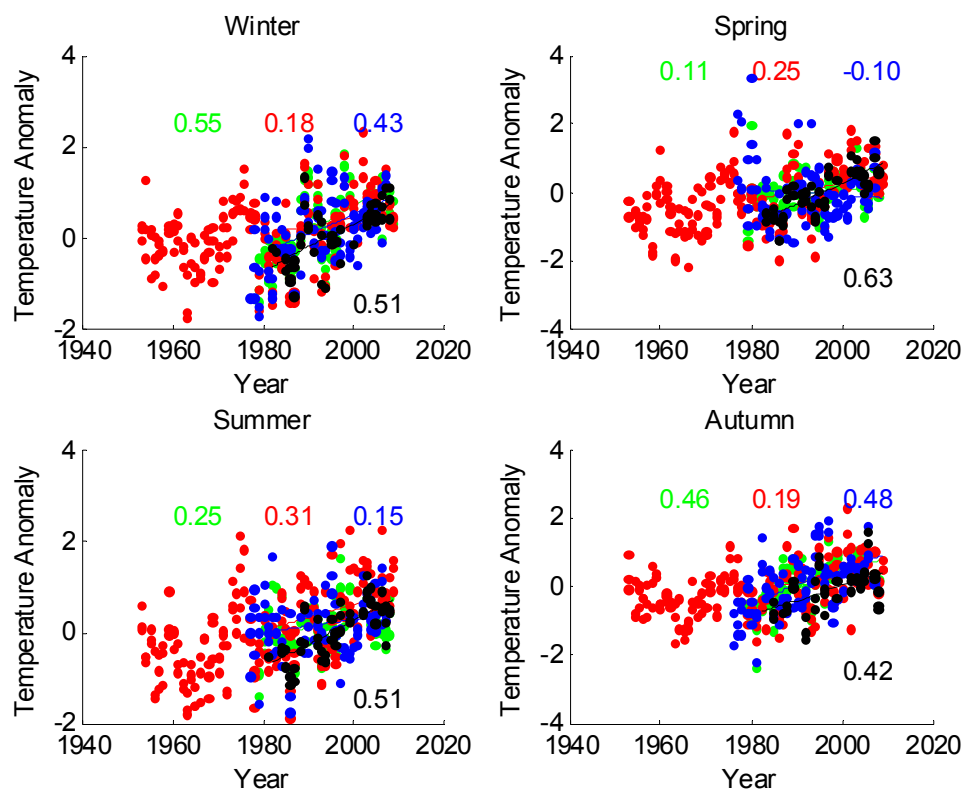


Figure 3.2: Long-term temperature anomalies plotted against year for each season: Red: Millport, Green: Fair Isle, Blue: Peterhead, Black: Tiree

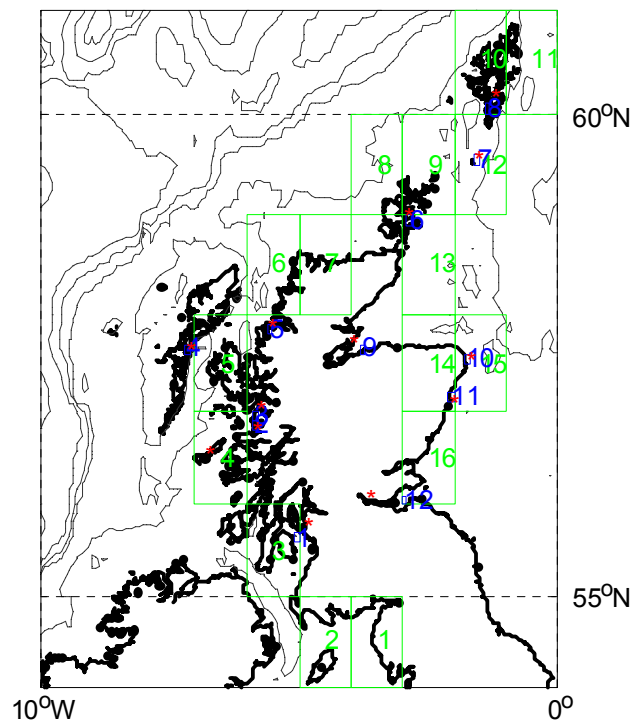


Figure 3.3: Map of Scottish coast. Red stars indicate inshore monitoring sites. Grid boxes were chosen to match each site as closely as possible, with OISST and HADISST represented as green boxes.

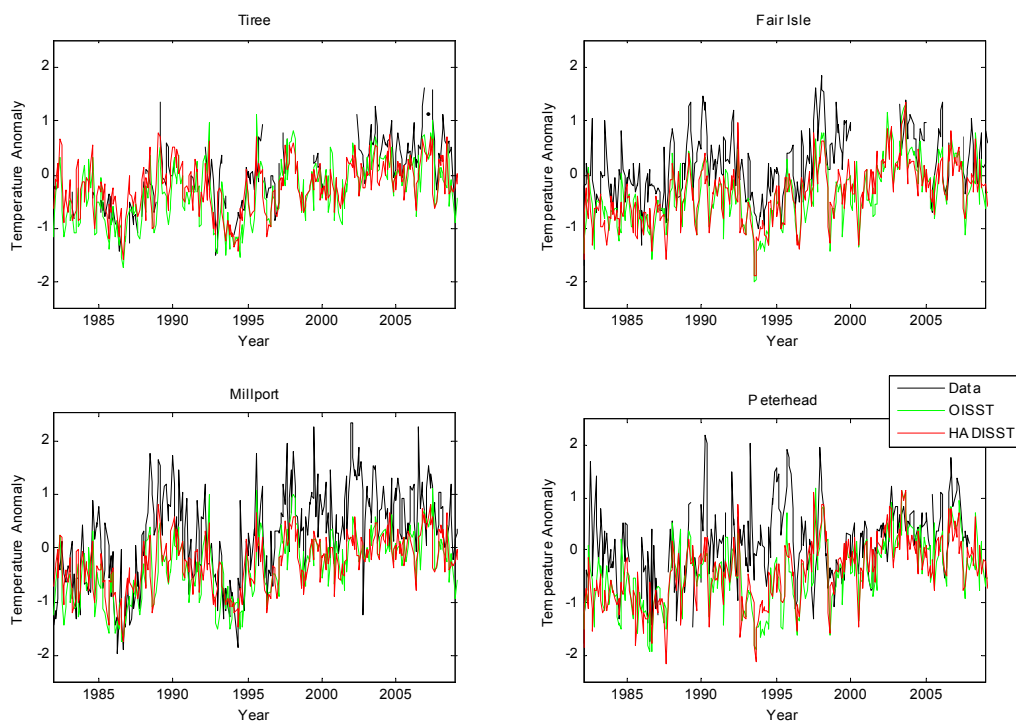


Figure 3.4: Temperature anomalies (relative to 1971-2000 mean) plotted against year for the period 1982-2009.

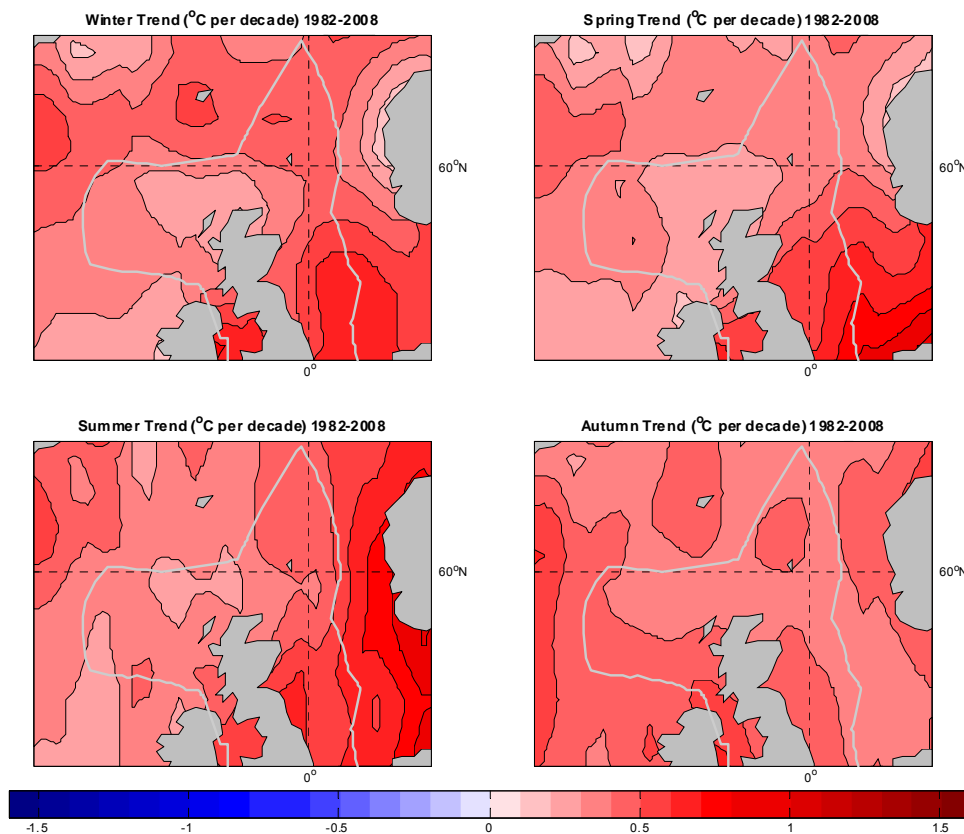


Figure 3.5: upper: Long term trend in sea surface temperature ($^{\circ}\text{C}$ per decade), calculated over the period 1982-2008. lower panel: Seasonal trend in sea surface temperature ($^{\circ}\text{C}$ per decade), calculated over the period 1982-2008. Winter = Jan., Feb., Mar., Summer = Jul., Aug., Sep. The gridded dataset is created from a combination of satellite and in-situ observations optimally interpolated to provide global coverage. OISST.v2 data provided by the NOAA/OAR/ESRL PSD, Boulder, Colorado, USA, from their web site at <http://www.cdc.noaa.gov>.

4 Remote Sensing – What Can it Tell Us ?

D McKee

4.1 Introduction

This section is intended to provide an introduction to Ocean Colour Remote Sensing capabilities and limitations, data availability for the Scottish west coast area and preliminary analysis of standard chlorophyll data for the duration of the currently available time series (approximately 12 years). A review of *in situ* optical studies and applications of ocean colour data in west coast waters is also provided. The aim of the report is to provide an easily accessible, informed view of the potential for ocean colour remote sensing to contribute to an improved understanding of the state of Scottish west coast waters.

4.2 Overview of Ocean Colour Remote Sensing (OCRS)

Ocean colour remote sensing is the study of materials suspended or dissolved in surface waters of the ocean through their impact on sunlight reflected back into space. Satellite-borne ocean colour sensors image the ocean from space at a number of wavebands and this data is then passed through algorithms and models to derive parameters such as chlorophyll concentration or primary productivity. The key strength of the method is the ability to map huge areas of the ocean surface on a regular basis (integrated systems give daily global coverage) and it provides synoptic views of upper ocean biogeochemical processes that are simply unattainable by any other method.

Ocean colour is determined by the concentration of optically active materials suspended and dissolved in seawater. In open ocean waters these are usually assumed to be water itself, phytoplankton and phytoplankton-derived materials (Case 1). The additional presence of coloured dissolved organic materials (CDOM) and suspended mineral particles increases the optical complexity of coastal waters (Case 2). Ocean colour sensors measure reflected sunlight at the top of the atmosphere and a number of corrections have to be applied to retrieve the quantity of light that was reflected from within the water column. By measuring this signal at a number of wavelengths it is possible to establish relationships with concentrations of materials in the water column as well as deriving other optical parameters such as diffuse attenuation coefficients.

Ocean colour can be used to:

- observe algal bloom formation and estimate phytoplankton biomass
- monitor sediment resuspension and transport
- assess impact of river outputs on coastal waters
- investigate links between physical and biological processes
- determine underwater light climate for photosynthesis and visibility

Satellite ocean colour measurements are affected by a number of limiting factors:

- visible/near infrared radiometers cannot see through clouds
- atmospheric correction is essential, but performs variably
- proximity to land or clouds affects sensor performance
- standard algorithms are generally optimised for the open ocean and often perform poorly in coastal waters.

4.3 Satellite-borne Ocean Colour Sensors (Past, Present and Future)

The International Ocean Colour Coordinating Group (IOCCG) maintains a comprehensive list of current, historical and scheduled satellite ocean colour sensors on its website (http://www.ioccg.org/sensors_ioccg.html). At the time of writing there were 11 historical sensors that had ceased operation, 11 sensors currently in operation and 7 sensors currently scheduled to be launched between 2010 and 2015.

Before 2009, all of the current and historic ocean colour sensors were mounted on polar orbiting satellite platforms and had multispectral wavelength resolution (between 4 and 36 bands). Spatial resolution varied between 250 and 6000 m with swath width varying between 200 and 2800 km. Of 11 historic sensors, 7 operated for less than two years. The experimental CZCS sensor, which flew on board the NIMBUS-7 satellite, operated between 24/10/78 and 22/6/86. There was then a significant gap of almost 10 years before the next operational ocean colour sensor (MOS, DLR Germany) was launched in 1996. Full global coverage became available with the launch of SeaWiFS (NASA, USA) in 1997 and has subsequently been bolstered by data from MODIS-Aqua, MODIS-Terra (both NASA) and MERIS (ESA). Complete global coverage is obtained every 2-3 days with these systems, though repeat cycles are more frequent for higher latitudes due to the polar orbits of the satellite platforms. Additional sensors have also been launched in this period by space agencies in China, India, France and Argentina.

The first hyperspectral (>100 wavebands) ocean colour sensor, HICO, was launched in 2009 as an instrument on board the International Space Station (ISS). This instrument combines high spectral resolution (124 bands between 380-1000 nm) with high spatial resolution (100 m), though it is limited to 50km swath and is only sufficiently sensitive for imaging coastal scenes. Unfortunately the ISS orbits between ~ 51.6°N and 51.6°S, hence the northern limit is too far south to be useful for Scottish coastal waters (approx. > 55° N). A polar orbiting hyperspectral sensor (HSI, DLR Germany) is scheduled to be launched in 2012 and will combine 200 bands between 420 - 2450 nm with 30 m spatial resolution over a 30 km swath.

The continuous global time series of ocean colour measurements currently provided by the SeaWiFS, MODIS, MERIS group of sensors is of increasing importance for monitoring long term changes in ocean conditions and associated climatic variability. This time series is expected to be extended with the launch of at least two OLCI sensors on board Sentinel

satellites operated by ESA/EUMETSAT, and two VIIRS sensors to be launched on NPP and NPOESS operated by NOAA/NASA. The nature of the operators of these satellite systems reflects a move from experimental sensors developed by space agencies towards operational sensors for monitoring and forecasting agencies.

A further key development is the launch of the first ocean colour sensor in a geostationary orbit (GOCI, KARI South Korea). Geostationary orbit offers the potential to make high temporal frequency images, enabling study of new processes on tidal and diurnal timescales. Despite the very high altitude required for geostationary orbit, the South Korean satellite is planned to have a 500m spatial resolution over a 2500 km swath. Geostationary sensors of this nature could be of great interest for countries at low to mid-latitudes. However the geostationary orbit is directly above the equator and the view angle probably limits useful observations to $\sim 50^\circ$ latitude, too far south for Scottish waters. For mid-high latitudes, high temporal resolution satellite imagery could be achieved through development of constellations of satellites, but no plans for such a system have currently been announced by any of the major space agencies.

4.4 OCRS Products and Performance in Coastal Waters

Algorithms have been developed to derive a wide array of products from ocean colour radiometry. Here we will consider standard products that are fully supported by space agencies operating ocean colour systems and a couple of experimental products (Fluorescence Line Height and Primary Productivity) that may be of particular interest to studies of the West Coast marine ecosystem. Standard products are generally better documented and have usually undergone more extensive validation than experimental products. Products are presented according to the method of calculation as there are often shared performance characteristics.

4.4.1 Radiometric Products

The basic output of an ocean colour system is a measure of the spectral distribution of reflected sunlight leaving the ocean. Radiometric data are expressed in absolute physical units of either radiance (e.g. water leaving radiance or normalised water leaving radiance – e.g. $\text{W m}^{-2} \text{nm}^{-1} \text{sr}^{-1}$) or reflectance (e.g. remote sensing reflectance – sr^{-1} , or irradiance reflectance – no units).

Radiances are measured at a number of wavelengths and are used as input for algorithms for derived products (see below). Combinations of radiances can be blended to produce a quasi-true colour image that gives an impression of how the scene might appear to a visual observer. Single wavelength radiance images and quasi-true colour images are both potentially useful for identifying major features in a scene (e.g. Figure 4.1, MODIS image from 11 May 2009).

4.4.2 Blue-Green Reflectance Ratio Algorithms

One of the earliest observations in ocean colour research was the effect of different concentrations of phytoplankton on reflectance signals in the blue-green spectral range. Over the years a number of algorithms have been developed that use various combinations of blue-green reflectance ratios to estimate the phytoplankton chlorophyll concentration (units: mg m^{-3}), e.g. O'Reilly *et al.* 2000. Diffuse attenuation (units: m^{-1}) is a measure of how rapidly light levels drop as a function of depth in the water column. Simple empirical relationships have been found relating this parameter to ratios of blue-green reflectances e.g. Mueller 2000. In general these standard algorithms have been tuned for open ocean waters and perform less well in turbid coastal waters (see below). Figure 4.2 shows estimates of Chlorophyll and diffuse attenuation obtained from blue-green reflectance ratios from the MODIS image of 11 May 2009.

4.4.3 Neural Network Products

Radiative transfer modelling enables calculation of theoretical reflectance spectra for any combination of water constituents. With a sufficiently large set of simulations, it is possible to train a neural network to identify combinations of constituents that are consistent with measured reflectances (e.g. Doerffer and Schiller, 1997). The power of this technique is strongly dependent on the quality and scope of the training data set, but it has the merit of implicitly handling optically complex waters. Figure 4.3(a,b) shows MERIS chlorophyll data from 3 May 2007 processed using the MERIS Case 1 (blue-green reflectance ratio) algorithm and the MERIS neural network (note colour scales are identical). There are significant differences in retrieved chlorophyll concentrations between the two approaches. Figure 4.3(c,d) shows neural network estimates of non-absorbing suspended particle concentration (units: mg l^{-1}) and CDOM absorption at 440 nm (units: m^{-1}).

4.4.4 Photosynthetically Available Radiation (PAR)

The absorption of light in the visible spectrum by phytoplankton is an essential component of the photosynthetic process in the marine environment. Ocean colour systems provide an estimate of IPAR, Instantaneous Photosynthetically Available Radiation, which is the irradiance integrated between 400-700 nm (units: $\text{micromol photon s}^{-1}$). Figure 4.4 shows MODIS IPAR for 11 May 2009, with higher values in the south and lower levels towards the north.

4.4.5 Fluorescence Line Height (FLH)

Phytoplankton have long been known to exhibit a distinguishing fluorescence characteristic, with a fraction of photons absorbed at blue wavelengths being emitted at red wavelengths. Neville and Gower (1977) demonstrated that a peak centred on 685 nm observed in surface leaving reflectance spectra could be attributed to sun induced chlorophyll fluorescence (SICF – Figure 4.5). Since then numerous studies have examined the relationship between

SICF and chlorophyll concentration, and attempts have also been made to retrieve phytoplankton physiological parameters from this signal. More recently it has been shown that the SICF signal is influenced by absorption and scattering by other materials found in seawater, with the result that SICF is more difficult to measure and interpret in optically complex coastal waters (McKee *et al.* 2007b). Figure 4.6 shows MODIS FLH and Chlorophyll for 11 May 2009.

4.4.6 Sea Surface Temperature (SST)

The temperature of the ocean plays a significant role in both physical and biological processes and is a fundamental parameter for many oceanographic studies. The temperature of the sea surface can be measured from space with radiometers operating in the mid- to far-infrared ($\sim 4 - 12 \mu\text{m}$) where black body radiation from the ocean is greatest and least affected by atmospheric absorption. SST measurements are made on only the top $10 \mu\text{m}$ of the water column and measurements are restricted to cloud-free zones. The radiometric signals used to generate SST data are potentially affected by atmospheric effects such as water vapour absorption, trace gas absorption and episodic variations in aerosol absorption, e.g. volcanic eruptions and desert dust events. Despite this, SST measurements are generally accurate to better than 1K, even in coastal waters. Figure 4.7 shows MODIS SST for 11 May 2009.

4.4.7 Primary Productivity

Plants use energy from sunlight to drive the photosynthesis process that enables them to produce organic material. The total amount of carbon that plants produce is known as gross primary production. Generally we are interested in net primary production, which is the gross primary production minus the amount of carbon needed for respiration. Marine phytoplankton are responsible for approximately half of the global (land and ocean) primary production and it is this process which supports virtually all life in the ocean.

It is not possible to directly measure primary productivity from space. Instead, models are used relating primary production to chlorophyll concentration, photosynthetically available radiation, water temperature and day length. As such, there are two important and distinct types of uncertainties associated with these estimates of primary productivity. The first is uncertainty associated with the choice of model. Various models exist with varying degrees of sophistication and different assumptions about phytoplankton physiology and interactions with water column physics. The second major source of uncertainty is associated with the quality of input data to the models. For example, each model requires an estimate of Chlorophyll concentration which, as discussed in this chapter, can be derived from satellite observations with variable success, and is generally poor in turbid coastal waters. Results from different Primary Productivity models for the Scottish west coast exhibit strong variability and these products require significant further analysis and validation before incorporation into modelling efforts.

4.5 Review of Studies in Scottish West Coast Waters

Relatively few optical studies have been carried out in Scottish coastal waters, with the distribution being spread between three main areas: sea-lochs such as Lochs Linnhe, Etive and Creran near Dunstaffnage (location of the Scottish Association for Marine Science), the Clyde Sea including associated sea-lochs and the Clyde estuary, and the Irish Sea (technically outside the scope of the current study, but included for information purposes).

The low level of activity reflects (a) limited availability of satellite coverage due to atmospheric conditions, primarily cloud cover, (b) limited numbers of active researchers in the field in Scotland and the UK, and (c) the relatively recent availability of high quality *in situ* sensors capable of providing data required for satellite validation studies. The location of these studies partly reflects ease of access from ports with research vessels (SAMS near Oban and the University of Wales Bangor), but also reflects the fact that these areas present wide ranges of optical diversity within very short geographical distances.

4.5.1 *In Situ Optical Studies*

Waters off the west coast of Scotland are generally optically complex, with optical signals being influenced to varying degrees by phytoplankton, minerals and coloured dissolved organic materials (CDOM). Physical properties of the water column also influence optical signals through the formation of stratified layers and well mixed regions.

Understanding links between optically active constituents (OACs) and optical signals has formed the bulk of activity in this area. Early studies looked at relationships between radiometric signals and concentrations of OACs (Bowers *et al.*, 2000; Bowers *et al.*, 2001; Tett *et al.*, 2001) and determined simple relationships for local areas. Some of this work was carried out with experimental prototype instruments with limited capabilities (e.g. spectral resolution) and the results must be considered with caution in light of subsequent advances in instrumentation. Nonetheless, these studies provided valuable first evidence of the potential for optical technology to reveal new insights into biogeophysical process in these coastal waters.

Another prototype radiometer system was developed at about the same time which was able to provide the first hyperspectral radiometric measurements. This surface floating instrument provided high spectral resolution data that facilitated initial studies into solar induced chlorophyll fluorescence and first attempts at modelling underwater and water leaving light fields using radiative transfer simulation software (Cunningham *et al.*, 2001; 2002). Further studies examined relationships between inherent and apparent optical properties in these waters and found results that were largely consistent with theoretical considerations (Bowers *et al.*, 1996; McKee *et al.*, 2003a, 2003b). Caution is required in the interpretation of IOP data from highly scattering waters however as it has been shown that there are still potentially quite significant errors in the scattering correction of state of the art *in situ* instruments (McKee *et al.*, 2005, 2008).

A number of consistent features became apparent from these early studies. In the sea-lochs, the combination of low salinity (density) and high CDOM in freshwater run-off meant that surface brackish layers were optically dominated by the CDOM except in times of very strong algal blooms and the effect of mineral particles was generally negligible (McKee *et al.*, 1999, 2002). Deeper water below the pycnocline could be affected by nonalgal particles, though this might have little effect on remote sensing signals if it occurred below the first optical depth.

More open waters of the Clyde and Irish Seas were more often strongly influenced by suspended mineral particles (Binding *et al.*, 2005). Devlin *et al.* (2008, 2009) have recently suggested that the diffuse attenuation of light in coastal waters such as the Clyde and Irish seas is heavily dominated by the influence of suspended particles. McKee *et al.* (2006) observed two optically distinct water types in the Irish Sea based on analysis of quantitative IOPs and OACs. It was subsequently demonstrated that these optical water types had corresponding differences in ocean colour signals that led to the breakdown in performance of standard algorithms that could be largely rectified by application of novel water-type specific versions (McKee *et al.*, 2007a).

These studies indicate that the degree of optical complexity found in Scottish coastal waters limits the performance of standard remote sensing estimates of OACs and necessitates development of tuned or novel algorithms for these regions.

4.5.2 *Studies using Satellite Ocean Colour Data*

The literature contains very few studies of ocean colour remote sensing data from Scottish west coast waters. One example is an attempt to estimate sea surface salinity from a blue-red ratio algorithm in the Clyde Sea (Binding and Bowers, 2003) where the restricted nature of the environment leads to a simple relationship between salinity and CDOM (McKee *et al.*, 1999). Although this algorithm may be restricted to this particular region, it does show the potential for extracting useful information about physical properties of surface waters from ocean colour data in coastal areas.

Primary Production

A key aim for the ocean colour remote sensing community is to be able to estimate phytoplankton primary production from space. Tilstone *et al.* (2005) generated maps of primary production in the Irish Sea by coupling remotely sensed concentrations of chlorophyll with integrated radiative transfer and spectrally resolved primary production models. Comparison with coincident *in situ* measurements of primary production showed variable quality of match up within a limited number of samples (~12). This study highlights the difficulty of accurately retrieving primary production from ocean colour data in optically complex waters, even using state of the art models. The limits stem from poor performance of chlorophyll and diffuse attenuation algorithms as well as uncertainties in the parameterisation of the photosynthesis and underwater light field models.

HABs

Harmful algal blooms (HABs) pose a serious threat to marine aquaculture activities and other social and commercial activities on the west coast of Scotland. A recent study examined the potential for using ocean colour data as an early warning indicator for HABs and found that there was significant merit in a multivariate classification scheme to discriminate harmful from harmless algae (Davidson *et al.* 2009). The proposed methodology correctly identified 83% of 674 samples containing *Karenia mikimotoi* in a training data set. However, as well as failing to identify the remaining 17%, the method also gave false positives for 3% of 2980 non-HAB samples. These results suggest that the method has strong merit if implemented as part of a broader sampling strategy, potentially as a means of directing effort to areas of greatest risk.

Fronts

Identification of fronts between water masses is a useful technique that has been applied successfully in Scottish waters using sea surface temperature data from the AVHRR satellite sensor. Sherwin *et al.* (2008) used the technique to identify the position of the front between North Atlantic Water and Modified North Atlantic Water in the Faroe – Shetland Channel. In this case remote sensing data assisted in the development of an improved understanding of the movement of water masses in an area that is critical for heat and salt transport from the Atlantic into Nordic seas.

The front identification method has also been used to locate the position of thermal fronts in the Clyde Sea and, together with satellite tracking of a tagged basking shark, provided new information that might explain the swimming behaviour of the shark at that time (Priede and Miller, 2009). Hedger *et al.* (2007) examined different methods of analysing high spatial resolution sea surface temperature data from the Airborne Thematic Mapper to provide synoptic views of estuarine spatial dynamics in Kirkcudbright Bay. All of these applications highlight the potential of remote sensing data to inform research into physical and biological oceanography in Scottish waters.

Seabed Morphology

High spatial resolution multispectral data from satellite sensors such as SPOT (~10m pixel resolution) and IKONOS (~4m pixel resolution) has potential for mapping shallow submerged habitats and other morphological features in localised areas. Malthus and Karpouzli (2003) analysed a high resolution IKONOS image of the Sound of Eriskay to identify underwater habitat features and found that the high spatial resolution improved classification accuracy, but the restricted spectral resolution was a limiting factor. These results are consistent with those of Gilvear *et al.* (2004) who examined the utility of airborne CASI hyperspectral imagery together with LIDAR images for assessing the geomorphology of rivers and estuaries.

Simulation of 4m IKONOS and 10m SPOT images from the Forth Estuary suggested significant loss of information for the lower spatial resolution data. Habitat mapping studies in coastal and estuarine areas will benefit in future from sensors with high spatial and spectral resolution mounted on airborne and satellite platforms. Frequency of return visits (allied to cost of image acquisition and weather constraints) is likely to remain a limiting factor in the application of these technologies however.

4.6 Survey of Data Availability from Current Sensors

4.6.1 Sensor Resolution and Repeat Visit Frequency

The International Ocean Colour Coordinating Group (IOCCG) list of past, current and scheduled satellite ocean colour missions can be found at:

http://www.ioccg.org/sensors_ioccg.html. Table 4.1 provides basic information for a selection of ocean colour sensors currently in orbit.

Table 4.1: Basic specifications for a selection of ocean colour sensors currently in orbit.

Sensor	Orbit	Agency	Resolution (km)	Swath (km)	Bands	Spectral Range (nm)	Launched
SeaWiFS	Polar	NASA	1.1	2806	8	402-885	08/1997
OCM	Polar	ISRO (India)	4	1420	8	402-885	05/1999
MODIS-Terra	Polar	NASA	1	2330	36	405-14385	12/1999
MERIS	Polar	ESA	1.2 / 0.3	1150	15	412-1050	03/2002
MODIS-Aqua	Polar	NASA	1	2330	36	405-14385	05/2002
POLDER-3	Polar	CNES (France)	6	2100	9	443-1020	12/2004
OCM-2	Polar	ISRO (India)	4	1420	8	400-900	09/2009
GOCI	Geo	KARI (Korea)	0.5	2500	8	400-865	06/2010

Many of the current generation of ocean colour sensors share broadly similar features such as sun-synchronous polar orbits, normal spatial resolution of ~1km, swath width between 1000-3000km, global coverage in 1-3 days, and 8-10 spectral bands in the visible – near infrared. With newer systems there is a trend towards additional availability of higher spatial resolution (~0.25km) data that will benefit studies in coastal waters. Other trends for newer systems include increased spectral resolution and broader wavelength coverage (e.g. MODIS sensors extend into the infrared to provide SST data).

The successful launch in June 2010 of the South Korean GOCI sensor into geostationary orbit represents a significant milestone in the development of OCRS. This sensor will provide hourly updates over a 2500 x 2500km area around the Korean peninsula. Although

this is an exciting development, it is unlikely that transfer of this technology will bring direct benefits to Scotland as the view angle for Scottish latitudes would be at or beyond the limits of acceptable spatial resolution.

4.6.2 *Data Download and Processing*

Ocean colour data is generally made available through web-based portals operated by the space agencies. NASA provides easy free access to MODIS data and registered users can access SeaWiFS data subject to limitations of the contract between NASA and the commercial operator of the spacecraft (GeoEye).

http://oceancolor.gsfc.nasa.gov/SeaWiFS/ANNOUNCEMENTS/getting_data.html

ESA generally requires registration for almost all products and research users usually have to specify a projected use of data before authorisation will be given for free access (Category 1 User).

<http://eopi.esa.int/esa/esa?cmd=aodetail&aoname=cat1>

Both space agencies offer online browse facilities to facilitate selection of data for download:

NASA Ocean Color Web - <http://oceancolor.gsfc.nasa.gov/> , and
ESA EOLi (Earth Observation Link) - <http://earth.esa.int/EOLi/EOLi.html>

After download, data can be processed using dedicated software packages that are made freely available to the community by the space agencies:

NASA SeaDAS - <http://oceancolor.gsfc.nasa.gov/seadas/>
ESA Beam - <http://www.brockmann-consult.de/cms/web/beam/>

Beam can be run on multiple operating systems (e.g. Linux, Windows, Mac) and SeaDAS can be run on either Linux, or Windows running a Linux emulator (provided as a complete package). A good specification computer with large data storage capacity is recommended for processing data from either system. All images in Section 1 of this report were processed by David McKee and Claire Neil at the University of Strathclyde on a HP xw4600 Workstation.

4.6.3 *Databases*

Downloading and processing individual satellite images is a reasonably time consuming activity requiring some degree of specialist knowledge. For many applications it is sufficient to be able to access data that has been pre-processed using standard settings. A number of websites provide access to processed OCS data, presented in formats that are designed to

facilitate associated research activities. A small selection of these data bases will be briefly introduced in this section.

The Giovanni database provided by the Goddard Earth Sciences Data and Information Services Center (GES DISC - <http://disc.sci.gsfc.nasa.gov/giovanni>) provides open access to processed data from SeaWiFS, MODIS Terra and Aqua as well as other EO satellite systems. A large number of standard and derived products are made available. Graphical tools are provided to select areas and periods of interest. A variety of visualisations can be selected including time series, maps and animations, with associated basic statistical analyses. Access to the database is free but due acknowledgement of data sources is requested in the usage guidelines. Time series data presented in this report have been generated with data from the Giovanni database.

The Oceancolor Web website (<http://oceancolor.gsfc.nasa.gov/>) provides access to NASA ocean colour data at various levels of processing. The Level 3 browser enables visualisation of the entire global ocean colour data set (SeaWiFS and MODIS Terra/Aqua) for a large number of standard and test parameters. Data can be downloaded for individual days or for composites over a range of time periods from 3-days to monthly, annual and mission duration. Data can be selected at either 4 or 9 km spatial resolution, but only global data sets can be downloaded as there is no provision for selecting sub-scenes. Two file formats are provided: .png images and .hdf for subsequent data analysis. NASA do not require registration or subscription for this service, but due acknowledgement of data sources is recommended.

The Globcolor project (<http://www.globcolour.info/index.html>) was funded by ESA to provide access to merged data sets from both MERIS (ESA) and MODIS (NASA). Data-merging is necessary to achieve global daily coverage, as no single sensor is capable of observing every part of the globe every day. Considerable effort has gone into data validation to ensure consistency across the full extent of each mission's spatial and temporal coverage. The full product data set is provided at 4km resolution from 1997 to the present day. A Near Real Time service is also available as are a number of demonstration products, such as primary production and heated layer depth.

The MyOcean website (www.myocean.eu.org) provides access to Earth Observation data, derived parameters, model forecasts and *in situ* data sets from a wide range of sources. It is EU funded and is the implementation project of the GMES Marine Core Service. Data are available on global and regional scales (e.g. Baltic Sea, Mediterranean Sea). Access to products is open and free, including for commercial activities, and data is available on a variety of time scales including near real time when possible. Parameters available include: Temperature, Salinity, Wind, Sea-level, Sea-ice, Current, and a wide range of ocean colour products.

NEODAAS (NERC Earth Observation Data Acquisition and Analysis Service – (www.neodaas.ac.uk) is a NERC Facility that receives, processes and archives data from a

number of satellite systems. It is located on two sites: Plymouth Marine Laboratory (Remote Sensing Group) and the Dundee Satellite Receiving Station (DSRS). It is primarily funded to provide UK research scientist with remote sensing data and information. Access to data is normally free for Academic research and may be negotiable for non-Academic research. Data from SeaWiFS, MODIS (Aqua and Terra) and MERIS can be accessed via a data portal with graphical selection of time and geographic location. Products include Chlorophyll and Quasi-True Colour (RGB) images. NEODAAS offers a Near Real Time service and support for specific research programmes and cruises.

Annual and monthly global net primary production estimates using three different photosynthesis models are available from the Ocean Productivity web page (<http://www.science.oregonstate.edu/ocean.productivity/index.php>). The quality of these products for Scottish coastal waters is essentially unknown and caution should be exercised in the interpretation of such data. Online products are provided as monthly 1080 x 2160 global grids in .hdf format, but other formats can be negotiated.

4.7 Capabilities and Limitations of Ocean Colour Remote Sensing

4.7.1 Applications of OCRS

Ocean colour remote sensing is a relatively new field of research that has still to reach full maturity. As discussed elsewhere in this document, the quality of standard products is a limiting factor for many applications. However, it is also true that there are few, if any, practical alternatives to the use of satellite data when the subject involves large expanses of ocean and extended time periods. Thus the applications of OCRS are many and varied, with many sectors of the marine economy increasingly dependent on this type of data.

The IOCCG has produced two reports on applications of OCRS that are relevant to this study: IOCCG Report Number 7 (IOCCG 2008) discusses general applications of OCRS and their societal impact, while IOCCG Report Number 8 (IOCCG 2009) focuses on application in fisheries and marine aquaculture. All of the IOCCG reports are available for download (http://www.ioccg.org/reports_ioccg.html) .

Through its sensitivity to phytoplankton and other materials suspended and dissolved in the water column, OCRS provides a unique window onto biogeochemical processes occurring in surface layers of the ocean. Ocean colour has been designated as an "essential climate variable" needed to support the carbon cycle monitoring requirements of the UNFCCC (United Nations Framework Convention on Climate Change). Environmental protection legislation and response to public concern has resulted in renewed focus on monitoring and evaluating the state of marine systems, with an increasing burden falling on government agencies. With suitable critical analysis, and further development of validated products, OCRS has the potential to make a significant contribution towards supplying information urgently required by policy makers and monitoring agencies.

Current general applications for OCS include:

- Assimilation and validation for ecosystem models on a variety of scales from local regional forecasting to long term global climate change;
- Investigating links between physical and biological process on a variety of spatial / temporal scales e.g. meso-scale eddies, regional upwelling events;
- Assessment of natural carbon reservoirs through determination of concentrations of POC, phytoplankton carbon, calcium carbonate and CDOM, as well as primary productivity;
- Water quality monitoring, including effects of eutrophication and determining water clarity;
- Assessing phytoplankton abundance and functional type on spatial / temporal scales ranging from coastal zones to ocean basins and short term forecasts to inter-annual trends;
- Monitoring natural hazards such as harmful algal blooms (HABs), sediment plumes and benthic habitat loss.

Applications with particular relevance for fisheries and marine aquaculture include:

- Monitoring local effects of climate change and natural ecosystem variability, e.g. timing of the spring bloom;
- Managing areas of special interest e.g. large marine ecosystems, coral reefs and conservation areas;
- Predicting areas for profitable fishing (profitable fishing zones – PFZs) e.g. India and Japan;
- Monitoring environmental conditions for marine aquaculture operations;
- Protection of endangered species (e.g. loggerhead sea turtles and north Atlantic right whales) by improving knowledge of preferred habitats and behavioural patterns.

4.7.2 Potential Limitations - Cloud Cover

For Scottish waters, the availability of cloud-free images is perhaps the greatest limitation on the utility of ocean colour. Figure 4.8 shows five years of cloud cover data for northeast Atlantic waters off the Scottish west coast obtained from MODIS data retrieved from the Giovanni database (<http://disc.sci.gsfc.nasa.gov/giovanni>) . Over this period there was greater than 90% cloud cover on ~38% of images and only ~5% of images had less than 50% cloud cover. Cloud cover is seasonal to some extent with clearest conditions most likely to occur in spring and generally high cloud fractions in winter.

Such generally high levels of cloud cover will limit the use of ocean colour imagery for certain types of applications such as real-time monitoring applications. Nevertheless, there is considerable scope for using ocean colour data in a variety of applications including monitoring long term trends in algal concentration.

Figure 4.9 shows 11 years of monthly averaged SeaWiFS chlorophyll concentration for northeast Atlantic waters off the Scottish west coast. The data shows significant seasonal and some inter-annual variability, though there are gaps in data availability each winter. The second panel reveals extreme variability in the number of pixels that contributed to the calculation of the mean chlorophyll concentration for each month. The statistical significance of such data will be affected by this type of data availability. Relaxation of solar zenith angle limits in the processing routine would greatly increase the number of data points for winter months.

Composite Images

Identification of persistent spatial features can be enhanced by generating composite images over extended time periods. A composite image blends data from multiple images into a single image, averaging pixels where more than one original value is available.

Figure 4.10 shows global 9 km resolution SeaWiFS chlorophyll images from: (a) 10 April 2009, (b) a composite between 7-14 April 2009, and (c) a monthly composite for April 2009. From an initial image that has large data gaps between adjacent swaths and areas missing due to clouds, increasing the composite period results in improved identification of major features such as the North Atlantic spring bloom. Note, however, that there is an implicit loss in temporal resolution and even this degree of compositing at this time of year leaves areas of the North Atlantic with blank pixels.

4.7.3 Atmospheric Correction and Adjacency Effects

Approximately 90% of the signal measured by an ocean colour radiometer at the top of atmosphere is generated by atmospheric scattering (Gordon, 1987) and has to be removed in order to access the true water leaving radiance signal. Early atmospheric correction schemes relied on an assumption that no light was reflected from within the ocean in the near infrared (NIR) due to overwhelming water absorption. It has since become apparent that this assumption is violated in turbid coastal waters and other regions where backscattering levels are sufficiently high to generate a significant reflectance signal in the NIR (Siegel *et al.*, 2000). New atmospheric correction schemes have been proposed to deal with this problem (e.g. Aiken and Moore, 2000), but application and validation remain patchy. Poor atmospheric correction is likely to reduce the quality of both radiometric retrievals and subsequent algorithm calculations by variable and largely unknown amounts.

The reflectance of light from the ocean and coastal waters is generally much lower than from land or clouds. Close proximity to bright areas causes unwanted artefacts known as adjacency effects. There are two different types of adjacency effect to consider. The first is caused by the response time of detector electronics to step changes in radiance levels as the sensor passes from land or cloud to sea and vice versa. As a result the first few pixels at the interface between land/cloud and sea are generally unreliable and should not be used. This places an additional constraint on how close inshore measurements can be obtained

beyond the intrinsic spatial resolution of the sensor design. The second type of adjacency effect is caused by multiple scattering of light reflected from land/cloud generating artefacts in measured reflectance signals from pixels up to 30 km away. Elevated CDOM levels in Figure 4.3 close to the large cloud above the north of Ireland are possibly an artefact of this type of adjacency effect.

4.7.4 Algorithm Performance in Coastal Waters

Vast areas of the global ocean are deep, clear blue oceanic waters that more or less satisfy the Case 1 optical classification in that their optical properties are dominated by water, phytoplankton and materials associated with phytoplankton. Variable concentrations of other optically active materials such as non-biogenic mineral particles and CDOM cause many coastal waters to fall into the Case 2 optical classification. Many algorithms have been developed either specifically for Case 1 waters or using data sets that have been dominated by Case 1 parameters, with the result that the algorithms are tuned to perform well in open ocean areas.

A number of studies have shown that the performance of standard algorithms (e.g. blue-green reflectance ratios) degrade significantly in turbid coastal waters. For example, McKee *et al.* (2007a) demonstrated that the standard SeaWiFS chlorophyll algorithm (OC4v4) generally overestimated chlorophyll by as much as an order of magnitude and the standard K490 algorithm systematically underestimated diffuse attenuation in turbid Irish Sea waters (Figure 4.11). Development of locally tuned algorithms can be achieved if sufficient quantities of *in situ* optical and constituent data can be amassed through campaigns of shipboard sampling. Figure 4.12 shows Irish Sea chlorophyll and K490 data for 29 January 2006 calculated using standard and locally tuned algorithms. Major differences are observed in turbid estuaries and other areas with strong resuspended mineral concentrations or high CDOM signals.

4.8 Preliminary Study of Available Data

4.8.1 NE Atlantic - Seasonal Composite Imagery

The optical properties of Scottish West Coast waters have not been extensively researched, with virtually no *in situ* data having been collected on the shelf outside the sea-lochs/Clyde Sea. The absence of such data places a significant limit on the extent to which OCRS data can be interpreted as algorithm performance is essentially unknown for this region. Previous experience in areas such as the Irish Sea can be used to give an informed opinion for this region, but it must be remembered that the following interpretation of OCRS data from Scottish West Coast is currently not supported by a strong knowledge base for this region.

Figure 4.13 demonstrates some of the potential pitfalls associated with interpretation of OCRS data. The MODIS standard chlorophyll product is based on a blue-green reflectance algorithm that is sensitive to the presence of resuspended sediments as well as

phytoplankton. Areas which are known to support significant loads of suspended sediments such as the southern North Sea, the Irish Sea/Bristol Channel and the Thames/Humber estuaries all appear to exhibit high levels of chlorophyll concentration, even in winter when concentrations would be expected to be low.

One way to assess the potential impact of suspended sediments on the performance of the chlorophyll algorithm is to identify areas with strong reflectance signals, as sediments are known to efficiently backscatter light. Figure 4.14 shows remote sensing reflectance signals at 555 nm for the same scenes. Figure 4.14(a) indicates raised scattering signals across all of the areas mentioned previously and elevated reflectance signals around the Scottish coastline, calling into question the quality of the chlorophyll estimate for almost the entire Scottish coastal zone in winter.

Reflectance levels in spring, summer and autumn are generally lower, probably as a result of reduced wind inputs and the onset of thermal stratification in surface waters. Slightly elevated reflectance signals in the Minch and off the north coast of Scotland in summer may be attributable to coccolithophore blooms which are known to form along the shelf edge during summer months and strongly reflect light from the calcite which forms the exoskeletons for these phytoplankton.

Returning to the chlorophyll images in Figure 4.13, it seems likely that Figure 4.13(b) presents a reasonable picture of the spring phytoplankton bloom, at least away from the immediate coastal zone, with chlorophyll concentration in the region of 1 - 10 mg m⁻³ across most of the western shelf. Chlorophyll levels generally drop significantly in the summer (Figure 4.13(c)) after the spring bloom has depleted nutrient levels in surface waters, though there may well be reasonable concentrations of chlorophyll still being produced on the northwest shelf thanks to a supply of nutrients from mixing processes in this area. Lastly, by autumn (Figure 4.13(d)) chlorophyll concentrations have reduced back to low levels around 1 mg m⁻³ or less across most of the region. It is noticeable that chlorophyll appears to remain high in the Clyde Sea and the Moray Firth, even though reflectance signals are low in these areas in autumn. It is possible that this is an artefact associated with high CDOM levels which can also disrupt the performance of blue-green reflectance ratio algorithms.

4.8.2 West of Scotland Monthly Averaged Time Series

The west coast of Scotland is a complex area with features ranging from semi-isolated sea lochs with restricted exchange to outside waters, through shallow shelf seas extending out to the edge of the continental shelf and deep oceanic waters further to the west. The current study is focused on ICES area VIa shown under the main grid in Figure 4.15. This region is home to the full range of complex bathymetry mentioned above and is subject to a series of interlinked physical mixing processes that, together with effects of materials entering the sea from terrestrial sources, strongly influence biogeochemical cycles in these waters. It is highly probable that the performance of standard remote sensing algorithms will vary

considerably across such a complex and varied region due to varying degrees of influence from suspended sediments and CDOM (e.g. from freshwater inputs).

The modelling effort for the Desk Study is restricted to shelf waters, defined as up to the 200m depth contour. Figure 4.15 shows how this area has been further partitioned into four sub-areas:

- Area 1: Shelf Edge – West
- Area 2: Shelf Sea – Minch
- Area 3: Shelf Edge – North
- Area 4: Clyde Sea

These sub-areas have been selected as being potentially representative of areas with different physical/biogeochemical characteristics and potentially different remote sensing algorithm performance.

Areas 1 and 3 are both shelf edge areas and have been selected separately to assess the degree of similarity.

Area 2 encompasses the Sea of the Hebrides, the Little Minch and part of the Minch as well as extensive areas of shelf sea to the north and south. This area is sufficiently shallow that strong mixing events, e.g. winter storms, could bring resuspended sediments to the surface and impact on remote sensing signals, but in summer months it is likely that thermal stratification in surface waters would result in low sediment concentrations. CDOM levels are essentially unknown for this area, but it seems likely that freshwater inputs would be quickly diluted and CDOM absorption is unlikely to have a big impact on remote sensing signals for most of this area.

The Clyde Sea (Area 4), however, is a region of strong freshwater influence and CDOM absorption measurements in this region have covered a broad range that could have a significant impact on remote sensing signals (e.g. McKee *et al.*, 1999; Binding and Bowers, 2003). Shallow areas of the Firth of Clyde are also potentially influenced by sediment resuspension as a result of vertical mixing processes.

Figure 4.16 shows time series of SeaWiFS monthly averaged chlorophyll concentration for each of these sub-areas. Also shown are corresponding numbers of pixels contributing to each monthly average. Data were extracted from the Giovanni database provided by the Goddard Earth Sciences Data and Information Services Center (NASA GES DISC), and cover the full mission period from September 1997 to February 2010.

Results

Regular seasonal patterns in chlorophyll concentration are easily seen for Areas 1-3, with peak spring bloom values in the 2 - 3 mg m⁻³ region. Note these are average values for

each sub-area; much wider ranges of values are found for individual measurements within each sub-area.

There is some inter-annual variability for Areas 1-3, with evidence for a late summer bloom in some years and not others. Area 2 shows marginally higher base levels of chlorophyll concentration ($\sim 1 \text{ mg m}^{-3}$) that might be an artefact from non-algal effects on the chlorophyll algorithm.

The Clyde Sea (Area 4) is significantly different and shows generally higher levels of chlorophyll in all seasons ($> 2 \text{ mg m}^{-3}$), with strong inter-annual variability in the magnitude of the spring bloom peak concentration (up to $\sim 8 \text{ mg m}^{-3}$). The Clyde Sea data must be treated with caution as it is the area where remote sensing algorithms such as the SeaWiFS blue-green chlorophyll algorithm used to generate this data are most likely to be influenced by non-algal materials.

Discussion

A key feature of this data set is the absence of data in winter months. Every winter there are entire months where no data has been collected that has passed through the quality control standards for the Giovanni database. This is primarily due to a combination of solar zenith angle limits and cloud cover. In fact it is possible to find reasonable quality images in the archives for clear days during these periods, which can be satisfactorily processed if one is willing to relax the solar zenith angle restriction. The large range of variation in the number of pixels contributing to each set of monthly averages reflects the difference in area of each sample region. The weighted average for the combined area is not strongly influenced by the Clyde Sea data as a result of the small number of pixels contributing to the total. It has to be emphasised that the quality of this data is unknown as algorithm performance has not been validated for these waters. As indicated above, there are reasonable grounds for suspicion about various aspects of the data.

4.9 Future Perspectives in Ocean Colour

Climate change studies require robust time series data over prolonged periods (typically decade plus as a minimum) in order to establish trends beyond seasonal, inter-annual and other known temporal fluctuations (e.g. ENSO, AMO). The expected operational lifetime of any individual satellite is generally less than this (< 10 years). A rolling sequence of satellite platforms with appropriate sensor capabilities is therefore needed in order to provide the duration of time series observation required for climate change applications. Table 4.2 outlines a selection of scheduled ocean colour missions that, if successfully launched, will extend the current unbroken ocean colour time series that commences with the launch of SeaWiFS in 1997. A noticeable feature of this schedule is a gradual shift from the development of technology by space agencies such as NASA / ESA to operational programmes run by NOAA / EUMETSAT.

Table 4.2: Basic specifications for a selection of scheduled ocean colour sensors.

Sensor	Orbit	Agency	Resolution (km)	Swath (km)	Bands	Spectral Range (nm)	Scheduled Launch
OLCI (3A)	Polar	ESA/EUMETSAT	0.3/1.2	1270	21	400-1020	2013
OLCI (3B)	Polar	ESA/EUMETSAT	0.26	1265	21	390-1040	2017
VIIRS (NPP)	Polar	NOAA/NASA	0.37/0.74	3000	22	402-11800	2011
VIIRS (JPSS1)	Polar	NOAA/NASA	0.37/0.74	3000	22	402-11800	2015
HIS	Polar	DLR (Germany)	0.03	30	228	420-2450	2013
S-GLI	Polar	JAXA (Japan)	0.25/1	1150-1400	19	375-12500	2014
MOC	Polar	INPE/CONAE (Brazil/Argentina)	0.2/1.1	200/2200	16	380-11800	2015

A key challenge for the exploitation of satellite Earth Observation data in climate studies will be successful merging of data from different sensors/platforms. This is a significant challenge because satellite data is subject to uncertainties for reasons ranging from sensor calibration, through differences in waveband selection to differences in atmospheric correction and algorithm selection (amongst many others). As a result, simple attempts to merge e.g. SeaWiFS, MODIS and MERIS data to provide long time series products over periods of time sufficiently long to inform climate change research run into difficulties due to sensor mis-match. The GlobColour project has attempted to merge data from SeaWiFS, MODIS and MERIS and has assessed different statistical approaches to the problem (http://www.globcolour.info/products_description.html). It is logical to anticipate that considerable future effort will go into merging data from multiple sources to provide multiparameter data sets in optimal formats for cataloguing and further exploitation. Within Europe, the GMES (Global Monitoring for Environment and Security) programme will likely continue to play a leading role in bringing together suppliers and users of Earth Observation data.

4.10 Conclusions and Recommendations

Scotland has stewardship of a large area and volume of ocean. The west coast in particular contains a huge diversity of environmental conditions ranging from intertidal zones through a variety of nearshore waters and shallow shelf seas out to the deep abyssal ocean.

Monitoring such an expansive and complex environment is a daunting task. Yet that is the challenge that is posed by legislation such as the European Union Marine Strategy Framework Directive, the Water Framework Directive and a number of other legally binding agreements. Scotland's economy is strongly dependent on the marine sector through energy production interests (renewable and fossil fuel extraction), fisheries and aquaculture, transport, tourism and leisure amongst others. Developing a knowledge base on the current state of the marine environment and monitoring its future development is an essential step

towards achieving the Scottish Government's vision for a marine environment which is: "...clean, healthy, safe, productive and biologically diverse managed to meet the long-term needs of nature and people".

Ocean colour remote sensing has a role to play as a tool for monitoring Scottish waters. The areal coverage required for such a large expanse of sea is a strong driver for further exploring the potential of OCRS. It has been shown here that OCRS can provide extremely valuable insights into large scale processes over extended periods of time in a manner that no other technology is capable of producing. Even with the acknowledged weaknesses of the technology, notably susceptibility to cloud cover and variable performance of algorithms in optically complex waters, there is clearly much to be gained from exploitation of OCRS.

OCRS does not provide a standalone solution for monitoring marine waters: data is patchy temporally and spatially, only surface waters are observed and many other parameters are needed to form a robust picture the state of ecosystems. However, OCRS can provide an additional dimension to other surveys: shipboard, moorings, airborne, AUVs and gliders. Understanding the limitations and realising the opportunities that the existing data sets and current technology afford is key to maximising the benefits of OCRS. Investment in the underpinning science and integration of OCRS into a broader monitoring and modelling strategy should be amongst the next steps in development of a strategy for managing the Scottish marine environment.

The first recommendation is for a survey of Scottish coastal waters to determine biogeochemical and associated optical characteristics across different regions and seasons in support of development of regional algorithms. Uncertain quality of OCRS data in this region is an unacceptable limitation that could be greatly ameliorated through a baseline study.

The second recommendation is to examine the potential for development of an integrated West Coast observatory (e.g. Figure 4.17), merging data from various platforms and regular surveys together with coupled physical/ecosystem models, to provide a multilevel overview of current and changing environmental conditions. With its wide variety of habitats, the west coast of Scotland would be an ideal natural laboratory for a program of this nature and the potential social and economic benefits would be substantial. This approach has been adopted by a number of very high profile programs in the USA and Canada, e.g. the NEPTUNE project (www.neptune.washington.edu) and the Martha's Vineyard Coastal Observatory (<http://www.whoi.edu/mvco>) , and in the UK by the Western Channel Observatory (<http://www.westernchannelobservatory.org.uk/>) and the Liverpool Bay Coastal Observatory (<http://coastobs.pol.ac.uk/>) . In situ and remotely sensed optical signals provide an essential link between different measurement platforms for each of these observatories.

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4.12 Figures

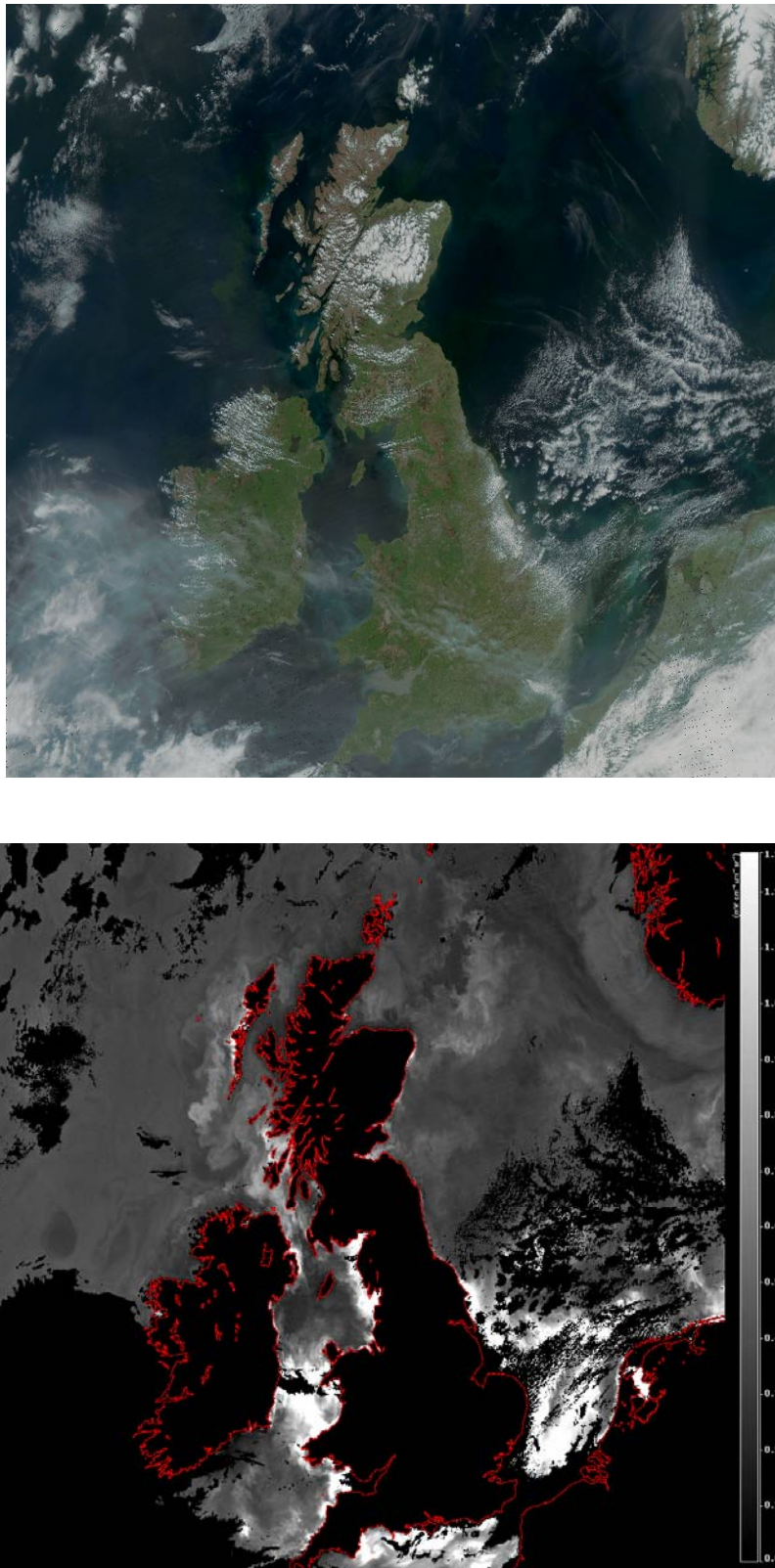


Figure 4.1: (a – upper figure) Quasi-true colour, and (b – lower figure) normalised water leaving radiance at 551 nm images from MODIS over the British Isles on 11 May 2009.

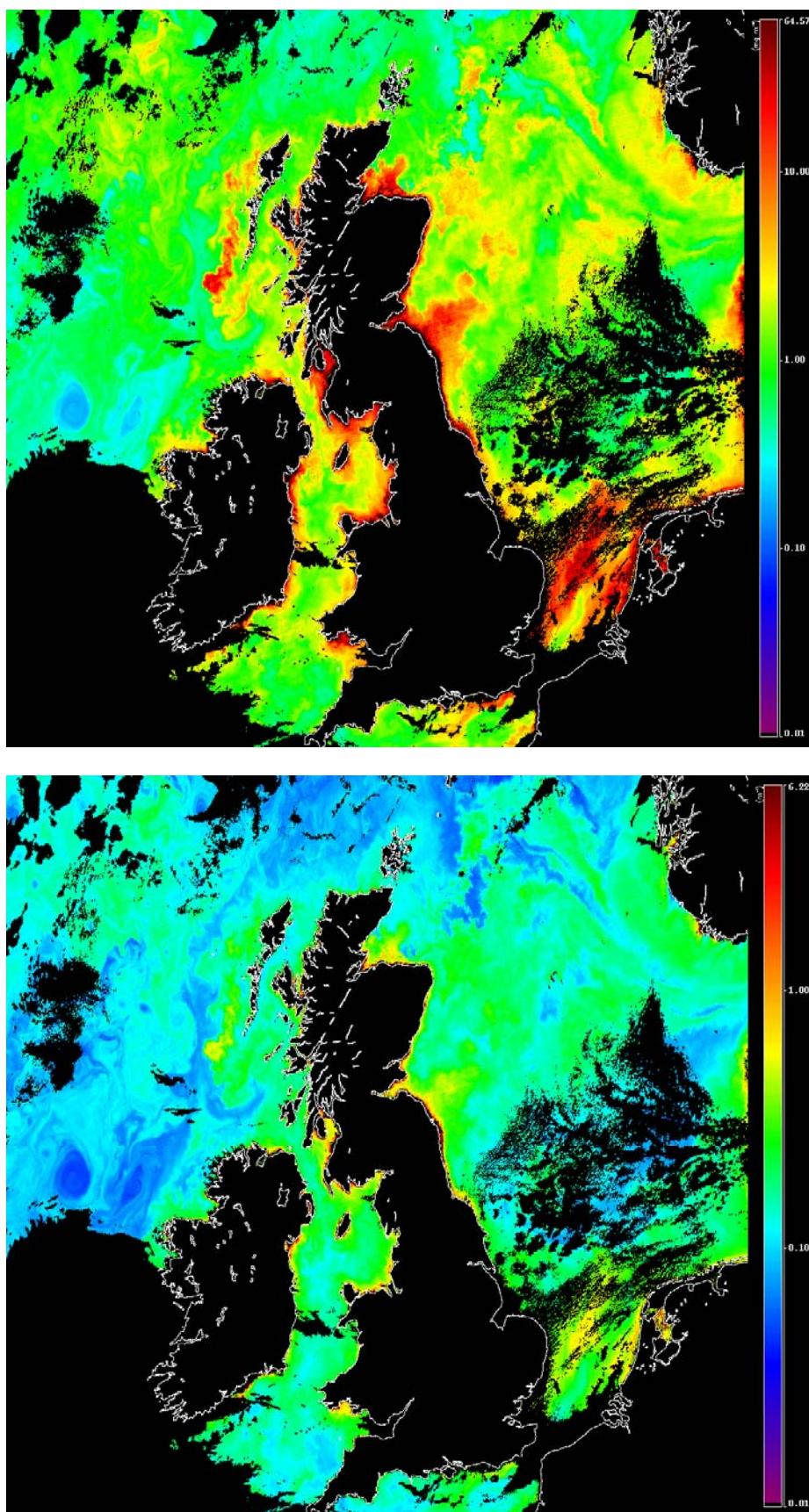


Figure 4.2: (a – upper figure) Chlorophyll, and (b – lower figure) Diffuse attenuation (K490) products generated using standard MODIS algorithms for 11 May 2009.

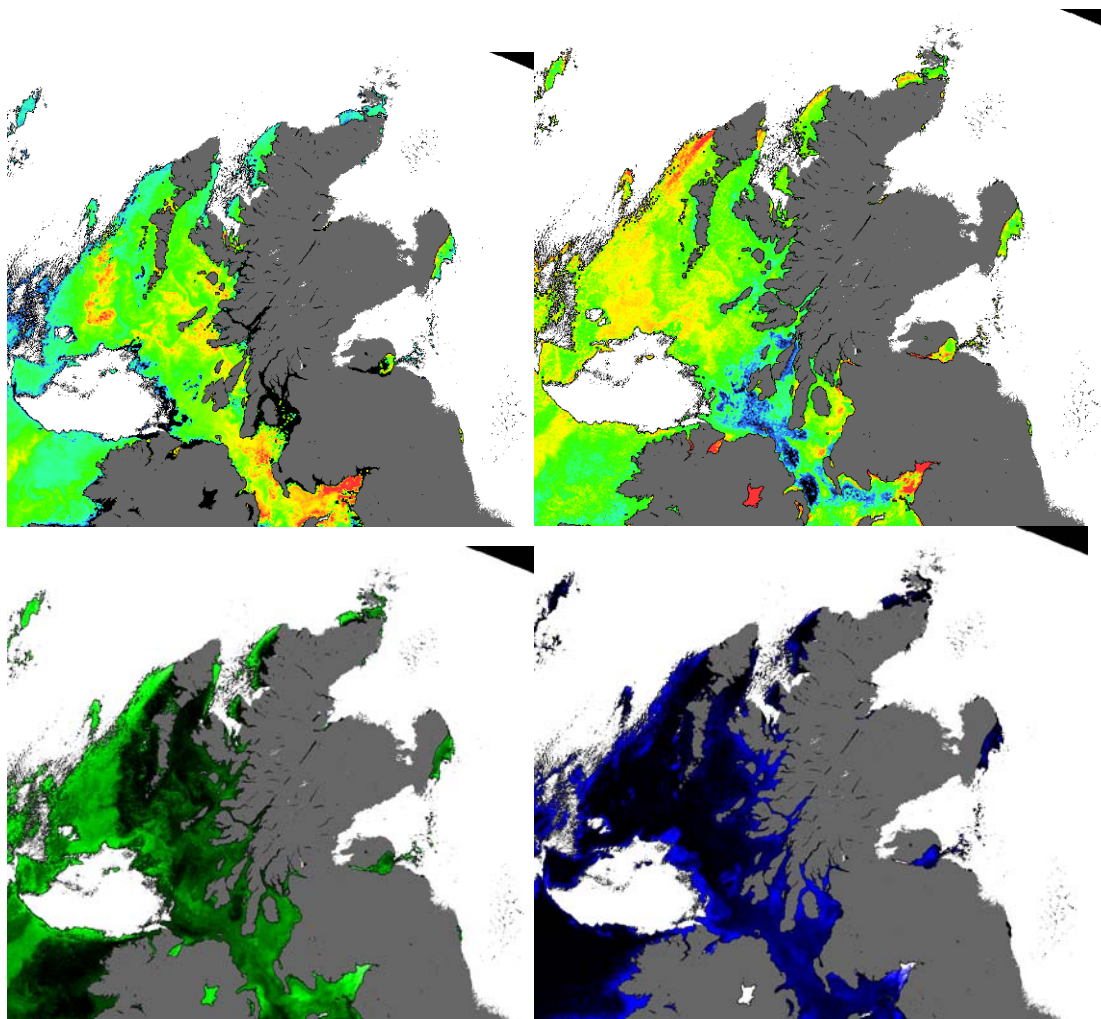


Figure 4.3: Scottish west coast waters imaged by MERIS on 3 May 2007. Estimates of chlorophyll concentration calculated from (a – upper left) blue-green reflectance ratios, and (b – upper right) a neural network show significant discrepancies. The neural network also provides estimates of (c – lower left) non-absorbing mineral concentration (a proxy for non-biogenic mineral particles), and (d – lower right) coloured dissolved organic material (CDOM).

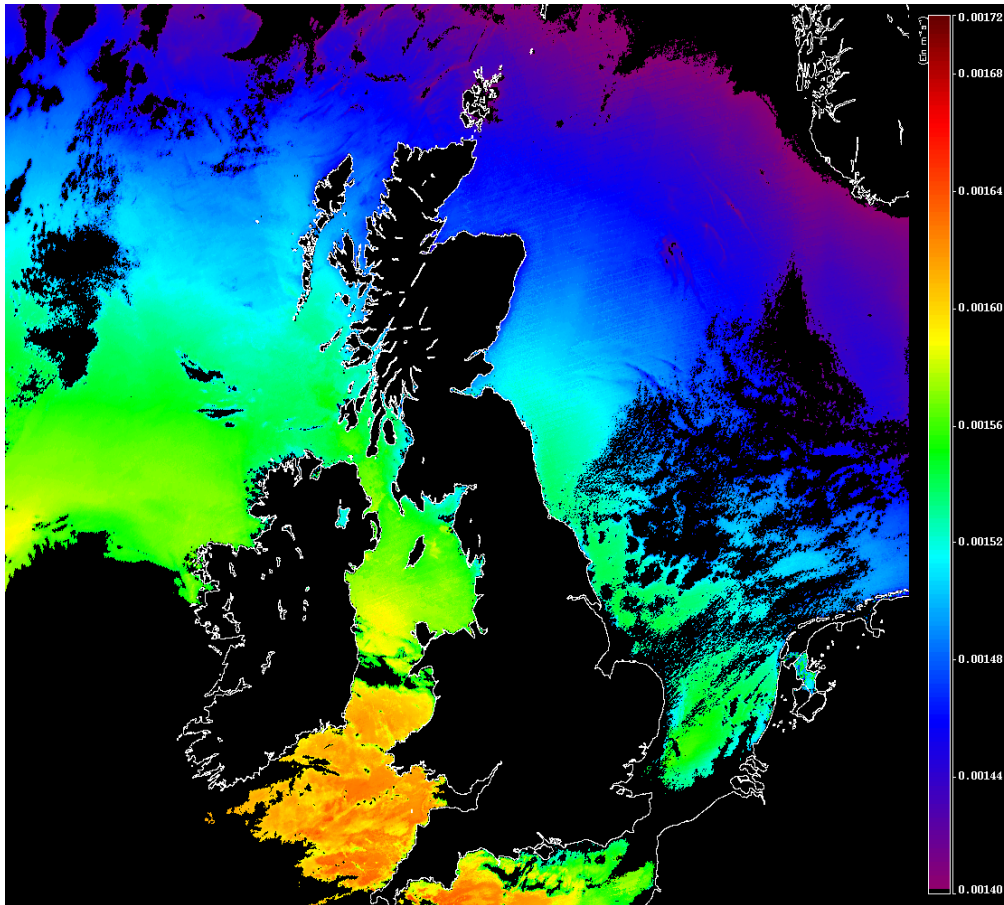


Figure 4.4: MODIS instantaneous photosynthetically available radiation on 11 May 2009.

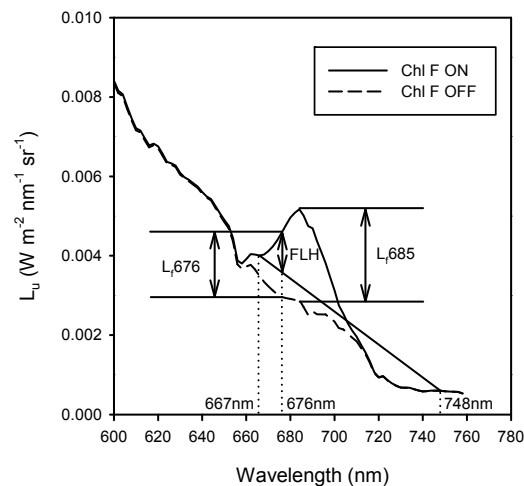


Figure 4.5: Simulated reflectance spectra with (solid line) and without (dashed line) Sun induced chlorophyll fluorescence. Although the peak fluorescence occurs at 685 nm, the MODIS Fluorescence Line Height (FLH) algorithm uses 676nm to avoid an atmospheric absorption band. The baseline is calculated using signals at 667 and 748nm.

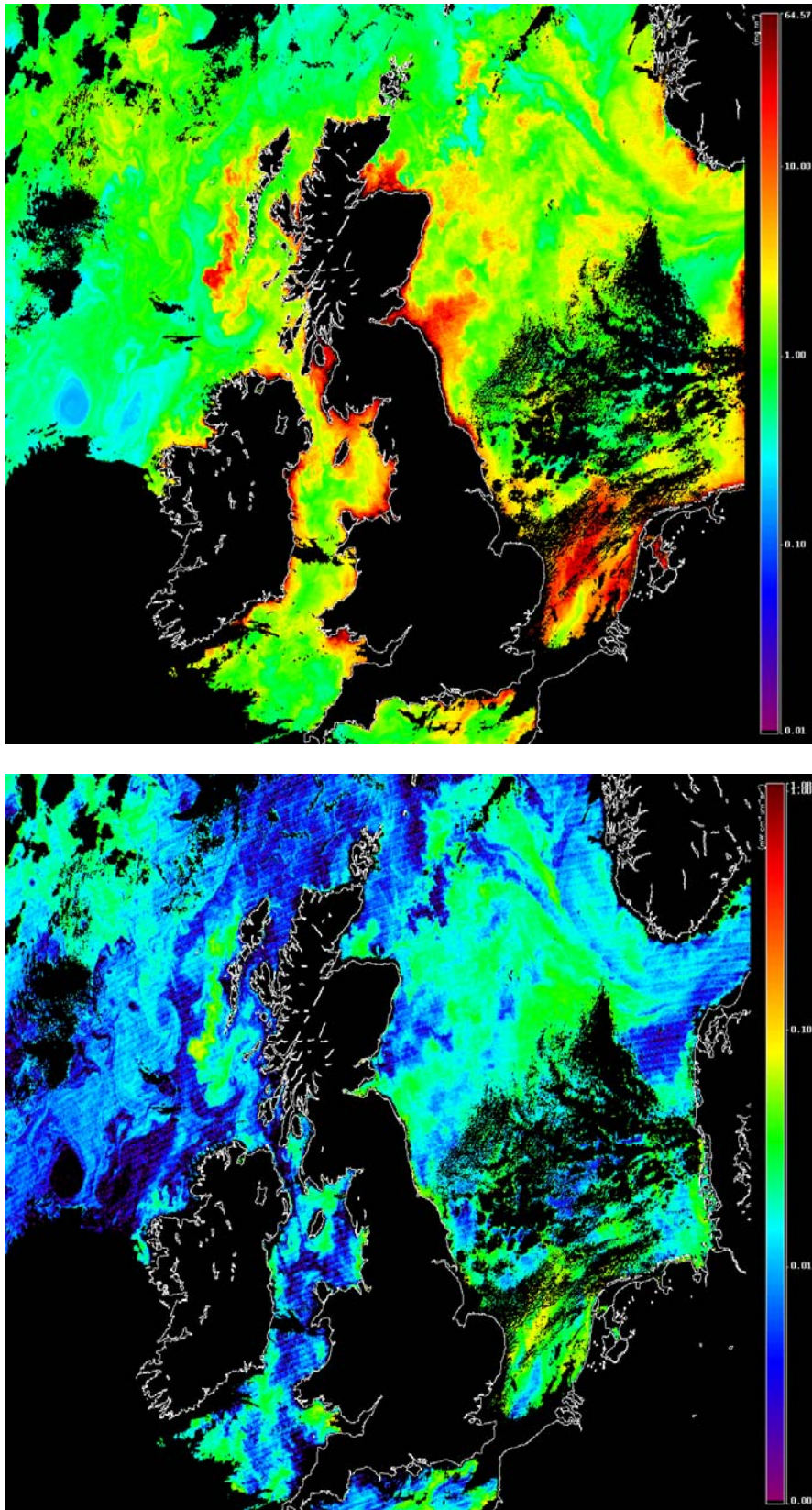


Figure 4.6: (a – upper figure) Chlorophyll, and (b – lower figure) Fluorescence line height (FLH) products generated using standard MODIS algorithms for 11 May 2009.

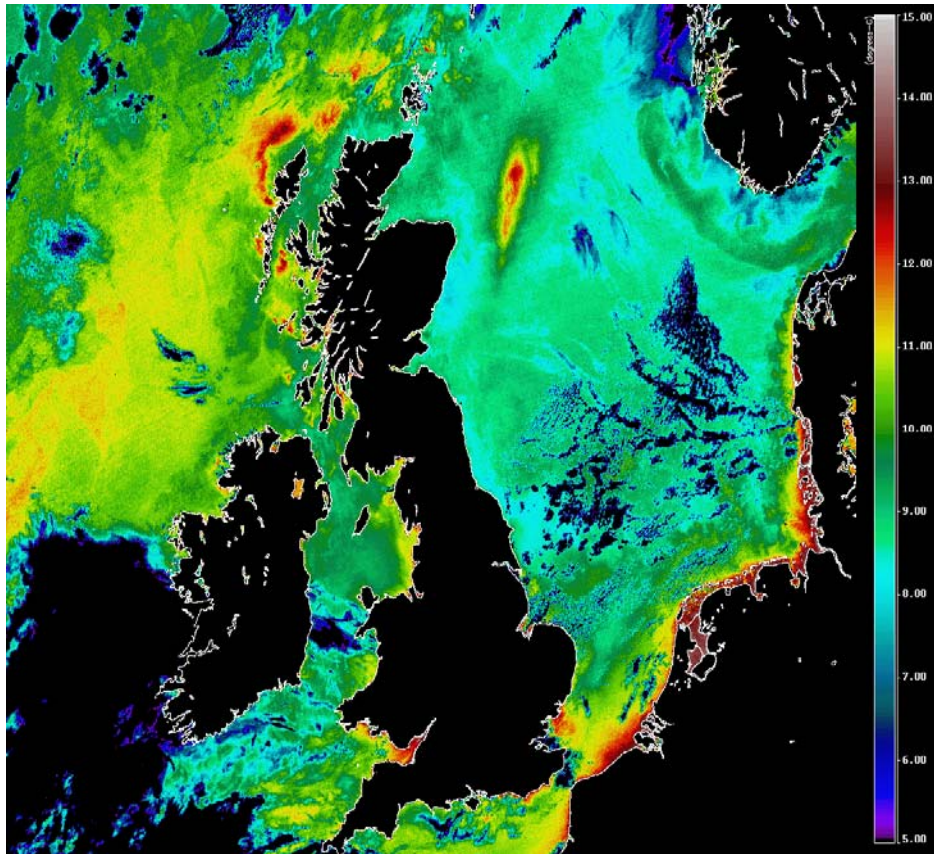


Figure 4.7: MODIS Sea Surface Temperature for 11 May 2009. Note artefacts in image associated with edges of clouds.

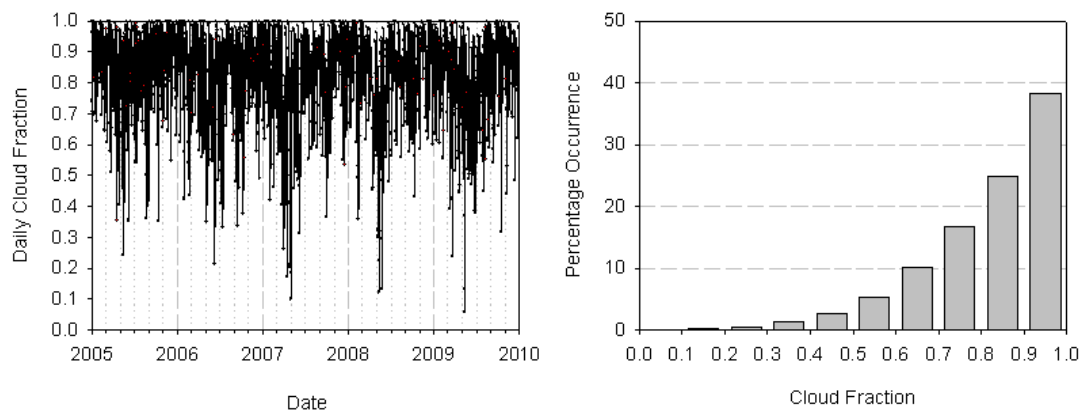


Figure 4.8: (a – left figure) Time series of daily cloud cover, and (b – right figure) distribution of cloud cover over the northeast Atlantic. ~95% of images have more than 50% cloud cover.

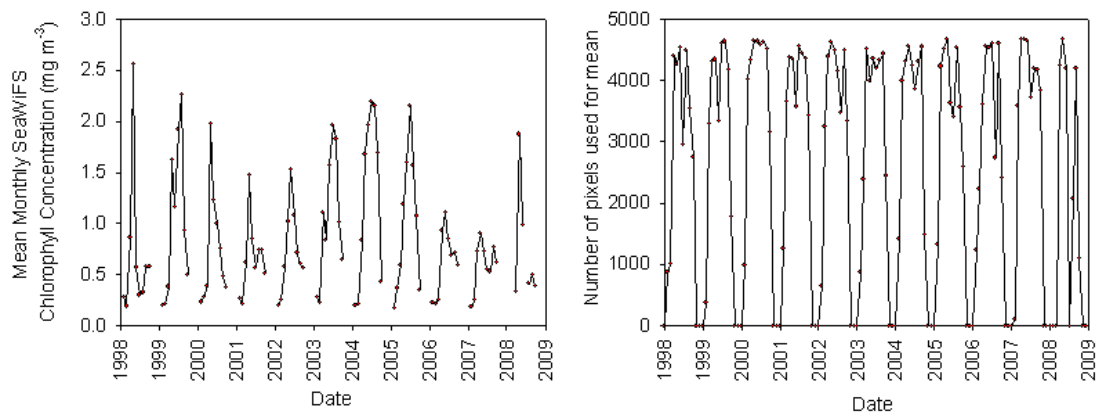


Figure 4.9: (a – left figure) Mean monthly SeaWiFS chlorophyll data shows seasonal and inter-annual trends, with gaps in winter months where non data is available due to cloud cover. (b – right figure) The number of cloud-free pixels varies seasonally and is often zero in winter months.

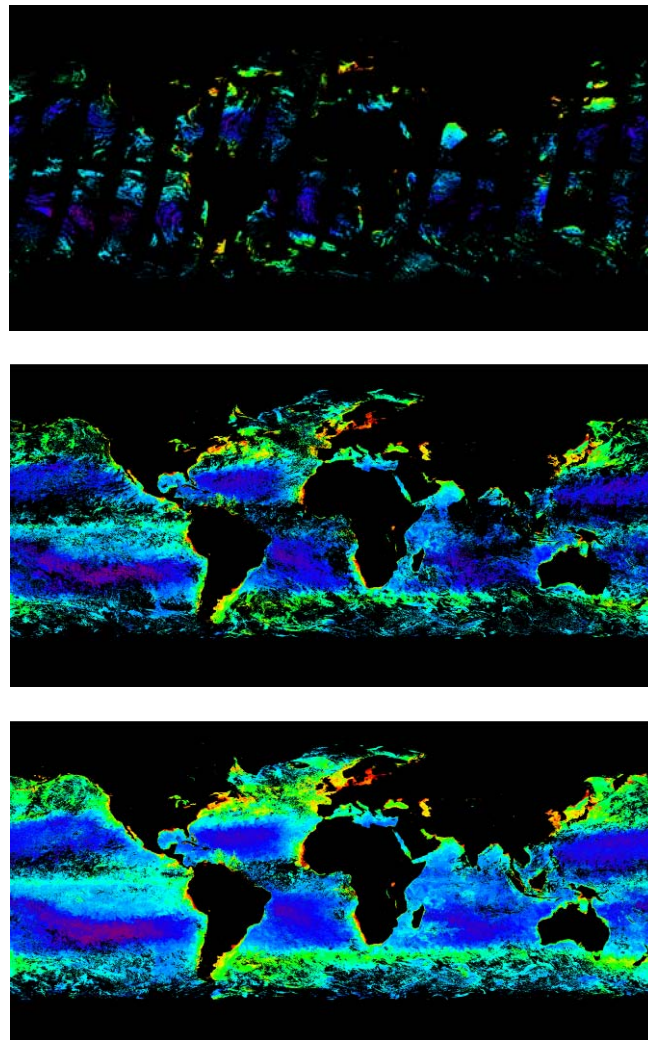


Figure 4.10: Global 9km spatial resolution SeaWiFS chlorophyll data from (a - upper) a single day, (b - middle) an 8-day composite and (c - lower) a composite for the month of April 2009. Averaging over longer time periods increases the probability of obtaining cloud-free data but even after a month there are many blank pixels.

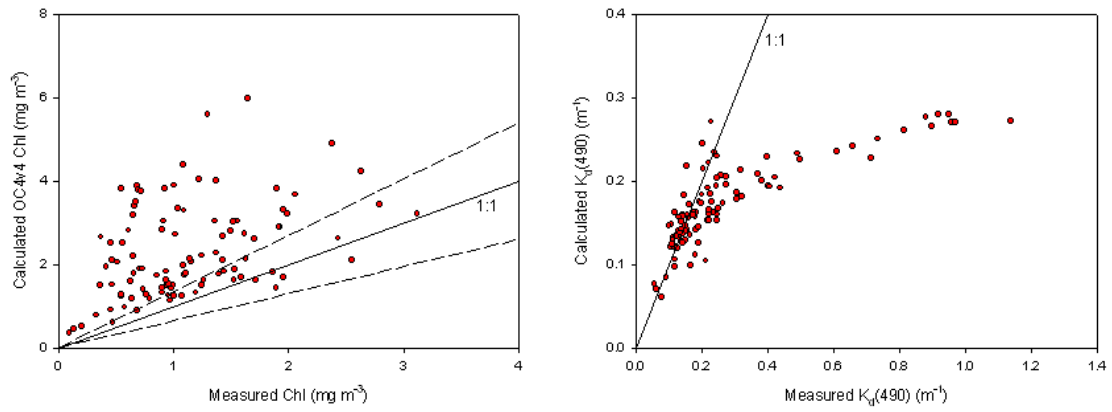


Figure 4.11: Data from a study in the Irish Sea. (a - left) OC4v4 chlorophyll derived from *in situ* radiometry generally overestimates measured chlorophyll by as much as an order of magnitude. Also shown are the SeaWiFS $\pm 30\%$ mission targets (dashed lines). (b - right) The standard K490 algorithm applied to *in situ* radiometry systematically underestimates the measured diffuse attenuation at 490 nm in turbid waters.

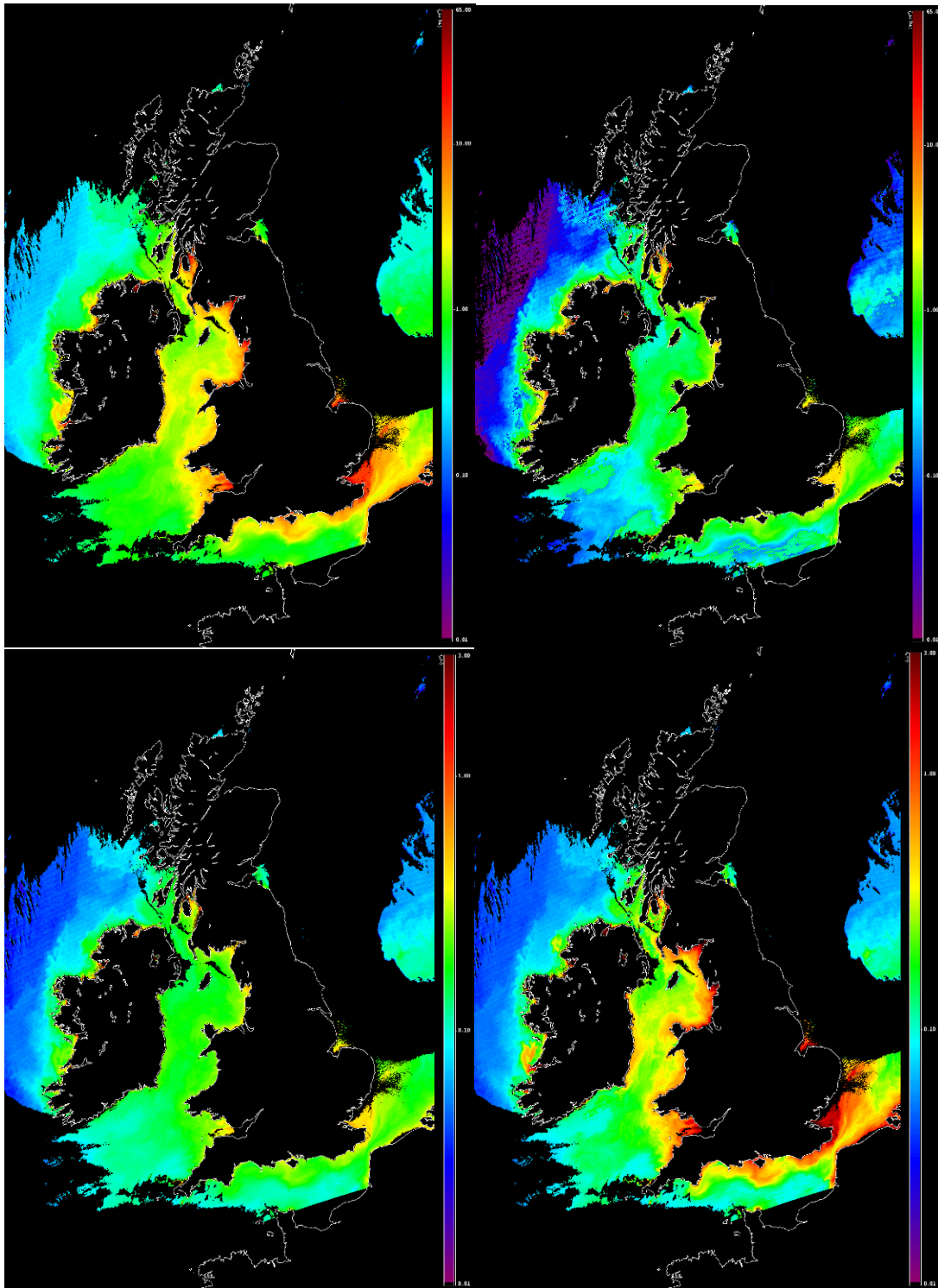


Figure 4.12: (a – upper left) Standard chlorophyll, (b – upper right) locally tuned chlorophyll, (c – lower left) standard K490 diffuse attenuation, and (d – lower right) locally tuned K490. Strong sediment resuspension causes standard blue-green algorithms to overestimate chlorophyll and underestimate K490 in UK coastal waters on the 29 January 2006.

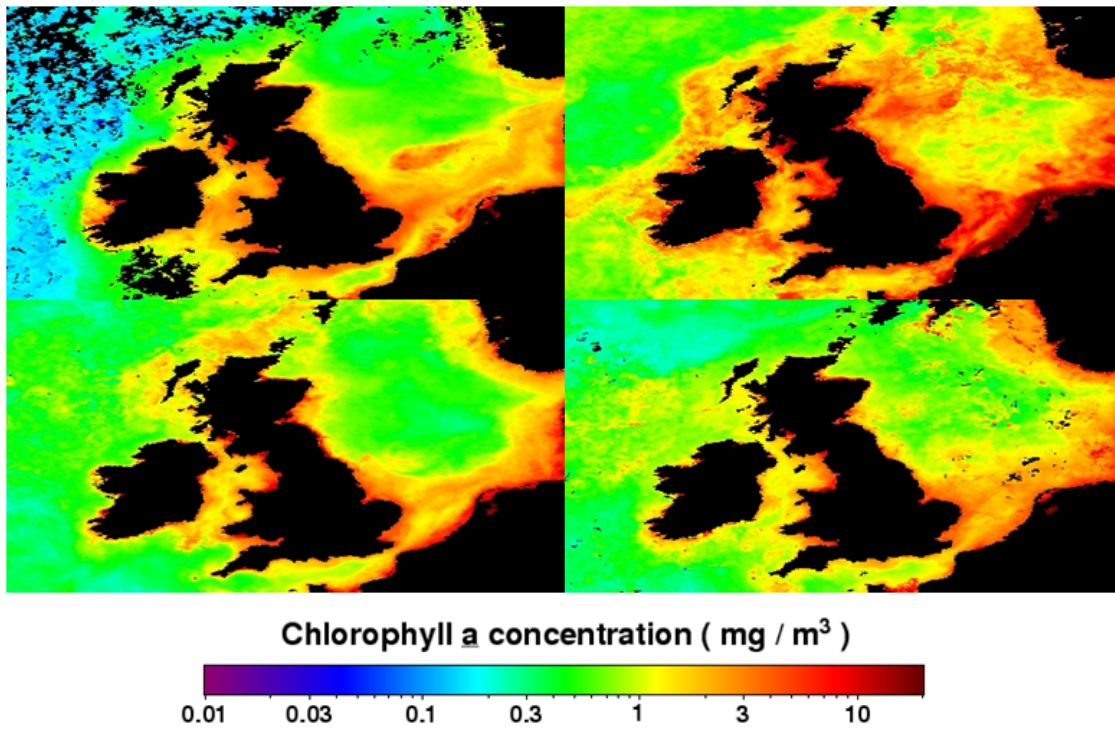


Figure 4.13: Seasonal composites of MODIS Aqua standard chlorophyll product at 4 km resolution for: (a – upper left) winter 06/07, (b – upper right) spring 07, (c – lower left) summer 07, (d – lower right) autumn 07.

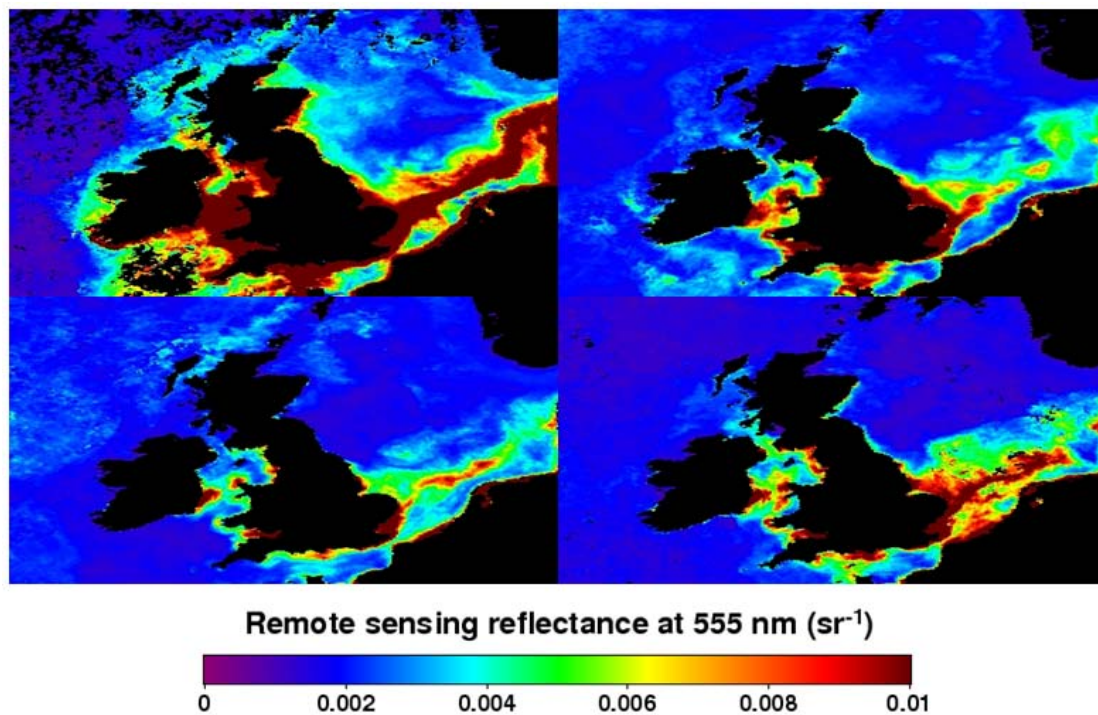


Figure 4.14: Seasonal composites of MODIS Aqua 555nm remote sensing reflectance at 4 km resolution for: (a – upper left) winter 06/07, (b – upper right) spring 07, (c – lower left) summer 07, (d – lower right) autumn 07.

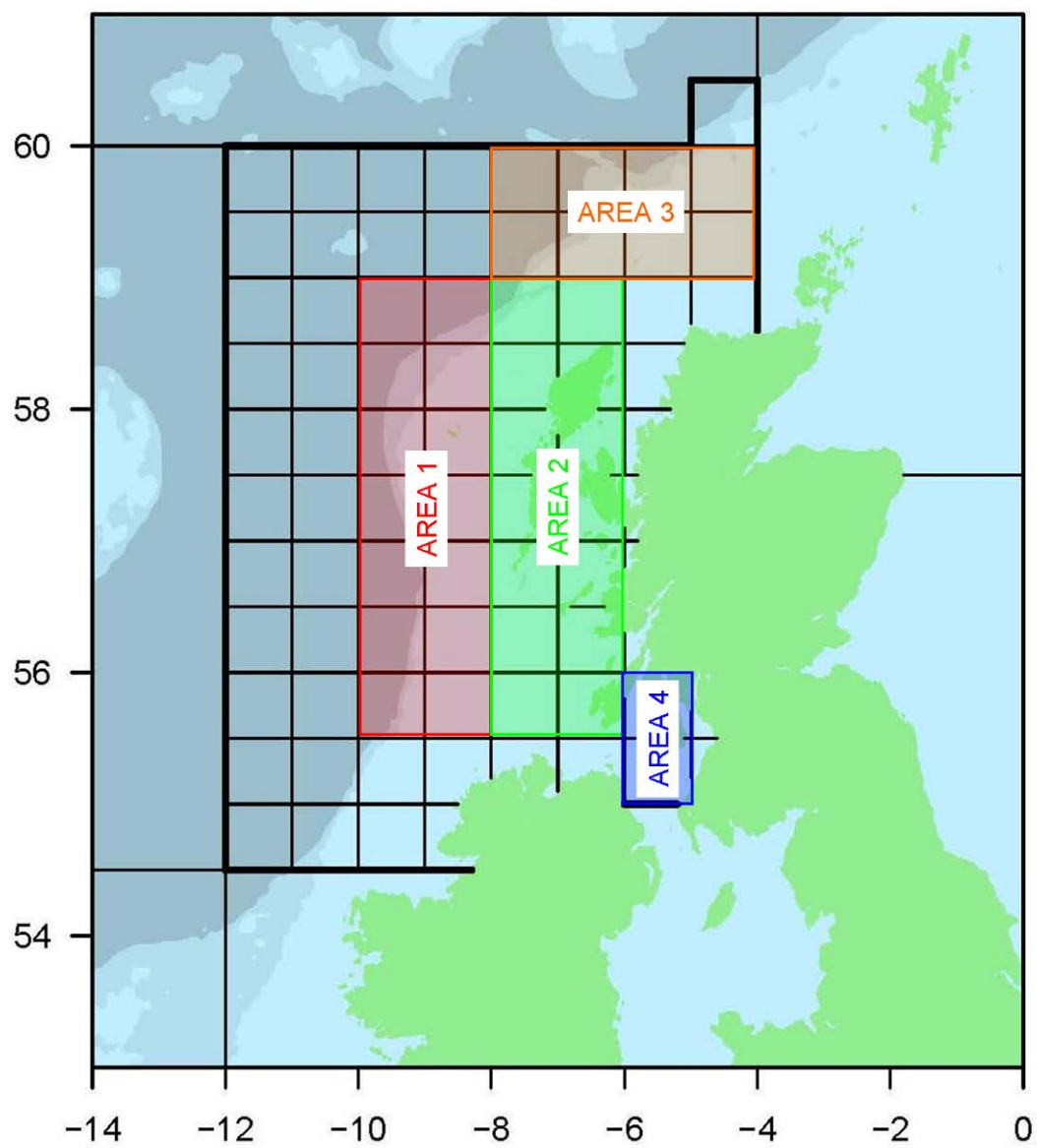


Figure 4.15: Bold grid denotes ICES area VIa with sub-areas highlighted for data extraction purposes.

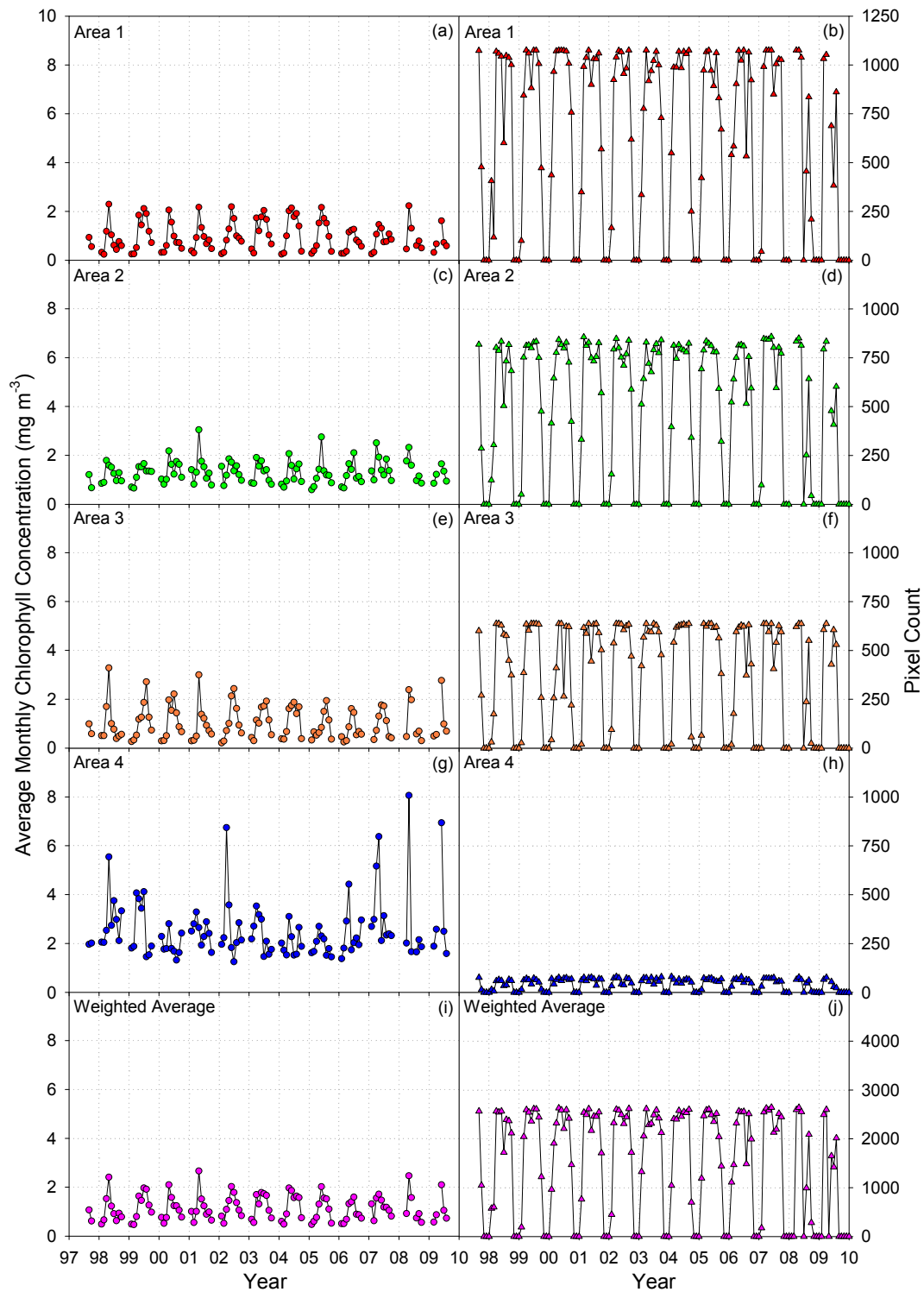


Figure 4.16: (a), (c), (e) and (g) Time series of monthly averaged SeaWiFS Chlorophyll a concentration for west coast regions. (b), (d), (f) and (h) Number of pixels contributing to each monthly average. (i) Weighted monthly averaged chlorophyll concentrations for the combined areas. (j) Total number of pixels contributing to each weighted monthly average (note extended y-range).

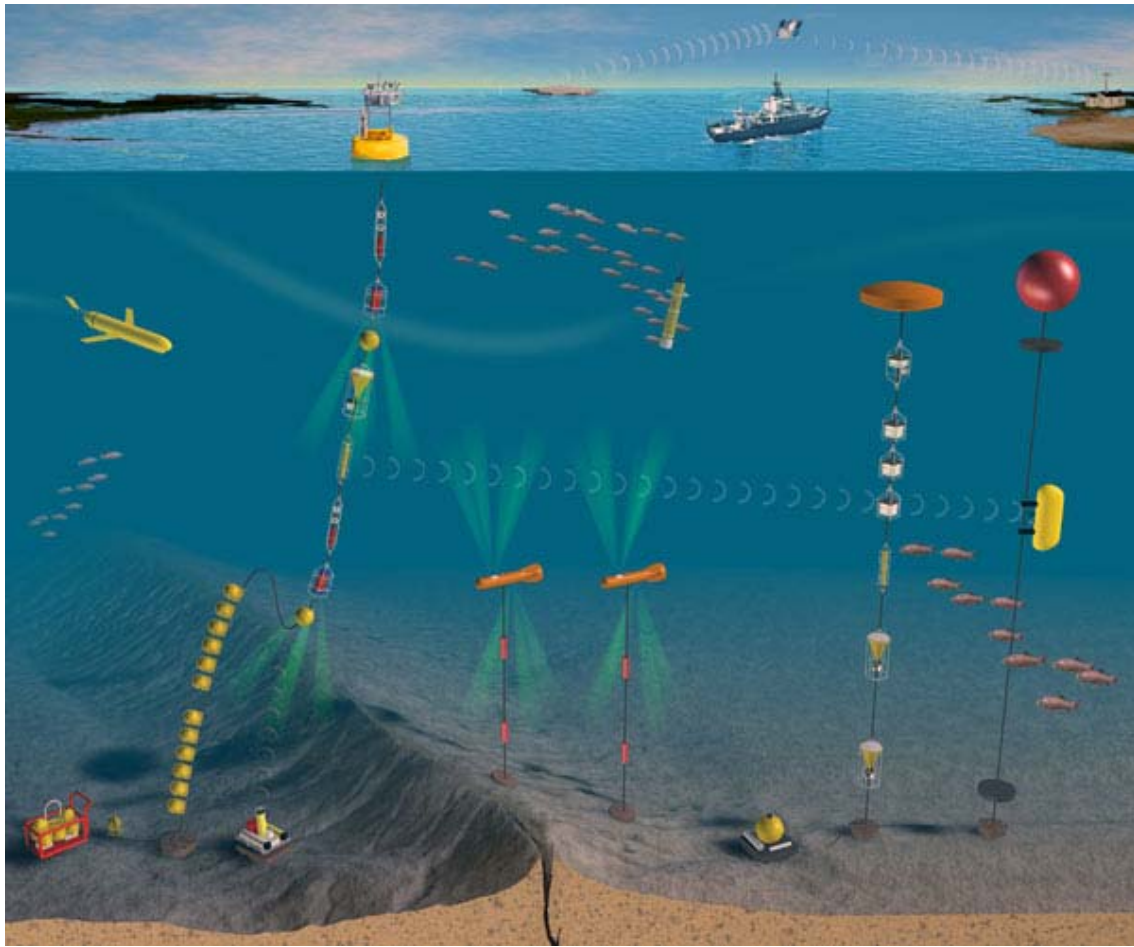


Figure 4.17: An example of a coastal observatory at the Woods Hole Oceanographic Institution. (Illustration by Jack Cook, Woods Hole Oceanographic Institution. Image courtesy of the OceanSITES program).

5 Whitefish Stocks: Role of the Coastal Zone

M R Ryan, L C Bellini, I P Smith and D M Bailey

5.1 Aims and Objectives

The purpose of this section is to review the available literature and data on the use of coastal zones and offshore reefs by Atlantic cod, *Gadus morhua*, haddock, *Melanogrammus aeglefinus* and whiting, *Merlangius merlangus* off the West Coast of Scotland. Furthermore, we will identify potential drivers of change in nearshore ecosystems that have the potential to affect overall fish stocks, and highlight how monitoring programmes in the nearshore environment can identify recruitment trends.

The problems affecting these populations have been reviewed extensively elsewhere, as has the fact that while North Sea populations appear to be responding to changes in management, the West Coast stocks are not. It is immediately apparent that the oceanographic and hydrographic regimes differ greatly between these two sea areas, with the West Coast having a much greater degree of habitat heterogeneity, with extensive rocky reefs, islands, straits and sea lochs. Only the far north-eastern area of the North Sea has a similar coastal environment to the West Coast of Scotland.

Many marine species occupy coastal or reef systems during critical periods in their ontogeny and processes occurring in these systems affect population dynamics. Because reef and nearshore fish communities are difficult to monitor using the standard fisheries survey methodologies, it is possible that unobserved changes to these systems are an important driver of fish numbers in the adult stocks.

This section reviews the current state of knowledge of the use of Scottish nearshore by cod, whiting, and haddock. We collected information from the peer-reviewed literature, and from ongoing sampling programmes for coastal marine fish. Relevant information on the study species from other areas in the North Atlantic is used where studies are at a more advanced stage than those in Scotland, and provide important information. These international studies also highlight the large gaps in our present knowledge of the Scottish nearshore environment, and why filling these gaps could be useful.

The section addresses the following questions:

- Do the three whitefish species use Scottish coastal and reef areas?
- Do the characteristics of the coastal or reef environment affect the success of whitefish inhabiting them?
- Does the success of whitefish inhabiting coastal and reef environments affect recruitment into the commercially-exploited stocks?

- Could the study of Scottish nearshore environments and fish communities provide useful management information and even advanced warning of threats to the future state of commercially exploited stocks?
- Could changes in Scottish coastal zone management improve recruitment of whitefish into the exploited stocks?
- What monitoring tools would be most appropriate for the study of Scottish nearshore fish populations?

5.2 Introduction

Juvenile cod and whiting inhabit Scottish nearshore waters during their first year of life. The availability of complex habitat, abundance of suitable prey items, environmental conditions, anthropogenic disturbances, and density-dependent processes have all been shown to affect juvenile gadoid abundance at various spatial and temporal scales. While most of this work has been conducted in other areas of the North Atlantic, sufficient evidence exists for Scotland to show that similar factors are probably important here.

In Scotland, areas such as maerl and eelgrass beds appear to be important as nursery areas for cod, but such habitats are easily damaged by human activities such as trawling. Although damage is known to have occurred around Scotland, we currently lack good data on the extent of habitat loss, and its consequences for cod populations.

Juvenile cod and whiting in the nearshore are connected to adult stocks through their spawning origins, and their destination following migration into deeper water as older fish. These connections are beginning to be understood, but are still unclear in many locations. Established protocols for sampling juvenile gadoids in the nearshore include beach seines and shallow water trawls, although numerous other gears and techniques have been tested. Monitoring programs in Norway and Canada contribute to knowledge about factors affecting year-class strength and species interactions during the gadoid juvenile phase. Understanding these processes and monitoring juvenile cod and whiting abundance in the nearshore provides useful information about the health of fish stocks.

This section will compare what is known about cod and whiting in the nearshore environment and highlight where additional information about the West of Scotland is required. A brief overview of haddock and their presence in coastal waters is also provided, indicating that the coastal environment is not critically important to this species.

5.3 Definitions

The terms 'inshore' and 'nearshore' are used throughout this section, and in the literature are often confused or used interchangeably. For the purposes of this study, 'nearshore' refers to shallow waters less than 30 m in depth, while 'inshore' refers to any waters within 3 nautical miles of the shore. Additionally, the term 'coastal' is used in reference to resident

populations of fish that remain within fjords/sea lochs, bays or other semi-enclosed water bodies throughout the year.

5.4 Cod

5.4.1 Use of Nearshore Habitats

Atlantic cod (*Gadus morhua*) are distributed throughout the North Atlantic Ocean from Cape Hatteras in the United States to the Bay of Biscay in the eastern Atlantic, and north into the Barents Sea. Population dynamics are highly variable, ranging from resident coastal populations to highly mobile offshore populations. Cod in the northern Atlantic spawn from February to June, with peak spawning occurring between March and May, releasing batches of buoyant eggs into the water column over periods of up to several weeks. The eggs float at the surface for approximately 2 months, and then hatch into planktonic larvae (Scott and Scott, 1988). At lengths of 25–50 mm, the pelagic juveniles settle into a demersal lifestyle, either in nearshore areas or on offshore banks (Scott and Scott, 1988; Lough and Bolz, 1989). Throughout the north Atlantic, settlement into nearshore areas occurs from June until October (Gibson *et al.*, 1993; Philippart *et al.*, 1996; Fromentin *et al.*, 2001; Svedang, 2003; Magill and Sayer, 2004; Ings *et al.*, 2008).

In the coastal waters of Scotland, surveys of juvenile cod indicate that the highest densities are found within 60 km of the coast (Figures 5.1 – 5.3), with a negative relationship between depth and juvenile cod abundance (Gibb *et al.*, 2007). Additional surveys specifically focussed on the North Sea indicate that 99% of juvenile cod are found in water less than 26 m in depth (Gibb *et al.*, 2008). This phenomenon of fish size increasing with depth is referred to as Heinke's Law, and is a trend observed in several locations where juvenile cod inhabit the nearshore (e.g. Espeland *et al.*, 2007; Gibb *et al.*, 2007; Knutsen *et al.*, 2007; Gibb *et al.*, 2008). Juvenile cod begin to move out of shallow coastal water in Scotland during their first winter (Gordon, 1990; Magill and Sayer, 2004), and at this time may be vulnerable to Nephrops trawlers as bycatch (Stratoudakis *et al.*, 2001). The carrying capacity of nearshore nursery areas has been identified as a key parameter setting the maximum level of recruitment into cod populations (Heath *et al.*, 2008).

Throughout the geographical range of Atlantic cod, juveniles settle into nearshore areas in the late summer/early autumn, and establish abundance gradients that are negatively associated with depth. These juvenile fish remain in the nearshore from anywhere from 1 to 6 years, at which time they move into deeper water and join adult populations (Gregory *et al.*, 1997; Cote *et al.*, 2001; Cote *et al.*, 2004; Scott and Scott, 1988). Compared to Canada and Norway, knowledge of juvenile cod distributions in the Scottish nearshore is sparse, as is knowledge of environmental factors affecting the timing and magnitude of recruitment to the nearshore, and the potential impact of bycatch on juvenile cod abundances.

5.4.2 Population Structure

Metapopulation analysis has identified several examples of fish stocks that are actually comprised of several sub-populations, indicating that managing fish stocks as single population may not be suitable (Svedang, 2003; Wright *et al.*, 2006b; Heath *et al.*, 2008). Identification of discrete population units and determining the linkages between juvenile and adult aggregations can assist with management policies aimed at protecting critical habitats and formulating other spatially-explicit controls on fishing.

In Scottish waters, metapopulation analysis suggests that semi-isolated populations may exist. Spawning cod sampled in the Clyde Sea area are distinct from those in both the Irish Sea and South Minch when compared using otolith shape analysis, suggesting the presence of resident populations (Galley *et al.*, 2006). Resident spawning populations in the Minch, Clyde Sea area, Inner Hebrides and Irish Sea are also indicated by egg surveys, as well as by tagging studies conducted in the 1960s (Wright *et al.*, 2006a). Otolith sampling suggests most adults in the Clyde, Minch and Inner Hebrides come from local nursery areas, although there does appear to be some mixing of adult fish between the Inner Hebrides and other areas (Wright *et al.*, 2006b). Similarly, age 0 cod in the Clyde Sea area and locations on the northeast coast of Scotland likely come from locally spawned eggs and larvae (Gibb *et al.*, 2007). Together, these data suggest the presence of several populations of cod along the west coast of Scotland that self-recruit using local nursery areas, and have some individuals moving between them.

Across the study area as a whole, age-0 cod were scarce, with numbers exceeding 10 cod km⁻² only being observed within 60 km of the shore. Densities of >100 cod km⁻² were only found in 4.1% of the survey area, but this small area contained 77% of all age-0 cod (Gibb *et al.*, 2007).

5.4.3 Diet in the Nearshore

In the Scottish nearshore, juvenile cod feed on fish (plaice), polychaetes, copepods, mysids, hermit crabs, and other decapod crustaceans (Hawkins *et al.*, 1985; Ellis and Gibson, 1995; Gibson and Robb, 1996). Lack of food availability in Scottish sea lochs over the winter may limit cod productivity in such locations (Hawkins *et al.*, 1985). Juvenile cod in the Scottish nearshore forage closer to shore at night, possibly to avoid larger predators that move into shallow waters after dark to feed (Gibson *et al.*, 1996).

Early in their life history, juveniles feed on zooplankton, before switching to benthic prey, and then fish. While plankton data may give some indication of early food availability (See Section 2), good time series on benthic food and other inshore fish are lacking at likely nursery sites in Scotland. This is unfortunate, as studies in Canada indicate that diet has a significant impact on the likelihood of juvenile cod surviving their first winter. Prey quality and quantity contributed to autumn condition factor, a crucial factor in overwintering success (Grant and Brown, 1999). Once juveniles switch to piscivory, inter-cohort cannibalism

occurs when prey fish are approximately one-third the length of the predator (Grant and Brown, 1998a). This behaviour has important implications for the dynamics of cod populations, and for the habitats chosen by juveniles.

5.4.4 *Habitat Preferences*

On the West Coast of Scotland, maerl beds (Figure 5.4) in Loch Sween support more juvenile gadoids, including cod, than algae-covered rock and gravel (Figure 5.5), suggesting that maerl is an important nursery habitat in the West of Scotland (Kamenos *et al.*, 2004). Rocky subtidal habitat in the Firth of Lorne supports a significantly higher number of juvenile cod than other habitat in the area (Magill and Sayer, 2002). Surveys of the artificial reef in Loch Linnhe show juvenile cod associating with the edge of the reef and the reef itself, suggesting that artificial reefs can increase the amount of suitable habitat for juvenile cod (Sayer *et al.*, 2005). Juvenile cod in the Firth of Clyde associate with shallow, complex substrate characterized by large sediment particle size and high algal abundance (Ware, 2009). In surveys to identify cod nursery areas along the Scottish coast, sheltered areas yielded higher abundances of juvenile cod than exposed sites, similar to results of some studies in the Norwegian Skagerrak (Fromentin *et al.*, 1997; Fromentin *et al.*, 2000; Gibb *et al.*, 2007).

The observation that complex habitat is important to juvenile cod is supported by numerous international studies. Work in the northwest Atlantic highlights the importance of eelgrass for juvenile cod. Overall catches of cod in eelgrass habitat (Figure 5.5) are higher than those in unvegetated habitat (Gotceitas *et al.*, 1997; Anderson *et al.*, 2007), and additional work has examined the nature of this association. Studies of how habitat type affects the success of juvenile cod are much more advanced than those in Scotland, but similar factors are likely to be important here.

In Canada studies have demonstrated differential predation risk between areas of increased complexity and bare substrate, and this is likely to be the reason why juvenile cod associate with eelgrass or cobble (Fraser *et al.*, 1996; Gotceitas *et al.*, 1997). Comparison of habitat types in the nearshore off Nova Scotia show that juvenile cod in eelgrass had the highest growth rates, but juvenile cod associated with cobble substrate and rocky reefs had greater survival (Tupper and Boutilier, 1995). Predation rates were lower in larger eelgrass patches, up to a certain threshold where increased predator density (caused by the attraction of predators to patches with high densities of prey) overcomes the reduction in predator efficiency caused by eelgrass (Laurel *et al.*, 2003a). Similarly, predation risk for age-0 cod is higher at greater depths and in unvegetated areas of the nearshore environment (Linehan *et al.*, 2001).

When found outside of complex habitat (e.g. over sand), juvenile cod tend to aggregate into groups, a common anti-predator strategy, suggesting an increase in perceived predation risk outside of protective habitat (Laurel *et al.*, 2004). The association with complex habitat may have a density-dependent component, as increased numbers of juvenile cod are found

outside of preferred habitat when overall abundances are high (Grant and Brown, 1998b). Brown algae (e.g. *Desmarestia* sp.) can also act as protective habitat for juvenile cod, and supports greater abundances of age-0 fish than areas of crustose coralline algae that have been grazed by sea urchins in coastal Newfoundland (Keats *et al.*, 1987). Older fish (age 2–4) show a preference for coarse or large-grained substrate (i.e. rock and boulder) and areas of high bathymetric relief in depths ranging from 15–120 m (Gregory *et al.*, 1997; Cote *et al.*, 2001; Cote *et al.*, 2004).

In the Norwegian Skagerrak, juveniles are found in greater numbers at sheltered sites in fjords compared to more wave-exposed sites outside of fjords (Fromentin *et al.*, 1997; Fromentin *et al.*, 2000). In contrast to this, another study in the Norwegian Skagerrak found that age-0 cod were larger and more abundant at more-exposed sites, possibly due to different settlement patterns between exposed and protected areas (Lekve *et al.*, 2006). These contradictory results may indicate inter-fjord differences in food availability or environmental conditions.

Age-0 cod in the Swedish Skagerrak utilize eelgrass (*Zostera marina*) beds as nursery habitat, while age-1 fish use both eelgrass and non-eelgrass habitat (Pihl *et al.*, 2006). Based on comparisons of fish assemblages at eelgrass and non-eelgrass sites in the Swedish Skagerrak, Pihl *et al.* (2006) calculated that the loss of 190 m² of eelgrass habitat may have resulted in the loss of 6.3 million juvenile cod recruits each year in the Swedish Skagerrak archipelago. Similarly, long-term fluctuations in juvenile abundance in the Norwegian Skagerrak has been linked to seagrass coverage (Fromentin *et al.*, 1998).

Removal of grazing sea urchins, leading to increased growth of macroalgae, leads to increased juvenile cod abundance (Keats *et al.*, 1987). A similar effect was observed in eastern Newfoundland, where removal of eelgrass at some sites, and addition of artificial eelgrass at others resulted in a decrease of juvenile cod abundance at removal sites, and an increase at sites where artificial eelgrass habitat was added (Laurel *et al.*, 2003b). Laboratory studies simulating the effects of bottom trawling suggest that losses of habitat from bottom trawling could result in increased mortality for juvenile cod (Lindholm *et al.*, 1999).

Trend information on the status of maerl, eelgrass etc. is, unfortunately, not available for Scotland. Even Natura sites have not yet had their second set of surveys following the collection of baseline measurements (D.Donnan, SNH, pers comm.). However, short-term studies of the potential effects of human impacts on these critical habitats have taken place in Scotland. The effects of hydraulic dredging and trawling on maerl beds and other benthic habitats have been examined (Hall-Spencer 1998, Hall-Spencer *et al.* 2003).

In general, juvenile cod associate with complex habitat during their time in the nearshore environment. It is likely that this reduces predation risk. The particular type of complex habitat used varies throughout the species range and includes eelgrass, macroalgae, cobble, rocky reef, maerl and artificial reefs. Based on studies in Norway and Canada (see

above citations), it appears that when complex habitat is lacking, juvenile cod are less abundant and subject to greater predation pressure. This association with complex habitat in the nearshore may determine the location of localized nursery areas. The availability of complex habitat is probably an important determinant of the carrying capacity of these nursery areas, and this important parameter can easily be reduced by impacts on these habitats. As noted above, the carrying capacity of nursery areas appears to be very important in determining the level of recruitment into the adult populations (Heath *et al.*, 2008)

5.4.5 Long Term Trends

Consistent juvenile cod surveys have not been conducted along the West Coast of Scotland. However, the presence of juvenile cod in the nearshore has been noted at several locations during other work. Juvenile cod have been observed at Tralee Beach and Saulmore Point in the Firth of Lorne, the Clyde Sea area, Loch Hourn, Gairloch, Upper Loch Torridon, Loch Sween, Loch Etive and Loch Linnhe (Hawkins *et al.*, 1974; Kislalioglu and Gibson, 1977; Hawkins *et al.*, 1985; Hall *et al.*, 1990; Gibson *et al.*, 1993; Ellis and Gibson, 1995; Gibson *et al.*, 1996; Hall *et al.*, 1996; Magill and Sayer, 2002; Kamenos *et al.*, 2004; Sayer *et al.*, 2005; Gibb *et al.*, 2007; Ware, 2009). Regrettably, despite large amounts of individual studies, there has been no consistent, long-term study of the same locations over several years. Many projects are individual contracts for Marine Scotland or Scottish Natural Heritage, involving one or a few research cruises. Other work has been as part of PhD projects, involving at most three years of fieldwork. As a result there is little consistency between studies in the methods or locations used, making it impossible to detect any trends in populations.

The following section describes potential long term data sources. We examined data from routine monitoring by the Scottish Environmental Protection Agency (SEPA), the Rivers and Fisheries Trusts coastal sampling, and other sources such as recreational fishermen.

5.4.6 Data Sources - Scottish Environmental Protection Agency (SEPA)

SEPA's remit extends to the 3-mile limit, within which they are responsible for the chemical and ecological monitoring required under EU directives. The relevant legislation is the Water Framework Directive (transposed into the Water Environment and Water Services (Scotland) Act, 2003) and its requirement to assess the chemical and ecological status of transitional and coastal waters.

Transitional waters (i.e. estuaries) fail this test if their fish fauna can be described as "A moderate proportion of the type-specific disturbance-sensitive species are absent as a result of anthropogenic impacts on physicochemical or hydromorphological quality elements". Fish populations in transitional waters must be monitored at an interval not exceeding 3 years, but no frequency of survey guidelines are provided for coastal fish. The methods to be used for fish surveys are described as "Relevant CEN / ISO standards when developed", unlike

freshwater invertebrate sampling, where numerous protocols are listed. As monitoring fish populations close to the shore falls within SEPA's remit, their sampling programmes are an opportunity to gather data that could inform the management of cod fisheries.

The existing data include long-term beam-trawl datasets for the inner Clyde (1979–2009, 6 stations, see Figure 5.7, >2 surveys per year), and short datasets (Figure 5.9) for Gareloch (2007–2009, 5 stations, annual), Loch Eil (2005–2009, 3 stations, annual) Ayr and Irvine (1974–1984, 4 stations, usually annual). In 2010, SEPA will be carrying out a detailed survey of the Clyde (inner Clyde and Gareloch) using trawls, beach seines, and fyke nets. Following completion of this survey, survey frequency will decrease to one survey every three years.

Data for the Clyde estuary will be presented here (Figures 5.7 and 5.8), but data for the other study sites are not included in this report. Summaries are to be found in Bellini (2010). It is not clear how most of the data can be used. Issues include the shortness of most of the datasets, use of a beam trawl (not ideal for gadoids, or for rough ground), lack of records regarding start and/or end points for some trawls, preventing estimation of the swept area, and the absence of fish length and weight measurements for most trawls.

The only clear information on whitefish to emerge from the SEPA data is the return of cod and whiting to the Clyde estuary around 2000 after being absent since the survey began in 1979. Few of either species have been observed in the last three years. The first appearance of cod and whiting appears to be associated with the first years in which the dissolved O₂ levels remained above tolerable levels for >90% of the time. The number of oxygen measurements falling below tolerable levels increased slightly from 1999–2006 and this may be why few of the target species were observed from 2007–2010.

Summary

The existing data are not of sufficient length or geographical extent to make conclusions about factors influencing cod, haddock, or whiting on the West Coast. While the return of cod to the inner Clyde is encouraging, this is a specific example where a very severe environmental impact (water pollution) has been reduced, and in no way indicates a general trend across the West Coast.

The proposed intensive sampling to be carried out this year by SEPA, including the use of beach seines, will be a major improvement, but the reduction in sampling frequency to one survey every three years removes virtually any utility that the data might have for fisheries assessment. Small changes in procedure (e.g. recording lengths for each individual fish) would make a large difference to the usefulness of the data.

5.4.7 Data Sources - Rivers and Fisheries Trusts

Some Rivers and Fisheries Trusts (RAFTs) carry out coastal fish sampling in order to study sea trout (*Salmo trutta*) and to assess the impacts of sea lice (Copepoda: Caligidae). Much of this work is associated with, and funded by, the Tripartite Working Group. We investigated the potential of this sampling to provide information on coastal whitefish populations through a meeting with local trusts (Loch Lomond, Clyde, Argyll and Ayrshire) and via e-mail contact with other trusts. Data were provided by Lochaber, West Sutherland and Argyll. The available data are summarised below.

The West Sutherland Fisheries Trust

The West Sutherland Fisheries Trust conducts sampling using beam trawls and sweep nets. Sweep net fishing has taken place since 1998 in Loch Laxford, where cod and “codling” are often caught. Sweep netting has been carried out at Polla (1998–2009), Dionard (1995–2009) and Rhiconich (1999–2000). Young saithe (‘cuddies’, *Pollachius virens*) were usually caught at these locations, with cod/codling occasionally observed at each. The four surveys at Dionard only captured flatfish, and the exact location of this site is unclear.

Beam trawling conducted in 1995 in Loch Laxford (18 tows) captured pollack and occasionally whiting. Large numbers of “juvenile gadoids” were also recorded. Beam trawling in 1995 and 1996 in Loch Eriboll (8 and 6 tows respectively), caught small numbers of cod and pollack (*Pollachius pollachius*). Some “juvenile gadoids” were also caught.

Argyll Fisheries Trust

Argyll Fisheries Trust conducted beach seine sampling at ten locations in 2009. These surveys caught some gadoids (cod, pollack, and ling - *Molva molva*) off Arran, but not at the other nine locations.

Lochaber Fisheries Trust

Lochaber Fisheries Trust conducts ongoing beach seining at seven locations, including year round sampling at Camas na Gaul and Kinlocheil. The other sites are surveyed once or twice between mid-April and early July. The surveys have been carried out since 2002, but do not catch any of the study species. The intensively surveyed sites are either in Loch Linnhe opposite Fort William or still further inland to Loch Eil.

Summary

The available data are not sufficient to show trends, though the longer sweep-net time-series and short beam-trawl series carried out by West Sutherland Fisheries Trust at least provide some information. Some length information is available from the West Sutherland surveys, but these are unfortunately only for 230 fish from six tows in Loch Laxford.

These studies show that Fisheries Trusts observe and record the presence of target species during their activities, but on a sporadic basis, and usually without recording fish length. Their existing sampling is not sufficient to provide the data required to assess nearshore whitefish populations, but there is potential to collaborate with the RAFTs to carry out sampling which meets this requirement.

Other efforts to collect trend data included contacting fishing magazines and associations. More details of these efforts are available from Bellini (2010). No quantitative data were obtained, though there was much qualitative evidence of greater abundances of inshore cod in previous years

5.4.8 How Useful is Nearshore Sampling to the Management of Cod Stocks?

As noted above, Scotland currently lacks a consistent nearshore sampling programme for cod. Such programmes do exist in other countries, and the following material is included in order to illustrate what can be achieved.

In Canada, temporal trends in juvenile cod populations have received a great deal of attention, with a focus on detection of year-class strength driving much of nearshore monitoring programmes. Comparison of year class strength determined from surveys at various life history stages of Atlantic cod suggest that abundance during the early juvenile phase is the most reliable predictor of recruitment to fishery stocks (Campana *et al.*, 1989; Bradford, 1992; Helle *et al.*, 2000; DFO 2009).

In the northwest Atlantic, the collapse of the northern cod stocks was preceded by decreasing year class strength from 1989–1991. However, this trend was only detected through retrospective analysis of age-2 and 3 fish from those cohorts (Anderson *et al.*, 1995). The implication is that monitoring numbers of juveniles could have provided an early warning of the impending collapse of the stock. This finding provided impetus for establishment of nearshore monitoring as an integral part of the fisheries management system.

Comparisons of different nearshore surveys conducted in Newfoundland indicate that more accurate indications of relative year class strength are obtained when relatively few stations are sampled at regular intervals throughout the settlement period, compared to a large number of stations sampled inconsistently (Ings *et al.*, 1997). Since 1995, relative year class strength has been monitored using nearshore surveys of age-0 and age-1 cod in a fjord in Newfoundland (Gregory *et al.*, 2002; Gregory *et al.*, 2004). This data set has yielded considerable information about the patterns of settlement, genetic relationships between recruits and adult stocks, and relative year class strength, and is now used in stock assessments of the northern cod stock complex (Beacham *et al.*, 2000; Ings *et al.* 2008, DFO 2009).

Long-term beach seine surveys in the Norwegian Skagerrak have led to a great deal of information on nearshore recruitment patterns in cod and the factors that may affect recruitment. Juvenile cod abundance decreased during the years 1919 to 1939, increased from 1945 until 1965, and then declined from 1965 to 1995, with a small period of higher abundance from 1972 to 1975 (Fromentin *et al.*, 1997). Throughout this time, periodicity was found in cod recruitment, on the scale of approximately 2–2.5 years, suggesting density dependent inter-cohort competition (Fromentin *et al.*, 1997; Bjørnstad *et al.*, 1999b; Bjørnstad *et al.*, 1999a; Stenseth *et al.*, 1999; Fromentin *et al.*, 2000; Fromentin *et al.*, 2001). Density-dependent processes and their potential effects on fish recruitment are highlighted by the “concentration hypothesis” (Beverton, 1995; Iles and Beverton, 2000). This suggests species that concentrate as juveniles relative to their distributions as adults are more likely to undergo density-dependent mortality than those species which do not concentrate during the early life history (Beverton, 1995; Iles and Beverton, 2000).

Pollution may interact with density-dependent effects to affect populations at local scales. One heavily polluted sampling station in the Norwegian Skagerrak showed more inter-annual variability in abundance of age-0 cod and stronger direct density-dependent mortality than other sites studied, suggesting that high levels of pollutants may disrupt normal population dynamics (Fromentin *et al.*, 2001).

Results from the International Young Fish Survey and the Dutch Groundfish Survey of the North Sea indicate greater densities of young cod in locations with low temperatures and low salinities (Heessen, 1993). These results together suggest that both local environmental factors and density-dependent effects can affect recruitment variability from year to year.

Even with density-dependent processes reducing variability in recruitment from year-to-year, considerable variability does exist, and abnormal recruitment years have the potential to influence stock structure. In a population of coastal cod in northern Norway, sampling from 1989 to 1994 showed the presence of a very strong year class of recruits from 1987 (Berg and Pedersen, 2001). This year class dominated catches until the end of the study period, at which point that year class had reached the age of 7 years (Berg and Pedersen, 2001).

In the autumn of 1988, a toxic algae bloom spread throughout upper 20 m of the water column in the Norwegian Skagerrak, killing approximately 60% of age-0 cod and resulting in additional mortality of that year class in 1989 (Chan *et al.*, 2003b). In the six years following the bloom, an alternating pattern of strong and weak year classes was detected in the affected area (Chan *et al.* 2003b). The authors suggested that this pattern was due to reduced inter-cohort cannibalism on the 1990 year class, resulting in a strong 1990 year class that then exerted above-average predation pressure on the subsequent year class, and so on until the effect was dampened over time (Chan *et al.*, 2003b). Overall, the algal bloom was believed to have no significant effect on stocks (high and low years averaged out), but resulted in higher variability in recruitment throughout the period following the bloom (Chan *et al.* 2003b). This variability would have been inexplicable without the nearshore data, and could have led to inappropriate and futile management efforts.

5.4.9 Sampling Methods

Methods appropriate for the study of coastal marine fishes are described in (Davies *et al.*, 2001). Such methods have been widely used in the UK, including the West of Scotland. The methods described vary according to the bottom type involved, but include SCUBA, fyke nets and beach seining. Surveys of juvenile cod abundances have been conducted using beach seines, otter trawls, beam trawls, SCUBA surveys, hook and line fishing, underwater cameras, fyke nets and tagging (Gibson and Robb, 1996; Philippart *et al.*, 1996; Fromentin *et al.*, 1997; Cote *et al.*, 1998; Ellis *et al.*, 2000; Cote *et al.*, 2002; Magill and Sayer, 2002).

Comparison of beam trawls and beach seines for sampling epibenthic crustaceans and fishes in seagrass in Australia indicate that beach seines are more effective at sampling both types of animals (Guest *et al.*, 2003). Comparisons of day and night sampling in the same study indicate the importance of night sampling, and the authors recommend either a combination of day and night sampling, or night sampling only (Guest *et al.*, 2003).

SCUBA surveys in eastern Newfoundland recorded an increase in the abundance of older juvenile (age 1 and 2) cod close to the shore at night, suggesting that night sampling may provide a more reliable representation of the nearshore fish community than day sampling alone (Keats 1990). One study in Newfoundland compared estimates collected using beach seines of various sizes, gillnets, trawls, hook and lines, and SCUBA observations (Methven and Schneider 1998).

Trawls and beach seines proved most effective at sampling fish less than 200 mm in length, and calibrating catches across gears was not possible, suggesting that consistent sampling with a single gear type is necessary for accurate temporal and spatial comparisons (Methven and Schneider 1998). Consistent sampling of several locations by beach seine have been used in Norway for the past 90 years to determine year class strength, diet, habitat preferences, the effect of stocking programs and community dynamics of juvenile cod populations (Tveite 1971, Fromentin *et al.* 2000, Chan *et al.* 2003a).

Taken together, these results suggest that long-term sampling using beach seines and shallow water trawls in a combination of day and night sampling at a relatively small number of sites is likely to provide the most accurate assessment of inter-annual variation in juvenile cod abundances. In 2001, ICES recognized the need for more detailed monitoring of the settlement phase of gadoid life histories, owing to the importance of this life history period in determining year class strength (ICES 2001).

Nearshore sampling using beach seines and trawls are established forms of monitoring juvenile cod abundances, though these will clearly not be suitable for all of juvenile cod's preferred habitats. A range of other methods are described in (Davies *et al.*, 2001). Night sampling is advisable, if logistically possible. Consistent sampling of the same sites each year is necessary to generate accurate indications of relative year-class strength.

5.5 Whiting

5.5.1 Use of Nearshore Habitats

Whiting (*Merlangius merlangus*) are distributed throughout the north-eastern Atlantic Ocean, North Sea, Baltic Sea, and to a limited extent in the northern Mediterranean Sea (Cohen *et al.*, 1990). Whiting in the North Sea and along the west coast of Scotland spawn between February and June at depths ranging from 36 m to 80 m (Bowers, 1954; Nagabhushanam, 1964; Gordon, 1977a; Cooper, 1983). In the two to three months after spawning, the eggs hatch into planktonic larvae that feed on zooplankton in the water column (Nagabhushanam, 1964).

Following metamorphosis and a short pelagic juvenile phase (Nagabhushanam, 1964), age-0 whiting settle into a demersal lifestyle at lengths of 30–60 mm (Nagabhushanam, 1964; Moore and Moore, 1976; Gordon, 1977a, b; Gibson and Ezzi, 1987). The chemical composition of whiting otoliths changes through their life, probably indicating differences in the chemistry of the water in which they were living at the time when the otolith material was laid down. This finding suggests that juvenile and adult habitats are spatially separated (Ware, 2004). Additional studies suggest young whiting may actively migrate to find suitable areas for settlement (Gordon 1977a, Gordon 1990).

It is as demersal juveniles in their first year of life that the Scottish nearshore environment plays a significant role in the life history of whiting, with some juvenile whiting settling in nearshore environments, and some settling in offshore areas (Nagabhushanam 1964). Juvenile whiting settle in the nearshore from June to December, then move offshore as larger age-1 fish from January to the following summer (Hall *et al.*, 1990; Gibson *et al.*, 1996; Philippart *et al.*, 1996). Hislop (1996) suggested that whiting do not have specific nursery areas. However, the studies reviewed in the present report suggest that nearshore areas may fulfil this function for whiting to some extent (Figure 5.10).

Whiting in the Lynn of Lorne in Scotland migrate into deeper water at lengths of approximately 165 mm (Gibson and Ezzi, 1987). When juvenile whiting move into deeper water during their second summer they are large enough to be caught in mixed-demersal and Nephrops trawl fisheries (Stratoudakis *et al.*, 2001). Whiting may make up to 40% of fish discarded from Nephrops trawlers, with a mean discard length of 19 cm (Stratoudakis *et al.*, 2001). This suggests that juvenile whiting are subject to fishing mortality as bycatch in their first year (Stratoudakis *et al.*, 2001), prior to maturation and spawning at age 2 (Cooper, 1983). Whiting begin to reach the minimum landing size at approximately age 2, and fluctuations in recruitment have almost immediate effects on the fishing industry (Howard *et al.*, 1987).

Summary

Whiting settle into nearshore areas in the late summer and early autumn as juveniles 30–60 mm in length. Whiting along the Scottish coast leave the nearshore at smaller sizes than

juveniles in Norway, and may be subject to fishing mortality as bycatch in their first year. Additional information is needed regarding possible environmental factors affecting the timing and magnitude of nearshore recruitment, and the distribution of whiting in the nearshore along the West Coast of Scotland.

5.5.2 *Population Structure*

Off the west coast of Scotland, juvenile whiting in the Sound of Sleat and Firth of Lorne show no differences in otolith elemental composition, suggesting they originate from the same parental stock (Ware, 2004). However, juvenile whiting from these two areas differ significantly from those in the Clyde Sea area, suggesting that the Clyde whiting are from a different parental stock (Ware, 2004). Juvenile whiting in the Clyde Sea area appear to originate from Irish Sea whiting stocks whose eggs and larvae are carried into the Clyde Sea area by currents (Ware, 2004). It is unknown whether juveniles from the Clyde Sea area return to the Irish Sea as adults. However, tagging studies on adults indicate minimal mixing between the Clyde Sea area whiting and whiting stocks in other locations (Hislop, 1986). Given the high degree of fishing pressure exerted in the Clyde Sea area, it is also possible that tagged whiting have been captured by the fishery before having a chance to migrate from the area (Hislop, 1986). It has also been suggested that a resident population of whiting is present in Loch Etive and may be responsible for the juveniles in that area (Gordon, 1977a; Gordon, 1990). However, additional tagging and genetic studies are required to test this hypothesis.

Summary

Whiting in the nearshore appear to come from offshore stocks or a combination of offshore and inshore stocks, although little is known about potential whiting metapopulations. In the West of Scotland, whiting appear to come from two separate sources, hypothesized to be the Irish Sea and another parental stock. Genetic analysis is required to confirm the linkage between the Irish Sea whiting and juveniles in the Clyde Sea area, and to identify the source of juveniles on the northwest Scottish coast. Identification of spawning locations, in addition to tagging and genetic work, is required to elucidate potential metapopulations of whiting on the west coast of Britain, and the degree of mixing between them animals using each spawning location.

5.5.3 *Diet*

Along the west coast of Great Britain, small whiting (less than 100 mm) are generalist predators, feeding on a wide range of food similar to their Norwegian counterparts, including copepods, amphipods, mysids, and krill (Nagabhushanam, 1964; Gordon, 1977b; Gibson and Ezzi, 1987). Whiting may shift their prey intake depending on food availability. Dominant prey items for age-0 whiting in the Severn estuary are brown shrimp (*Crangon* spp.), gammarid amphipods and gobies (Moore and Moore, 1976), and the authors

concluded that it was lack of availability of other, more-preferred items that resulted in the predominance of these prey species in gut contents.

Turbidity does not appear to affect the ability of age-0 whiting to catch prey: fish captured in turbid environments do not show increased frequencies of empty stomachs (Moore and Moore, 1976).

At a length of approximately 100 mm, whiting along the coast of Great Britain switch to piscivory, specifically feeding on clupeids (herring), sandeel, sprat, Norway pout, poor cod, gobies, shrimp, cuttlefish and flatfishes (Nagabhushanam, 1964; Gordon, 1977b; Patterson, 1985; Gibson and Ezzi, 1987; Hall *et al.*, 1990; du Buit, 1991; Ellis and Gibson, 1995).

Summary

Juvenile whiting in the nearshore are generalist predators, and may compete with juvenile cod for prey. Turbidity does not affect their prey capture ability and older piscivorous whiting may have a negative effect on abundances of other commercially valuable fish species such as herring. Juvenile whiting probably undergo diel migrations for food and to avoid predation. Additional information is needed to determine the potential effect of whiting on other fish stocks along the coast of Scotland.

5.5.4 Habitat Preferences

Habitat requirements for juvenile whiting have not been well studied. However, the limited data available suggest that complex habitat plays a role in minimizing predation risk. Laboratory studies of whiting ranging from 20 to 26 cm in length show that in the absence of a predator stimulus, whiting prefer sand habitat (Atkinson *et al.*, 2004). When given a predator stimulus, larger whiting showed no alteration in their preference, while smaller whiting increased their preference for emergent foliose bryozoan habitat (Atkinson *et al.*, 2004). Field studies of fish and eelgrass distributions in Sweden indicate that juvenile whiting may use vegetated coastal habitat as a nursery area (Pihl *et al.*, 2006). Juvenile whiting sampled along 210 km of coastline in the Norwegian Skagerrak were more abundant at sheltered sites within fjords compared to more wave-exposed sites outside fjords (Fromentin *et al.*, 1997).

Reefs may provide habitat for whiting, although direct observations *in situ* provide mixed evidence for this. In SCUBA surveys of nearshore reefs located at the mouth of Loch Etive and off Great Cumbrae in the Firth of Clyde, only five juvenile whiting were observed in the vicinity of the reefs (Magill and Sayer, 2002). Underwater camera surveys of a subtidal reef in the Lynn of Lorne did not detect the presence of juvenile whiting (Nickell and Sayer, 1998), and whiting were also not observed in earlier surveys of the rocky sublittoral environment off the Isle of Cumbrae (Sayer *et al.*, 1994). In contrast, in SCUBA surveys of an artificial reef in Loch Linnhe, whiting were recorded around the outer edges of the reef (Sayer *et al.*, 2005). It should be noted that the underwater camera surveys conducted by

Nickell and Sayer (1998) focussed on a crevice in the reef, not the edges, which may partially explain the absence of whiting in that particular study.

Various foraging and predator avoidance behaviours have been described for juvenile whiting in nearshore waters. Juvenile whiting in the Firth of Lorne, Scotland, undertake diel migrations, moving offshore during the day to minimize the risk of attack from avian predators (Gibson *et al.* 1996). Age-0 whiting may also undergo diel vertical migrations in the water column, moving closer to the surface during the night and retreating to the bottom during the day (Gordon 1977a). Bailey (1975) found that only older whiting underwent diel migrations in the water column; however, that study was conducted further offshore, and may indicate behavioural differences between pelagic and demersal juveniles.

Summary

Juvenile whiting use complex habitats as refuge from predators. Little information is available on specific habitat associations of juvenile whiting, but seagrass, reefs, and sheltered areas appear to be preferred habitat for juvenile whiting. Along the coast of Scotland (and throughout the range of this species), more information is needed about whiting habitat preferences, and factors that affect habitat associations.

5.5.5 Long Term Trends

No study has conducted a widespread survey of the West Coast of Scotland to determine the distribution of young whiting in the nearshore (SEPA data for whiting are provided alongside the data for cod in Figure 5.10). However, several studies of nearshore fish communities have examined the biology of whiting, or noted the presence of this species.

Juvenile whiting have been found in a depth range 1–90 m at Ardanish Bay, the Sound of Sleat, Tiree Passage, Loch Linnhe, the Firth of Lorne, Loch Spelve, Loch Etive, Loch Sunart, the Clyde Sea area, and Bloody Bay, Isle of Mull (Gordon, 1977a; Cooper, 1983; Ware, 2004, 2009). Tralee Beach in the Firth of Lorne is the site of several studies on nearshore fish communities and has consistently been shown to support juvenile whiting populations (Gibson *et al.*, 1993; Ellis and Gibson, 1995; Gibson and Robb, 1996; Gibson *et al.*, 1996).

In addition to juveniles, Loch Etive appears to also contain a significant number of adult whiting in deeper areas, indicating that there may be spawning occurring within the loch (Gordon, 1977a, b; Gordon, 1990). While many of the sites listed above recorded the presence of older (mainly age 1, occasionally age 2+) whiting, the Firth of Lorne is notable in that only age-0 whiting were found there, suggesting this location may serve exclusively as a nursery area (Gordon, 1977a).

Beam-trawl surveys along the west coast of Scotland were used to calculate relative whiting year class strength from 1969 to 1973 and identify differences in populations and fish behaviour at different locations (Gordon, 1977a). Overall, the 1969 year class was poor, the

1970 year class was average, the 1971 and 1972 year classes were very strong and the 1973 year class was very weak (Gordon, 1977a). In Loch Etive, the year-class variation was considerable from 1969 to 1973. Gordon (1977a) noted that the strong 1971 year class continued to dominate the catches at that location well into 1973, a phenomenon likely due to the comparatively weaker 1972 year-class only appearing briefly in September and October of 1972 (Gordon, 1977a). Beam-trawl surveys in the Forth estuary have also been used to calculate relative year class strength from 1981 to 1988, with the strongest to weakest cohorts as follows: 1984, 1981, 1987, 1988, 1983, 1985, 1986, and 1982 (Elliott *et al.*, 1990).

Salinity appears to have a significant effect on juvenile whiting abundance. Studies of fish captured on cooling water intake screens of power stations in the Thames estuary from 1980 to 1989 show increased juvenile whiting abundance associated with high salinity levels and large amounts of suspended solids in the water column (Araujo *et al.*, 2000). Sampling of the same system from 1977 to 1992 confirmed that whiting abundance was correlated with salinity. However, there was a stronger, negative relationship between abundance and water temperature (Power *et al.*, 2002). Similar sampling in the Bristol Channel showed a positive relationship between salinity and whiting catches, and specifically there was a substantial reduction in catches during the winter of 1976–77, when salinity fell due to abnormally high rainwater run-off (Potter *et al.*, 1988). In the Severn estuary, whiting were the most abundant gadoid from 1972 to 1977, and apparently moved out of the area entirely when salinity dropped below 10‰ (Claridge and Potter, 1984).

The effects of pollution on fish populations are not fully understood, but it is likely that pollutant run-off from land will be detrimental for fish that spend at least part of their life history in nearshore environments. Contamination of adult whiting gonads by DDT, PCBs and chlorinated hydrocarbons has a negative effect on the hatching success of eggs (von Westernhagen *et al.*, 1989). The levels of pollutants examined were equal to that found in herring and flounder, but whiting exhibited a greater negative response, suggesting they may be more susceptible to pollutants (von Westernhagen *et al.*, 1989). This type of pollution poses a direct threat to fish stocks by compromising the reproductive success of inshore fish populations (e.g. possible resident whiting in Loch Etive; Gordon, 1977a; Gordon, 1990).

Summary

Various studies provide indications of year-class strength throughout the range of whiting, but consistent monitoring of juvenile whiting abundance occurs only in the Norwegian Skagerrak. Whiting recruitment varies across spatial and temporal scales, and factors affecting recruitment have only been studied in individual systems. Factors identified as having an effect on whiting recruitment are salinity, temperature, the NAO index, predator abundance, and chlorophyll levels. Pollutants may also have a negative effect on the reproductive ability of adult whiting. These studies have been conducted throughout the North Sea, and currently there is no information on factors affecting whiting recruitment along the west coast of Scotland. Monitoring studies showing variability in recruitment at

large and small spatial scales suggest that recruitment trends on the west coast of Scotland cannot be inferred from examination of North Sea data, and that specific monitoring programs are necessary.

There have been a number of studies examining various aspects of whiting biology in Scottish waters. However, no studies have conducted a widespread survey of juvenile whiting distribution and abundance in West of Scotland nearshore waters. More information is required on spatial and temporal distributions of juvenile whiting in order to conduct further analysis on factors affecting recruitment.

5.5.6 Sampling Methods

Whiting are sampled using beach seines, beam trawls, fyke nets, SCUBA surveys and otter trawls. Beach seining during the night has yielded higher whiting catches than the same sites sampled during the day (Nagabhushanam, 1964), suggesting that night sampling, or a combination of night and day sampling, may provide a better estimate of juvenile whiting than day sampling alone.

Summary

Sampling in the nearshore with beach seines and shallow water trawls, using a combination of night and day sampling is advisable for gaining the most accurate estimates of whiting abundance on the Scottish west coast.

5.6 Haddock

5.6.1 Spawning and Juvenile habitats

Haddock (*Melanogrammus aeglefinus*) spawn from January to May, forming large aggregations in mid-water and producing large quantities of planktonic eggs that float at the surface for up to 90 days (depending on temperature) before hatching (Bailey, 1975; Scott and Scott, 1988; Olsen *et al.*, 2010). Planktonic larvae metamorphose into pelagic juveniles, which eventually settle to the sea bed at lengths of 40–80 mm, although juveniles can sometimes be found throughout the water column (Bailey, 1975; Fahay, 1983; Olsen *et al.*, 2010). Haddock are not believed to have specific nursery areas (Hislop, 1996), and juveniles and adults can occur sympatrically (Campana *et al.*, 1989, Marshall and Frank, 1999b). The period encompassing the late planktonic stage and early settlement phase appears to exert a large influence on year-class strength (ICES, 2001).

Haddock are found predominantly in shelf waters, in depths from 50 to 250 m (Bergstad *et al.*, 1987; Scott and Scott, 1988; Campana *et al.*, 1989; Daan *et al.*, 1990; Albert, 1994; Marshall and Frank, 1995, 1999a; Olsen *et al.*, 2010). Adult haddock can also be found in inshore areas, including sea lochs, generally in water depths greater than 30 m, but sometimes in shallower depths (Sayer *et al.*, 2005; Mente *et al.*, 2008; Ware, 2009; Olsen *et*

al., 2010). Juvenile haddock have been found along the Swedish Skagerrak coast, but in much lower abundances than offshore areas (Svedang, 2003). Along the Norwegian coast of the Barents Sea, young (1–2 year old) haddock have also been observed close to the coast, but their presence in the nearshore is inconsistent from year to year (Olsen *et al.*, 2010).

Spawning on the west coast of Scotland occurs near the northern Outer Hebrides. However the degree of fidelity to spawning locations in haddock is unknown (Gibb *et al.*, 2004). Otolith microchemistry suggests adult populations comprise individuals that spent their early life in several different areas, suggesting significant dispersal in this species (Wright *et al.* 2010). Adult haddock in offshore West of Scotland waters appear to be derived from inshore juveniles in the West of Scotland, and there appears to be no metapopulation structure in this area (Wright *et al.*, 2010).

Summary

Water currents play a major role in the distribution or concentration of eggs and larvae following spawning. In some locations, these currents serve to facilitate self-recruiting populations, while in others they distribute eggs and larvae over vast distances, and result in populations with widespread juveniles. Linking adult and juvenile populations is in the early stages, and specific nursery areas have not been conclusively linked with adult populations. Haddock in the West of Scotland do not exhibit metapopulation structuring, suggesting that the effects of poor recruitment may be apparent at the stock level (in contrast to cod, where individual populations may be affected to different degrees by recruitment variation).

5.6.2 Habitat Preferences

On Georges Bank in the northwest Atlantic, juvenile haddock show a preference for pebble and gravel habitat, a habitat type which is similar to their body colouration, possibly affording some protection from predators through crypsis (Lough *et al.*, 1989). Surveys of an artificial reef on the west coast of Scotland show the presence of haddock (size unknown) associating with the reef (Sayer *et al.*, 2005).

Young haddock may fall prey to cod, pollack (*Pollachius pollachius*) and hake (*Merluccius merluccius*), while seals are a major predator of adults (Scott and Scott, 1988). Studies in the Barents Sea suggest that cod predation on young haddock has the potential to affect recruitment, but the exact magnitude of such an effect has not been quantified (ICES 2008).

5.6.3 Use of Nearshore Habitats

The presence of juvenile haddock has been noted at several nearshore locations along the west coast of Scotland, albeit sporadically and in no great numbers. Several studies have noted the presence of haddock at Tralee Beach in the Firth of Lorne. In the summer of 1990, eleven juvenile haddock were captured in beach seines and beam trawls at depths ranging from 0.5 m to 5 m (Gibson and Robb, 1996). Two individuals were captured in night sampling at Tralee beach in the summer of 1992, but no haddock were found in the 1993 samples at the same location and time of year (Ellis and Gibson, 1995).

A total of 17 haddock were captured in late summer, autumn, and winter at depths ranging from 10 m to 18 m in Gairloch using a variety of gears, but the lengths of these fish are unknown (Hall *et al.*, 1990). Haddock were caught in Irvine Bay and the Lynn of Lorne in 1978–79 and 1975–76, respectively, but not in sufficient numbers to include in community composition analysis (Nash and Gibson, 1982).

Adult haddock have been observed in inshore locations along the West Coast of Scotland throughout the year, and have historically comprised a large proportion of the inshore fishery in the Clyde Sea area (Hislop, 1986). Significant abundances of haddock have been sampled in Lochs Hourn, Kishorn, Duich and Nevis (Mente *et al.*, 2008). The presence of haddock has been noted at several locations in the Clyde Sea area and the Firth of Lorne, in depths ranging from 19 m to 60 m (Ware, 2009). Spawning aggregations in the North and South Minch have been identified and supply large areas of the West of Scotland with eggs and larvae (Wright *et al.*, 2010).

Summary

Haddock are located in offshore waters along the West Coast of Scotland. Adults are also found in some inshore locations, but rarely in shallow nearshore waters. Juveniles are dispersed over large distances, including, rarely, in nearshore areas, and then in small numbers only.

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5.8 Figures

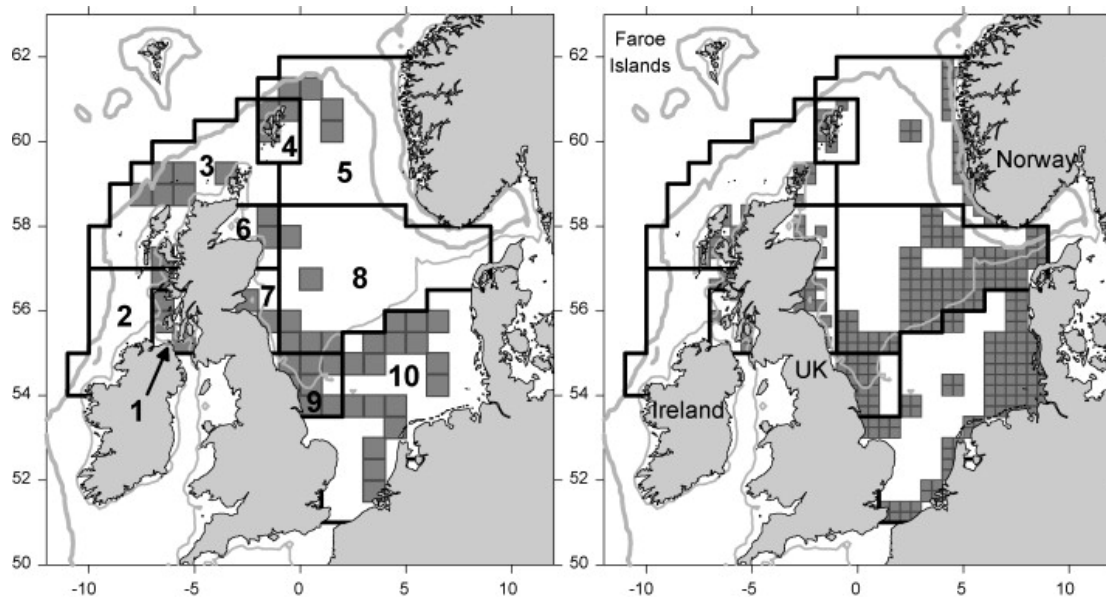


Figure 5.1: A) spawning; and B) nursery areas for Atlantic cod (*Gadus morhua*) (Heath *et al.*, 2008).

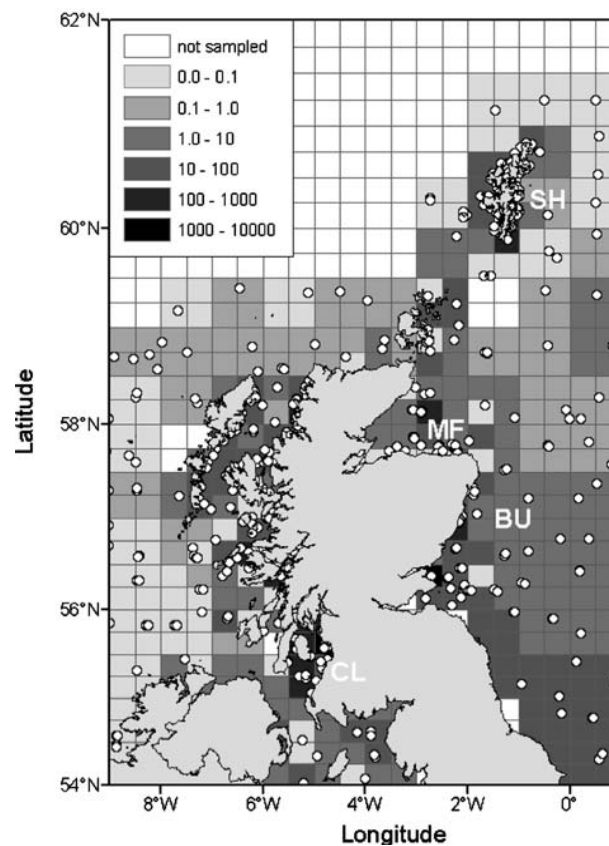


Figure 5.2: Distribution of age-0 Atlantic cod (*Gadus morhua*) around Scotland. Circles mark the trawl stations, while the darkness of the grid squares indicates abundance in fish km^{-2} (Gibb *et al.*, 2007). SH Shetland, MF Moray Firth, BU Buchan, CL Clyde.

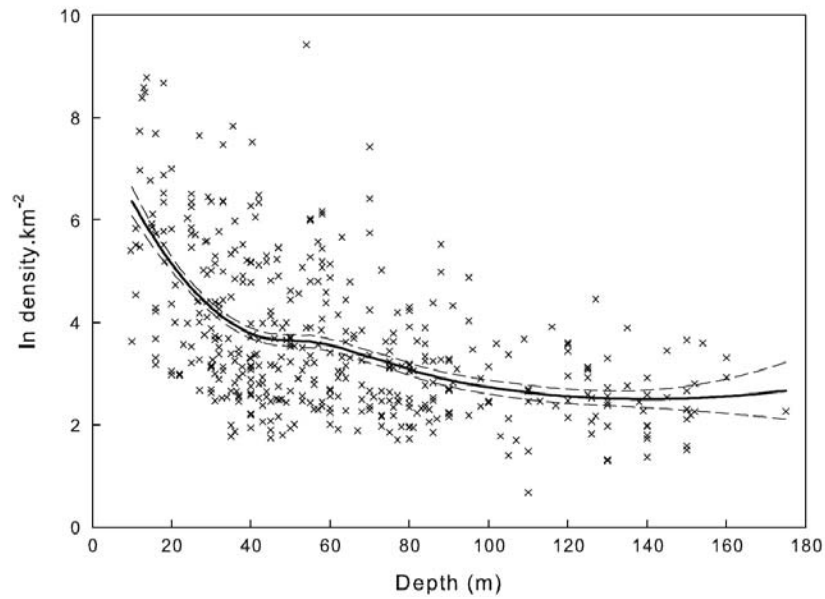


Figure 5.3: Relationship between depth and the abundance of age-0 cod (crosses). Solid line indicates the mean (LOESS smoother), while the dashed lines are the standard error (Gibb *et al.*, 2007).



Figure 5.4: On the West Coast of Scotland, intact maerl beds provide important habitat for juvenile fish such as cod. The bare sand area running from the bottom right corner of the frame shows damage to the maerl bed (in this case from anchor chains). The loss of habitat complexity is likely to leave juvenile cod vulnerable to predation. Photo by Dr N Kamenos, University of Glasgow.

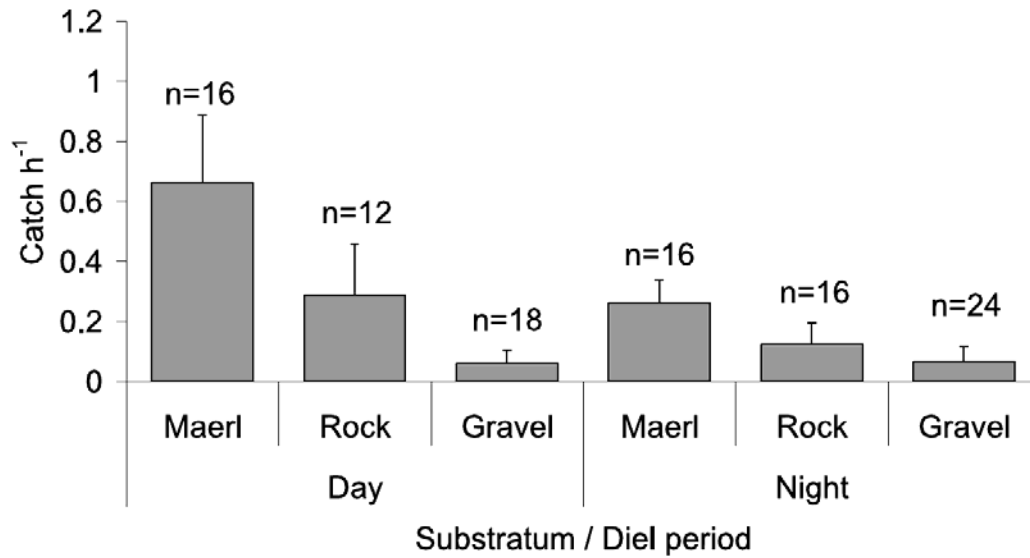


Figure 5.5: The bar chart shows the rate at which juvenile cod were caught by fyke nets used in areas of maerl, bar rock or gravel. Error bars are 95% CI. Significantly more cod were caught in maerl than in either of the other habitat types. (Kamenos *et al.*, 2004).



Figure 5.6: Maerl and eelgrass from the West Coast of Scotland. Both provide important habitat for juvenile fish, but both are easily damaged ©S.Scott/MNCR/SNH.

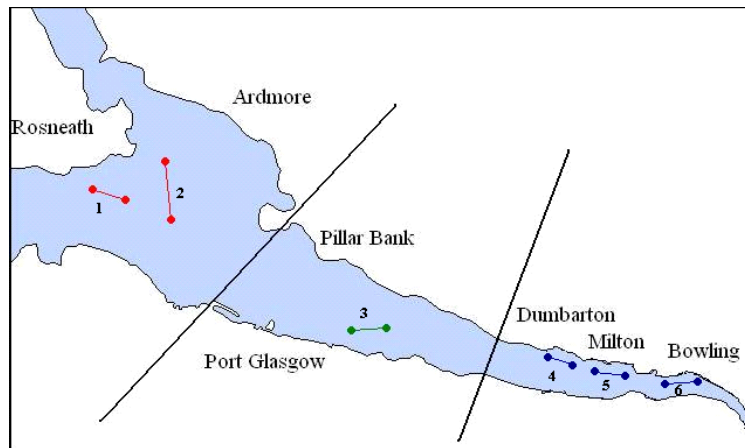


Figure 5.7: SEPA trawl sampling locations in the inner Clyde Estuary. Individual trawl locations are Rosneath (1), Ardmore (2), Pillar Bank (3), Rock (Dumbarton) – Petty Roy (4), Crannog – Milton (5), and Dunglass – Bowling (6).

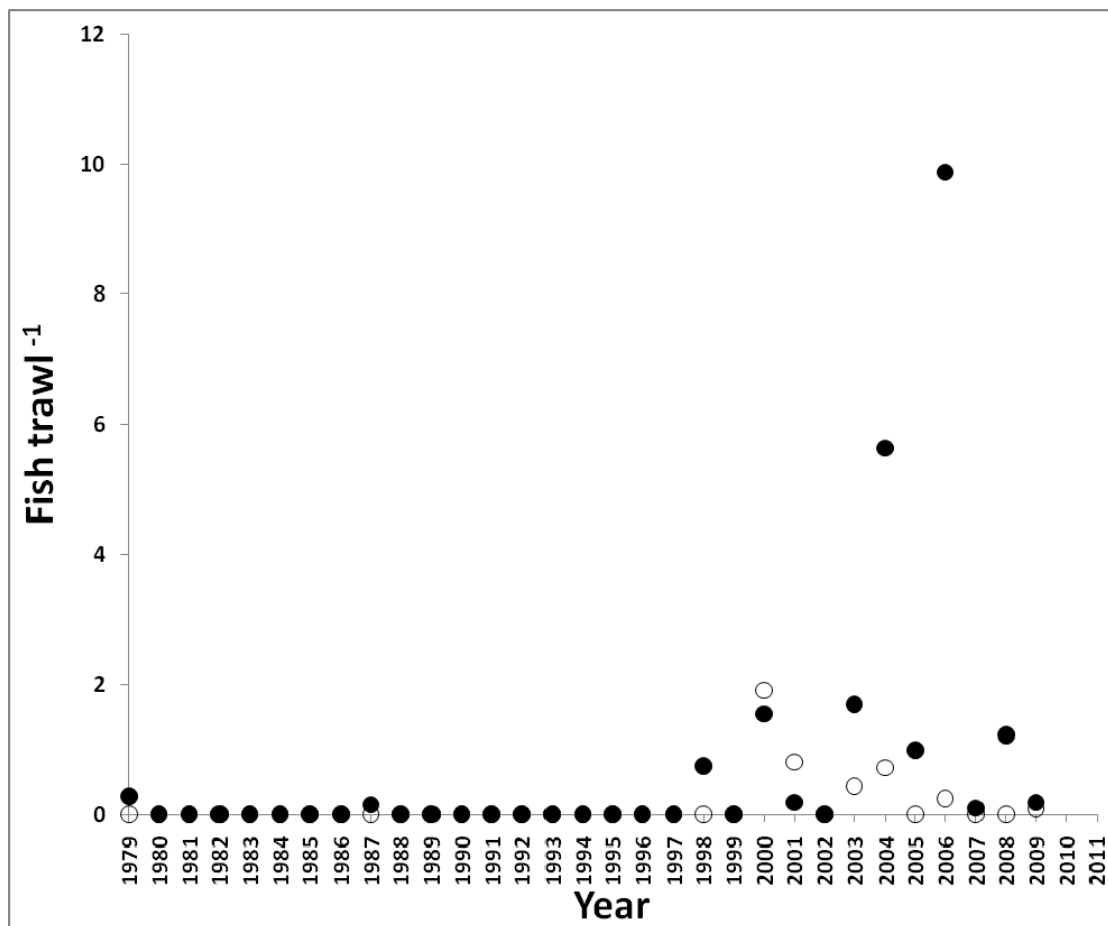


Figure 5.8: The number of cod (solid points) and whiting (open points) caught per trawl in the inner Clyde Estuary. Trawls were of varying or unspecified length, so it is not possible to calculate a measure of catch per unit effort. Two sites (Rosneath and Ardmore) were only surveyed in 2008 and 2009.

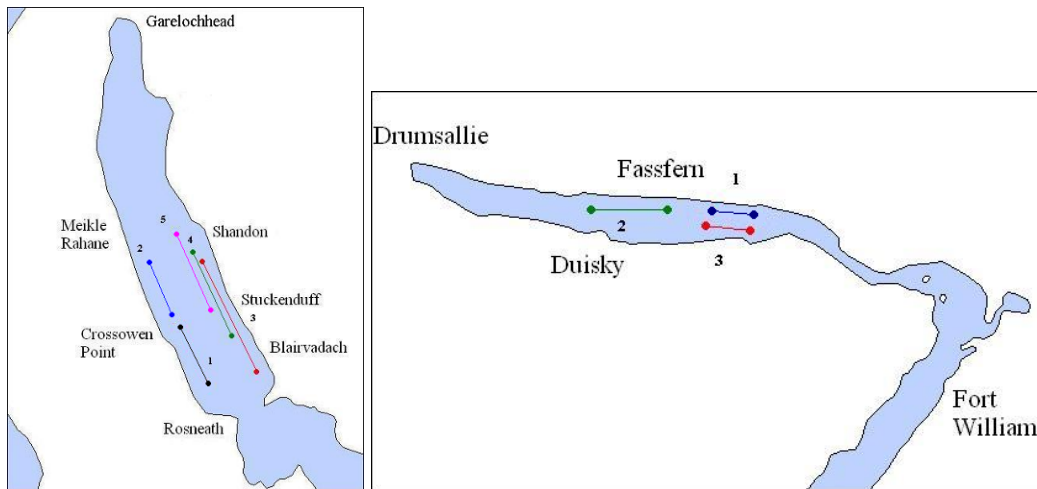


Figure 5.9: SEPA's other routine sampling sites are now at Gareloch and Loch Eil. These datasets run from 2007 and 2005 to present respectively.

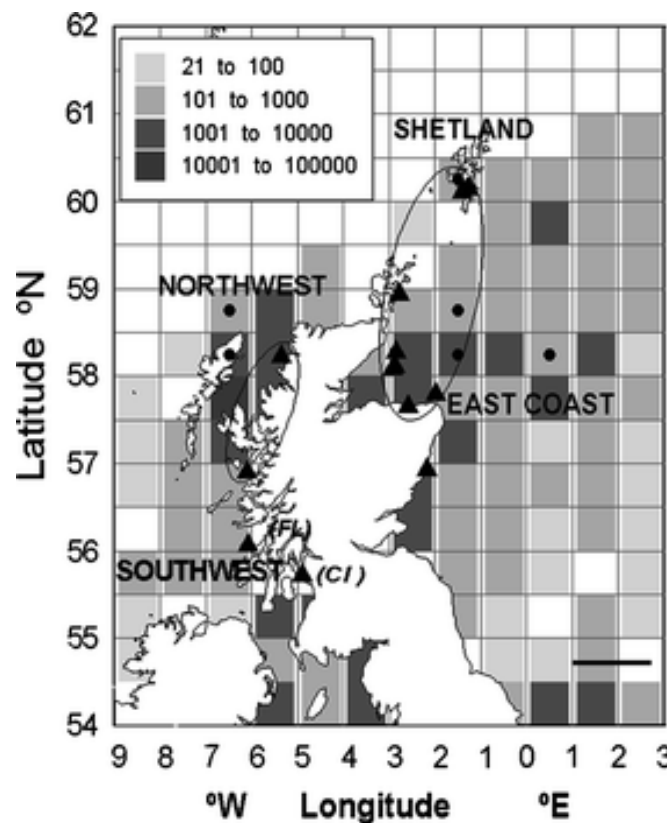


Figure 5.10: ICES survey trawl data for whiting collected from September to November between 1994 and 2003. Juvenile (0-group, black triangles) and adult (3-group, black circles) sample sites are presented. Shading refers to the 10-year average catch per unit effort (fish h⁻¹) data for 0-group whiting in each ICES statistical rectangle. Ellipses denote nursery regions based on the ability to discriminate between areas using otolith microchemistry. CI and FL refer to two distinct nursery microchemistry signatures in the southwest region. Scale bar equals 100 km (Tobin *et al.*, 2010). Scottish west coast could be viewed as a net source of recruits to the North Sea (Tobin *et al.*, 2010).

6 West of Scotland Demersal Fisheries

F D McIntyre and P G Fernandes

The seas to the west of Scotland are considered largely as ICES area VIa and although the sea does not have a specific name it encompasses many smaller seas such as the Hebrides, the Firth of Lorn, the Sound of Jura and the Firth of Clyde. In comparison to the North Sea, the fishing grounds, especially for demersal fisheries, to the west of Scotland are of a much smaller extent due to a rougher terrain and deeper waters.

This section reviews earlier assessments of the state of the demersal fisheries West of Scotland, and then goes on to present the most recent assessments.

6.1 Past Assessments West of Scotland

The fish and shellfish resources were reviewed for the seas adjacent to the Outer Hebrides by Bailey (1979) and for the Clyde sea area by Hislop (1986) as part of the Proceedings of the Royal Society of Edinburgh (see Figure 6.1 showing these two areas).

6.1.1 Outer Hebrides

The Outer Hebrides lie in the ICES statistical Area VIa to the west of Scotland. Bailey (1979) reported that in the late 1970's, demersal fisheries in the Hebridean seas were based mainly on haddock, whiting and cod mostly from local spawning grounds with some recruitment from the North Sea. At that time, although the state of exploitation of west coast demersal stocks was not well known the stocks in these grounds were described as "probably fully exploited".

The Fishery

In terms of the type of vessels fishing in the Outer Hebrides, the larger seiners and trawlers were mostly vessels registered on the east coast of Scotland working mainly north of a line from the Butt of Lewis to Cape Wrath. However, the fleet of smaller trawlers contains a higher proportion of vessels registered at west coast ports. Most of the coast west of Scotland had a very low population size and thus due to a lack of markets in the west of Scotland a very small percentage of the fish taken on the west coast were landed locally.

Landings

In terms of weight landed the most important demersal species exploited in Hebridean waters were cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), whiting (*Merlangius merlangus*) and spurdogs (*Squalus acanthias*). Lesser quantities of saithe (*Pollachius virens*), hake (*Merluccius merluccius*), ling (*Molva molva*), skate (*Rajidae*), anglerfish (*Lophius piscatorius*) and flatfish particularly plaice (*Pleuronectes platessa*). The

total landings for the main species cod, haddock and whiting in the Outer Hebrides from 1967-1976 can be seen in Table 6.1.

Table 6.1: Scottish landings (tonnes) of principle demersal fish species from the Outer Hebrides from 1967-1976 (a) Landings for the whole of Scotland (unpublished data), (b) Landings in the Stornoway district only. (Data from SSFST).

	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976
Cod										
(a)	2727	3545	4372	2880	1561	2345	1250	1814	1198	2309
(b)	69	81	84	32	29	32	107	149	135	214
Haddock										
(a)	5612	6420	9056	12697	17725	12322	8381	3702	3740	5071
(b)	45	29	74	115	60	61	56	49	44	129
Whiting										
(a)	4725	4194	4428	3715	7373	4442	2633	3748	4702	7704
(b)	119	245	239	137	123	107	312	497	823	931

Recruitment

Most of the demersal species had spawning grounds in the Minch to the west and the north of the Outer Hebrides. It was probable that a large proportion of the young fish originated from the Minch spawning grounds and once reaching sexual maturity were recruited directly to the fishery. It is also likely, however, that a proportion of the larvae and young fish from the spawning grounds west and north of the Hebrides were carried or moved to nursery areas in the North Sea. Movement of fish between the west coast and the North Sea was not in one direction only with studies from tagging experiments on cod, haddock and plaice showing that some adult fish move from the east coast of Scotland to western grounds. In terms of recruitment, fluctuations were known to have an effect on the relative importance to the fisheries of the main species, which varied from year to year.

State of the Stocks

In the late 1970's little was known about the demersal fish stocks and the extent of its geographical limits in this area and at that time they had not been subject to the international quota agreements. However, assessments of the west coast stocks had been made in preparation for the introduction of catch quota regulations for cod, haddock, whiting and saithe. Estimates made by ICES at the time indicated that fishing for cod, haddock and whiting was at or above the level that would generate maximum yield per recruit.

6.2 Clyde Sea

Hislop (1986) reviewed the information on the demersal fisheries in the Clyde Sea Area presenting data from the four ICES statistical rectangles 39E4, 39E5, 40E4 and 40E5 which are located to the southeast of Division VIa. The area contains a wide range of habitats; the substrate is muddy throughout much of the area but there are long sandy stretches along the

Ayrshire coast and there is hard ground along the edge of the deep channels in the North Channel and Loch Fyne. There are also narrow sea lochs with rocky shores and sandy bays.

Landings

The total demersal fishing effort in the Clyde was known to have risen in the 25 years prior to the 1990s almost entirely due to an increase in trawling for Nephrops from which there was a large bycatch of whitefish. According to the Scottish Sea Fisheries Statistical Tables (SSFST), which were published annually by DAFS, the annual weights of all demersal species landed at Clyde ports, in the early 1980's, ranged from 4,000 to 10,000 tonnes, valued at £2-5 million. In terms of value, cod and hake were the most important species in the Clyde fishery. This species composition of landings in the Clyde is unusual for a Scottish inshore fishery in that the proportion of haddock is rather small, while that of hake is relatively large. The annual landings for the three main species cod, haddock and whiting for the Clyde sea area from 1960-1984 can be seen in Table 6.2. It is worth noting that the prices of all major whitefish species rose steeply during the late 1960's and early 1970's, mainly because of the closure to the English distant water trawler fleet.

Table 6.2: Annual landings (tonnes) of the principle demersal species from the Clyde Sea Area from 1960-1984).

	1960	1961	1962	1963	1964	1965	1966	1967	1968	1969	1970	1971	1972
Cod	1146	560	440	414	732	1577	1598	2160	2265	2226	1808	1509	1703
Haddock	533	350	206	266	634	855	660	1093	888	789	699	568	663
Whiting	2050	1587	1926	725	2220	1672	1913	3166	1511	1402	529	419	825
	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	
Cod	1831	1565	1492	1582	1637	1524	1434	1716	2329	1861	1351	1348	
Haddock	424	206	108	129	125	110	79	224	403	420	164	186	
Whiting	1017	598	845	1240	1095	1080	718	1069	1983	1096	1160	804	

The Fishery

The demersal fishery was seasonal on the Clyde grounds and most of the fishing effort took place during the winter and spring, partly because catch rates were relatively high at this time of year and partly because the area provides shelter from the winter weather.

Recruitment

The Clyde fishery depends to a large extent on young fish and the relative importance of the principle species fluctuate from year to year in response to variations in the strength of the recruiting year class. Although there was no conclusive evidence that the Clyde populations were self-contained, all the major species are known to spawn within or close to the area and tagging experiments had shown that there was not much mixing between Clyde fish and those from surrounding areas.

State of the Stocks

In 1986 the demersal fish stocks in the Clyde were considered to be exploited intensively with the fishery there depending to a large extent on the younger age groups and thus the fishery was highly sensitive to variations in the strength of the recruiting year class.

6.3 A Note on Survey Nomenclature

The research vessel survey for demersal fish west of Scotland has had several names applied to it in a confusing manner.

For example, in the 2010 ICES Working Group for the International Bottom Trawl Survey (WGIBTS) the quarter one (Q1) survey is referred to as “Western Division Bottom Trawl Survey – Quarter 1”. However, the first paragraph of descriptive text then refers to it as the “Q1 Western Groundfish survey”. The internal figure legend is entitled “Q1 West Coast SCOGFS”, while the figure legend itself has “Q1WCSCGFS...” (no “O”) and the table legend has “Q1 SCOGFS ...”.

The same confusion occurs for the quarter four (Q4) survey, which is referred to as the “Western Division Bottom Trawl Survey – Quarter 4”, the “Q4 Western Groundfish survey”, the “Scotland Q4 IBTS survey” (i.e. “survey” twice), and the “Q4 SCOGFS”, all in the same Section (4.3.2.3, pages 59-60).

For the purposes of this report (and we suggest for all purposes from hence forth), we suggest the nomenclature;

- “Q1 Scottish VIa IBTS (Q1 Sco.VIa.IBTS)”
- “Q4 Scottish VIa IBTS (Q4 Sco.VIa.IBTS)”

as

- the surveys are part of the larger IBTS scheme;
- we need to distinguish them from the North Sea surveys by some reference to where they are carried out;
- the surveys are carried out by Scotland (i.e. are a subset of other western IBTS’ like the Irish groundfish survey and the “Q4 Western Groundfish Survey” which is carried out by England);
- and we need some reference to when they take place.

The names proposed are new and hence may add to the confusion, but the surveys are also new in some respects, so the new name would come at the same time as the change in design and gear too.

6.4 West of Scotland Fisheries Today

NOTE: Owing to the assessment of the state of fish stocks in area VIa being updated annually, this section has been removed as in any one year it will become out of date and hence may give an erroneous impression when read with the rest of this report. A separate report will be made available for each assessment year. Please contact Marine Scotland for details

6.5 The Q1 and Q4 Scottish VIa IBTS (Q1 Sco.VIa.IBTS and Q4 Sco.VIa.IBTS)

6.5.1 Introduction

The standard stock assessment techniques used for ICES WG assessments are all based primarily on landings-at-age data, which is considered to represent the actual stock. The research survey data, in general, is used to 'tune' the estimates from the landings data (Needle, 2003; Beare *et al.* 2005). The analytical methods used, such as Time-series Analysis (TSA) and virtual population analysis (VPA), mostly rely on the assumption that precise catch at age information is available (Patterson, 1997).

However, in the past there was growing concern over the quality of reported landings data which might have resulted in misleading scientific management advice. The concern was that the effect of restrictive TAC's did not lead to limited catches but rather caused a portion of the catch to go unreported, and thus this had severe consequences for stock assessments on which TAC's were based and generated (Cook, 1997). This led to a greater focus on research vessel survey data.

Fisheries research vessel surveys are conducted annually and provide a monitoring system which is independent of the landings. The uncertainties in the data has thus led to investigations and studies that consider alternative tools which do not depend on a long series of precise catch data but are based solely on survey data (Cotter *et al.* 2004). Also in response to the poor quality catch data, the European Commission identified research into the development of fishery independent assessment tools as a priority topic for research in support of the Common Fisheries Policy.

The main advantage to limiting an assessment to survey data alone is that research vessel data is under direct control and less prone to the biases affecting commercial data. In addition to avoiding the bias due to errors in the input data, survey assessments are not affected by errors in parameter estimates such as natural mortality rates as prior knowledge of this parameter is not required (Korsbrekke *et al.* 2001). The natural mortality coefficient is a difficult parameter to estimate and for cod is currently considered to be well below the actual value to the west of Scotland as a result of seal predation.

In addition, survey-based assessments can resolve the timeliness issue as the use of catch data causes the current one year delay between the 'data year' and the 'assessment year' which often attracts criticism (Cotter *et al.* 2004). The reason for this being that it often takes considerable time to collect and collate commercial fishery data and stock assessment estimates are not available until several months after a survey is completed (Pennington and Nakken, 2006).

However, there are several issues that need to be considered when using survey data as the sole source for carrying out stock assessments. The main issues that lead to uncertainty are;

- the surveyed area does not cover the stock area;
- the age classes sampled are not representative of the stock
- the possible variation in survey catchability (ICES, 2007a).

An inherent problem with survey based assessments is that it is critically dependent on the quality of the survey protocols, data and design. For example, when a research survey covers a wide range of species it is important to assess if the survey is adequate for each species included in the survey. In such situations there will inevitably be a compromise in the design.

The estimation of abundance from scientific surveys is a fundamental task for stock assessment and management. Resource management is based heavily upon stock abundance estimates and hence errors in these estimates directly affect the confidence in conservation measures applied. Unfortunately, many marine species show spatial autocorrelation and skewed spatial distributions that reflect a concentration of abundance in proportionally small sample areas (Spencer *et al.* 2009).

When limiting the assessments to survey data alone a further concern is that survey-based assessments estimate removals from the stock and total mortality rather than landings and fishing mortality. Hence survey-based assessments are more difficult to use as a basis of quota advice than corresponding catch-based approaches.

Abundance estimates from a survey based assessment are not an absolute measure of stock size. Rather, survey-based abundance estimates should be considered as a relative measure of stock size variability and presented using arbitrary scaling coefficients. This may have consequences in terms of the nature of the subsequent advice and the definition of reference points. Relative estimates of SSB and recruitment are sufficient if relative trend management is practised (ICES, 2004; Beare *et al.* 2005).

6.5.2 Survey Based Assessments – Research Projects

As the reliability of west of Scotland catch data is under question, it is important to investigate the ability of surveys to fulfil stock assessment requirements in terms of giving

reliable and valid scientific management advice. The level at which survey based assessments perform depends on the precision of the survey (Patterson, 1998). If a survey is very precise it would be sufficient to use it for stock assessment purposes. Conversely, Patterson (1998) suggested that if only noisy survey information is available, it may be more desirable to use tuned VPA methods unless the errors in catch reporting are very high.

In addition, there has been a shift to an 'ecosystem based approach to fisheries management' which at the least implies that the interactions between ecosystem processes and fisheries needs to be taken into account when determining management advice (ICES, 2007). This shift in priorities will affect current stock assessment methods and data collection. In terms of ecosystem based management there is a broad consensus that the traditional methods of fish stock assessment is not an option given the huge gaps in knowledge and the cost of collecting the appropriate data. The possible alternative method is to look at indicators that reflect the state of marine systems and to monitor changes in such indicators.

EVARES

The EVARES project was established in 2001 in order to assess the quality of research surveys in terms of management advice (Beare *et al.*, 2003). The project covered both data analysis and simulations for a pan-European selection of fish stocks and surveys. In the case of ICES Division VIa, raw data from the surveys were used to construct survey indices for the area. Length frequency data by haul and associated age length keys were used to give a Catch Per Unit Effort (CPUE) index. Simulations were then used to measure the variability in the assessment of fishing mortality (F) and spawning stock biomass (SSB) due to the variability in the survey data and to investigate the effects of modifications in individual surveys such as a shift in sampling intensities.

Correlation coefficients calculated over years between the abundance for age a and year y and age $a+1$ and year $y+1$ offer a good indication of the ability of a survey to track year class strengths. Initial results from the EVARES report for Division VIa showed that the west coast surveys are able to track population changes over time.

This was considered a promising result in terms of survey usefulness and reliability as a basis for management advice. In addition it was found that the west of Scotland surveys (Q1 and Q4) were both positively correlated with the numbers-at-age and stock-weights at age output by the TSA assessments of VIa cod, haddock and whiting.

FISBOAT

FISBOAT (Fishery Independent Survey Based Operational Assessment Tools) was set up by the EU in response to the concerns over the quality of the catch data and the resulting stock abundance estimates. The project consisted of participants from 11 research institutes with the aim of developing specific methods for analysing fishery independent stock assessment data to provide managers with relevant information about the stock and its exploitation.

The project evaluated fishery independent models based on simulated data. Methods used that were intended to estimate abundance included age-structured (SURBA, TSA, YCC) and length structured (LENSUR) assessment methods. Simulation methods to assess the effects of changes in biological or management parameters were represented by ALADYM.

Results from this investigation showed that there is no inherent defect in the rationale of these survey-based methods and that all methods perform quite similarly in terms of patterns in summary stock plots (Cotter *et al.*, 2007). However, it is important to note that the methods tended to behave as smoothers of noisy indices and could possibly miss quick transient changes in stock abundance (Woillez *et al.*, 2009).

6.5.3 Examination of Research Vessel Survey Quality

The Scottish Vía International Bottom Trawl Survey (Sco.Vía.IBTS) is carried out annually in Quarters 1 and 4 during March and November respectively of each year. Survey data is available from 1985 for the Quarter 1 survey and from 1996 for Quarter 4.

The survey series are made available to the ICES Working Group in order to tune the stock assessments. To date, the assessment for cod has not used the Quarter 4 survey data. However, both Quarter 1 and Quarter 4 surveys have been used to tune haddock assessments since 2006. The haddock assessment uses a modified TSA assessment method which excludes catch data after 1994.

As mentioned earlier, an assessment for whiting has not been performed since 2006 and this assessment was based solely on survey data due to the concern over the reliability of catch data.

The quality of the assessments for the demersal fish stocks west of Scotland has often been questioned and in particular by fishermen. However, despite their importance relatively few studies have been undertaken to investigate the reliability of assessments and to determine the origin of potential error sources (Leseuer, 2004).

In order to examine the quality of the Scottish Vía IBTS, particularly in terms of its usefulness for stock assessment purposes, some exploratory analysis was performed on cod, haddock and whiting survey data for Division Vía.

6.5.4 Survey Effort Comparisons – North Sea cf. West Coast

There is some perception that more effort is given to the North Sea International Bottom Trawl Survey (IBTS) compared to the west coast IBTS. After reviewing the information from the last 6 years, it was found that both surveys average about the same number of hauls per ICES rectangle between the Quarter 1 and Quarter 4 surveys. As an example, Figure 6.2 shows the number of hauls per ICES rectangle for the Quarter 1 survey to the west of Scotland and in the North Sea in 2007.

6.5.5 Application of a Survey-Based Assessment Model (SURBA)

Background

Modified assessment models based largely on survey data (i.e. Time Series Analysis models – TSA) are becoming the standard assessment method in several ICES assessments for which problems exist with commercial catch data, including cod and haddock in Division VIa. Other models used, which are based solely on survey data include BADAPT and SURBA.

The SURBA model is currently a survey based assessment which has a long history in catch-based fisheries stock assessment and was first applied to European research vessel survey data by Cook (1997). The model assumes that fishing mortality (F) is separable into an age effect (s_a) and a year class effect (f_y) so that $F = s_a \times f_y$. The model estimates the s_a and f_y parameters along with a year class effect r , by minimizing the sum-of-squared differences between the observed and fitted survey derived abundances using an assumed fixed vector of catchabilities-at-age (q_a), which does not depend on year (Beare *et al.* 2005). The simple basis of the model has been expanded and adapted over the years and has been recently used experimentally in ICES assessment working groups such as in the assessment of Northern Shelf Demersal Stocks (WGNDS now incorporated into the WG Celtic Seas - WGCS) for west of Scotland ICES Division VIa.

Survey Selection

For this report, SURBA 3.0 was used to analyse the survey data from the Quarter 1 survey ScoGFSQ1 for cod, haddock and whiting in the west of Scotland. Attempts to fit the model to ScoGFSQ4 survey proved unreliable and this may be a result of the relatively short time series for this survey (DATES). It was found that SURBA could not fit an optimum model for the Q4 survey, particularly in the case of whiting. The survey data for ScoGFSQ1, however, is available from 1985-2009 for all three species, cod haddock and whiting.

It should be noted that the working group assessment for haddock uses a modified TSA assessment method since 2007, which excludes catch data after 1994 and uses only survey data from then on. It is also the only assessment that uses both Quarter 1 and Quarter 4 survey data.

Model Parameters

The parameters set for the SURBA models were as follows:

	Data	Ages
Cod	ScoGFS Q1	1-6
Haddock	ScoGFS Q1 and Q4	1-7
Whiting	ScoGFS Q1	1-6

In addition, for all three species lambda was assumed to be 1, equal catchabilities were assumed and data from all available years were used.

Results

The summary plots resulting from the SURBA run were compared to the working group TSA assessment summary plots for cod, haddock and whiting (see Figures 6.3 - 6.5). Although the most recent assessment for whiting was a SURBA model assessment using survey data in 2006, a TSA assessment was conducted in 2004 and the output from this is compared with the SURBA 3.0 run for this report. All plots were produce using R version 2.10.0

The results of the analysis show that in general the surveys support the stock trends estimated from the ICES assessments for cod and haddock with some disagreement between the two analyses for whiting. The SURBA results for cod in Division VIa supports the perceived long term decline in the stock with some stabilization in recent years. Trends in recruitment, SSB and Z show the most similar patterns between the working group assessment and SURBA outputs for haddock and this may be attributed to the fact that commercial catch data is excluded from the assessment for haddock from 1994 onwards.

The whiting comparisons with the SURBA run show more conflicting results, which could be explained by the fact that the TSA model run in 2004 was found to behave in a manner difficult to explain as the information from the catch and survey data was quite divergent. Considering the outputs for whiting, SSB appears to have reached an all time low and recruitment also appears to have fallen in the more recent years (see Figure 6.5).

The SURBA method appears to provide consistent estimates of relative trends in population parameters from research vessel survey indices. Clearly it is possible to get detailed information from survey based assessments and use them for stock assessments when catch data is thought to be unreliable.

Year Class Curves

Year class curves can be a very useful tool for fisheries managers to estimate the mortality rate of a fish stock. They can be constructed for multiple year classes as a snapshot of the individuals alive in a population at a given time, so each line in the plot describes the fate of

successive cohorts as they enter the population. Year class curves have three areas of interest; the ascending left limb represents age classes of fish that are not fully recruited to the gear used (Ogle, 2008). The domed portion of the year class curve in general represents the age classes of fish that are nearly, but incompletely recruited by the gear used. The relative width of this domed region gives an indication of how quickly fish are recruited; a sharply pointed dome indicating that fish recruit very quickly and a relatively more rounded dome indicating that the fish recruit at a slower rate to the exploited population more slowly. The descending limb of the curve represents the regular decline of the fully-recruited individuals (Ogle, 2008).

The year class curves were plotted for cod, haddock and whiting using the input data for the SURBA run i.e. the survey data. The curves are plotted in Figures 6.6 -8 using R version 2.10.0. The cod year class curve shows a somewhat 'hooked' nature at the younger ages with abundances at age two higher than that at age one. This 'hooked' nature of the curve is also apparent in the haddock year class curve but is not the case in the whiting curve.

Within-Survey Consistency

Correlation coefficients calculated over years between the abundance index for age a and year y can offer a first indication of the ability of a survey to track year class strength effects. Within-survey consistency was examined by calculating the correlation between successive age groups. For example, all age 1 fish are correlated against age 2 the same for age 2 and age 3 etc and also the correlation between ages separated by a year or more. The internal consistency of the west coast survey Q1 for cod, haddock and whiting is shown in Figures 6.9, 6.10 and 6.11 respectively. All plots were produced using R version 2.10.0.

Within-survey correlation for cod was significant and positive between all age groups. Cod showed positive correlation between the younger age groups which weakens slightly as the fish mature (see Figure 6.9). Haddock also show a highly positive correlation between successive age groups which weakens slightly as the age gap increases between the correlated ages (Figure 6.10). The within survey correlation for whiting is highly variable, although there is good correlation between successive age groups, the correlation weakens dramatically when the age gap between the correlated ages increases (Figure 6.11).

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6.7 Figures

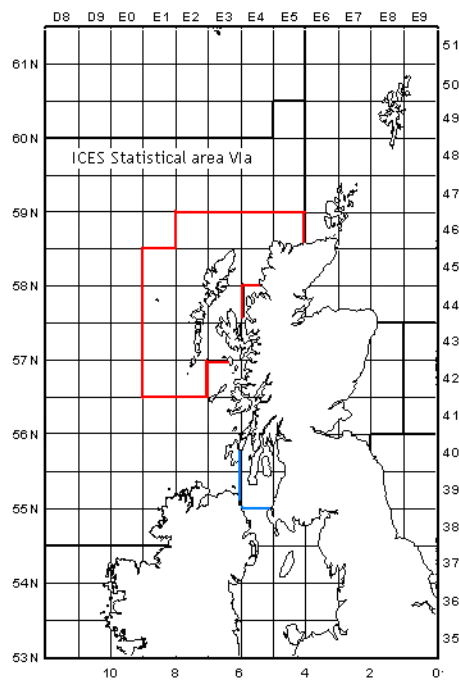


Figure 6.1: ICES Division Via. The Outer Hebrides area is denoted by the red box and the Clyde Sea area by the blue line.

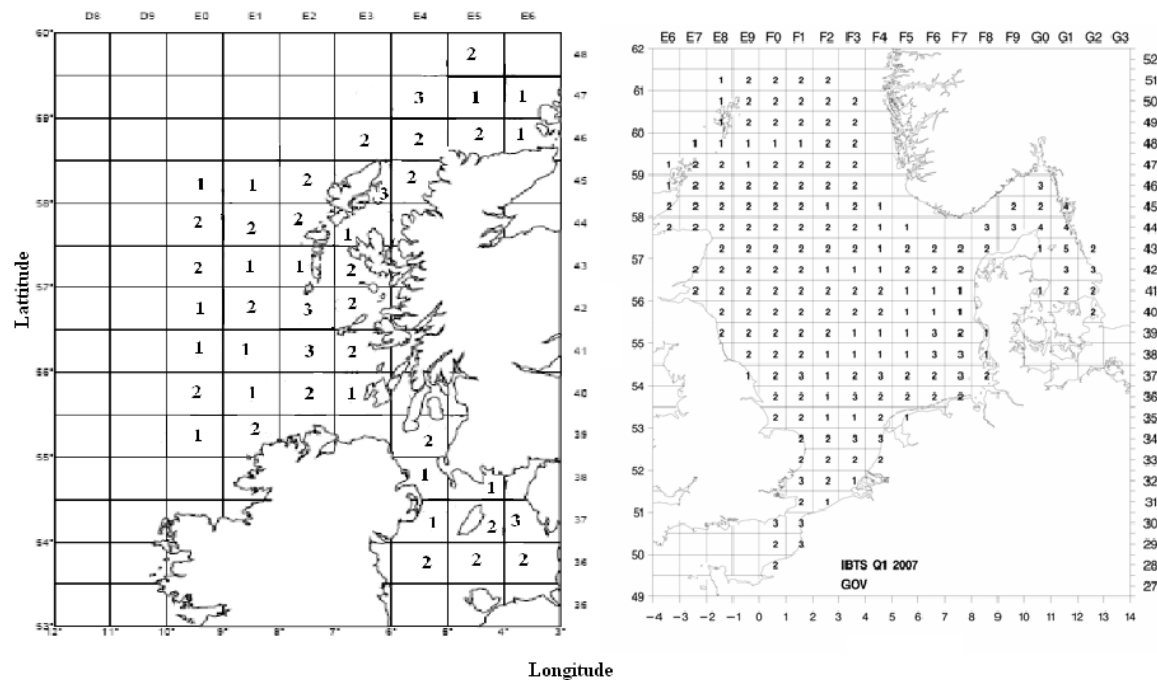


Figure 6.2: Survey effort comparison. The number of hauls in each ICES rectangle for the west coast Q1 and North Sea Q1 surveys in 2007.

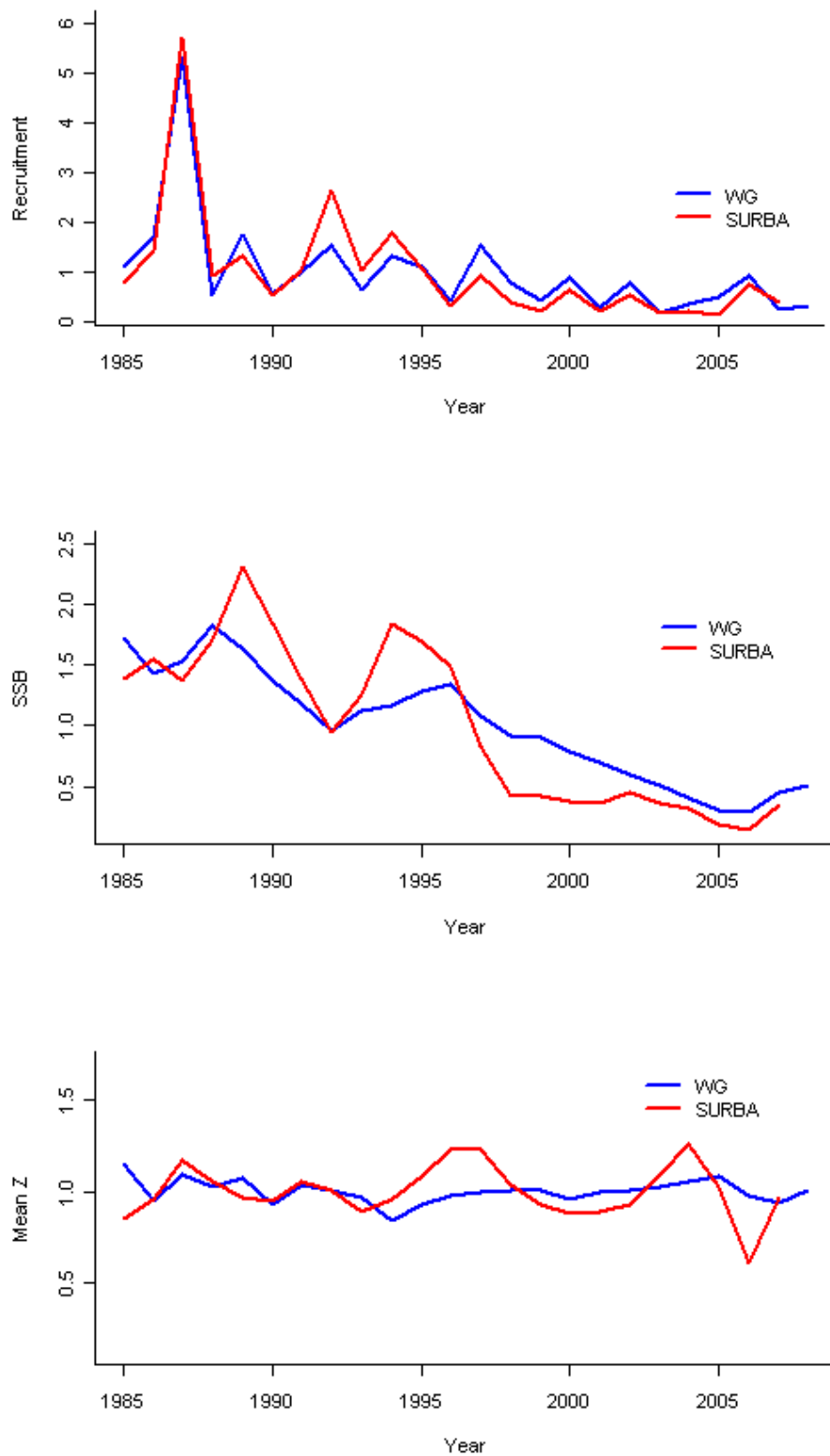


Figure 6.3: Cod summary plot comparisons between the recent working group TSA assessment and SURBA outputs.

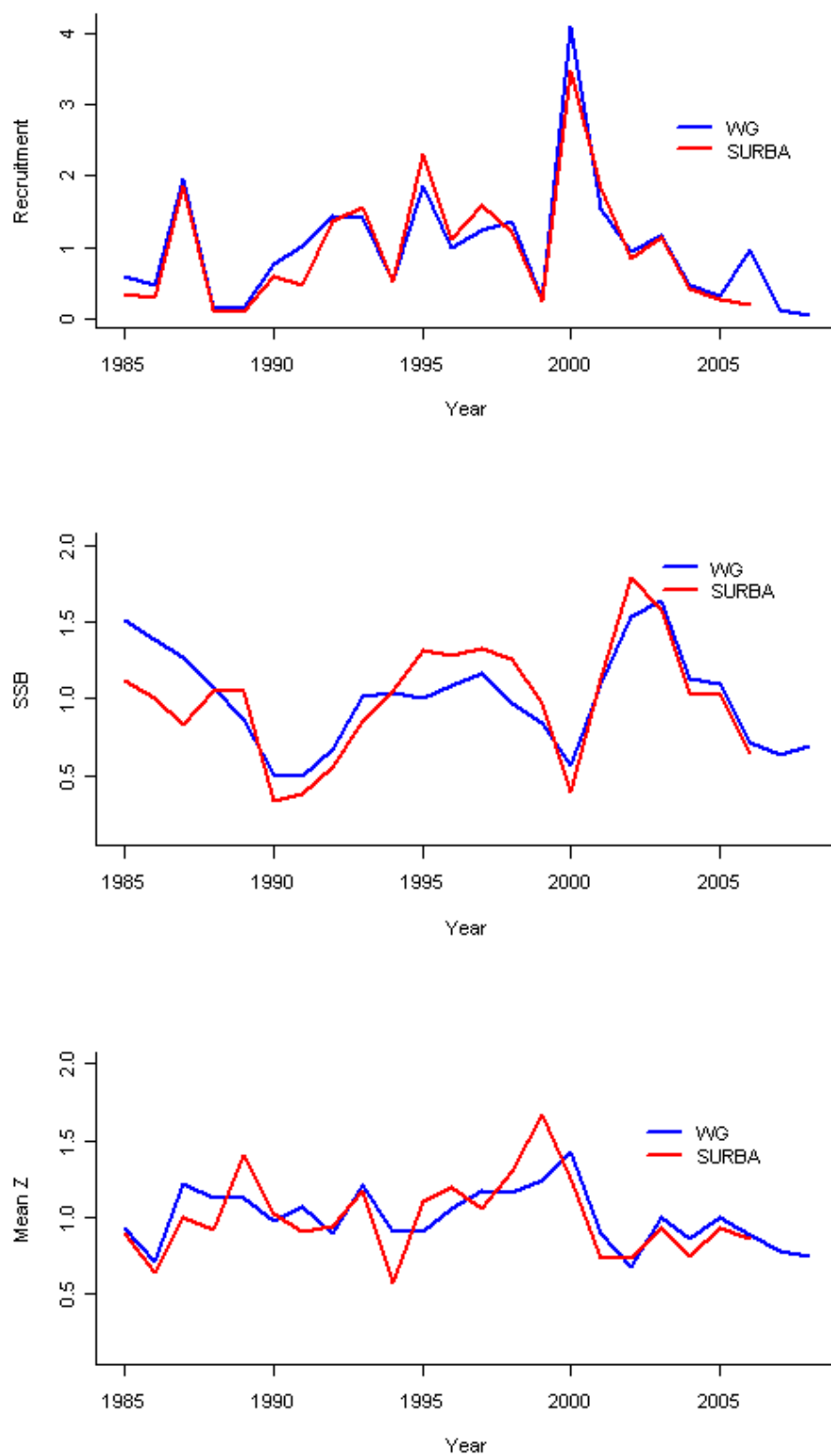


Figure 6.4: Haddock summary plot comparisons between the recent working group TSA assessment and SURBA outputs.

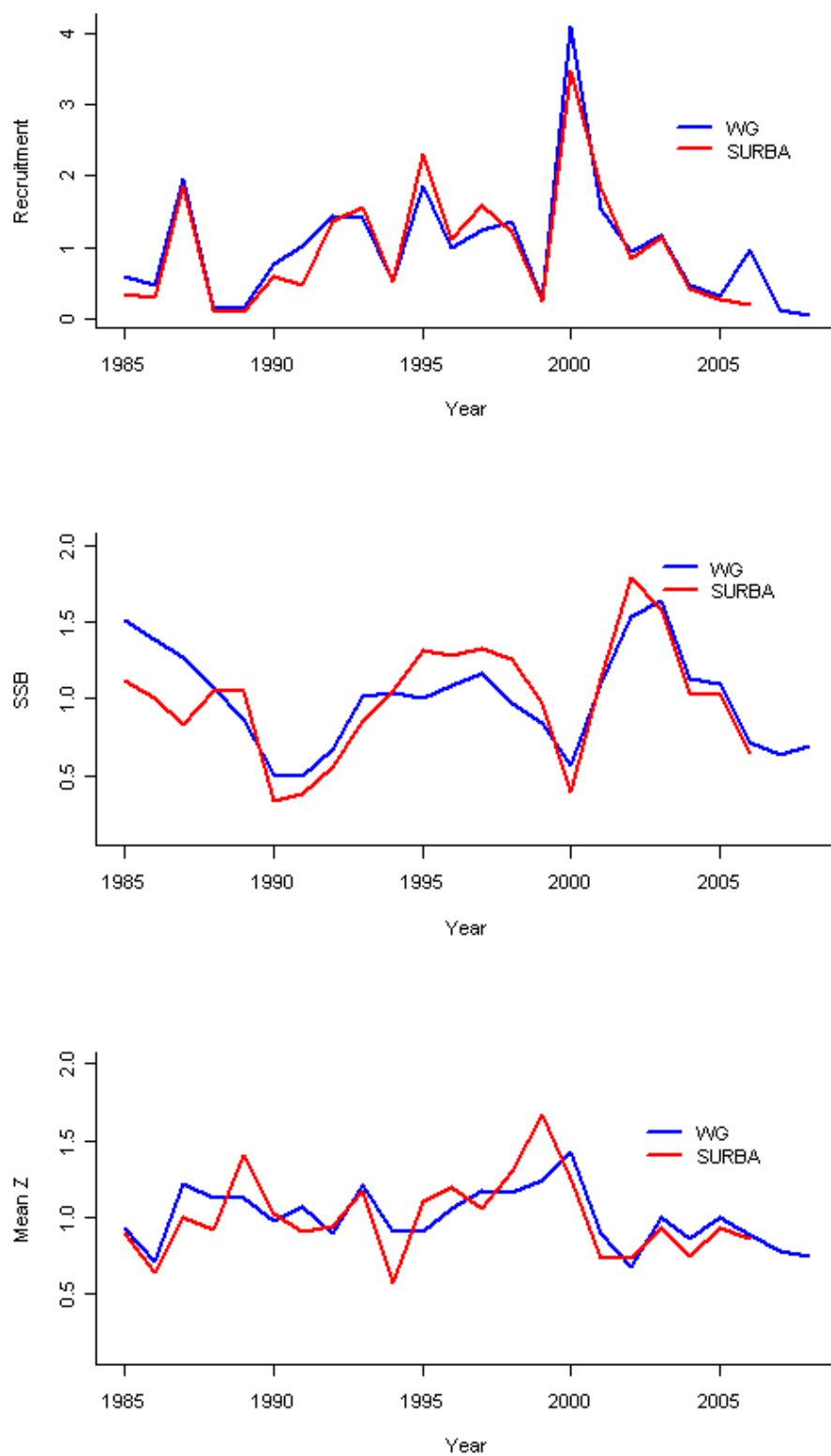


Figure 6.5: Whiting summary plot comparisons between the recent working group TSA assessment and SURBA outputs.

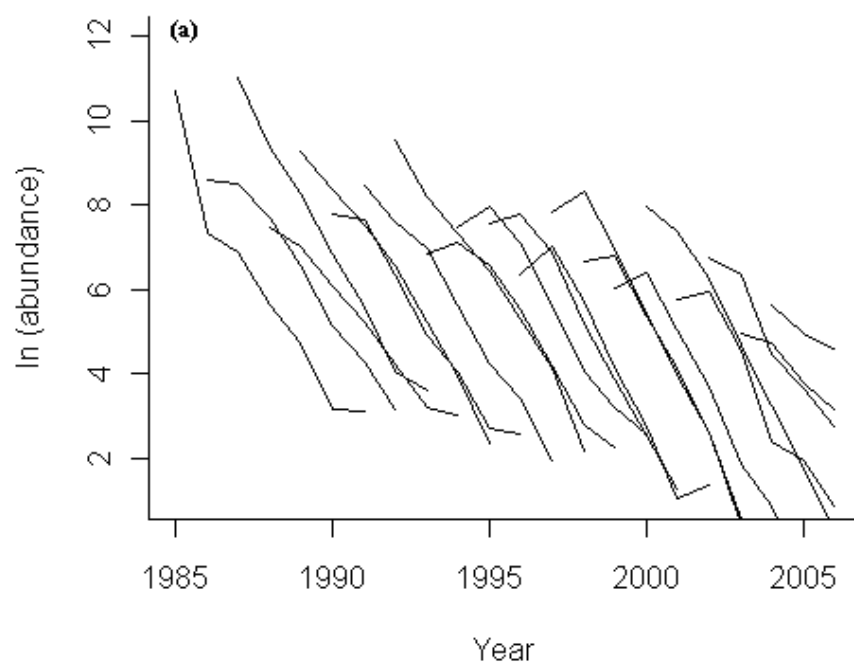


Figure 6.6: Cod in Division VIa. Year class curve from the Q1 Scottish VIa IBTS.

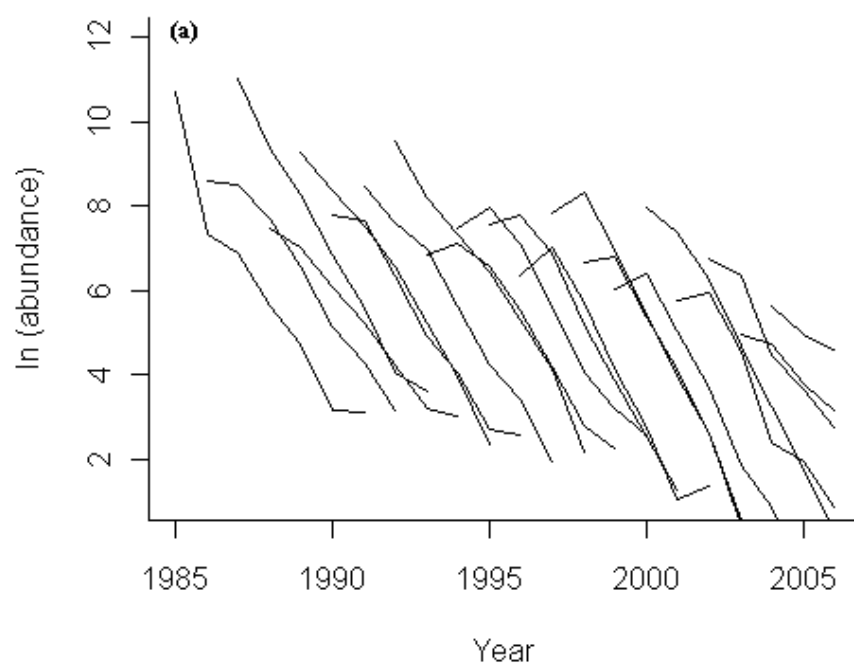


Figure 6.7: Haddock in Division VIa. Year class curves from the Q1 Scottish VIa IBTS.

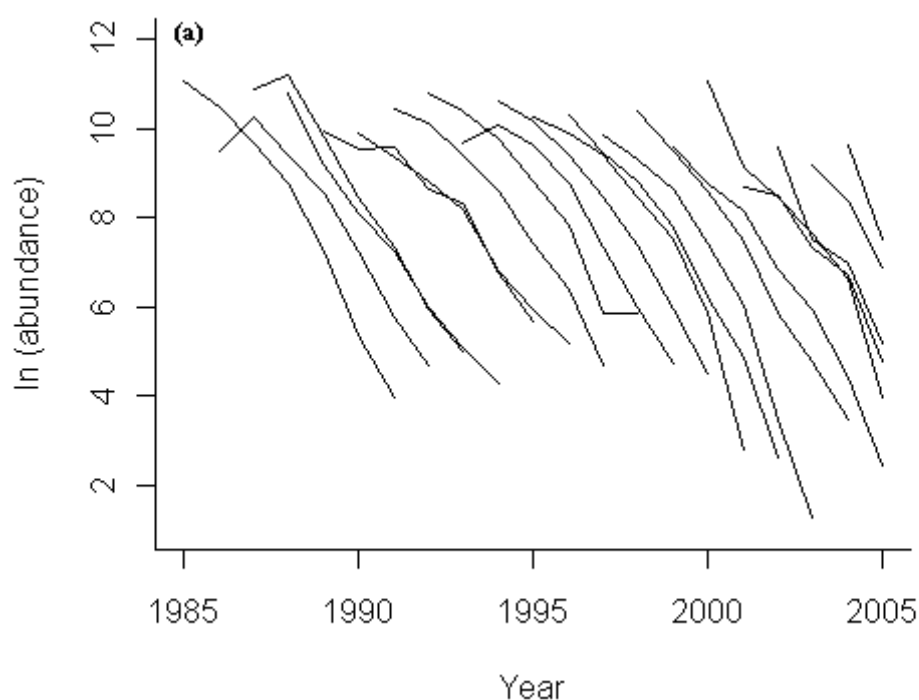


Figure 6.8: Whiting in Division VIa. Year class curve from the Q1 Scottish VIa IBTS.

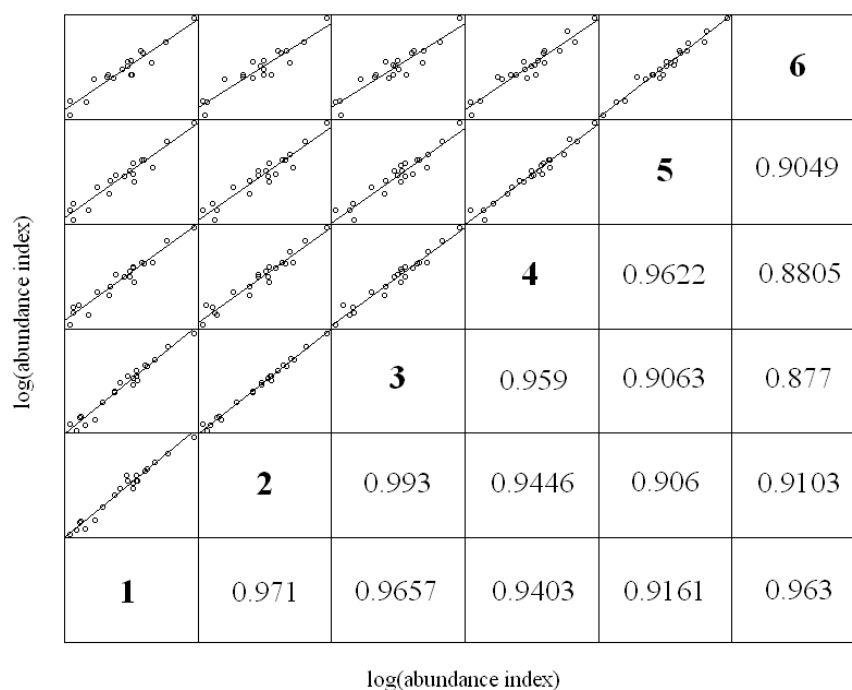


Figure 6.9: Cod in Division VIa. Internal consistency within the Q1 Scottish VIa IBTS. The upper left part of the plots shows the relationship between log index-at-age within a cohort. Linear regression line shows the best fit to the log-transformed indices. The ages plotted on the x and y axes of each panel can be found by moving either vertically (for the x-axis) or horizontally (y-axis) to the main diagonal. The lower-right part of the plots shows the regression coefficient (r^2) for the two ages plotted in that panel.

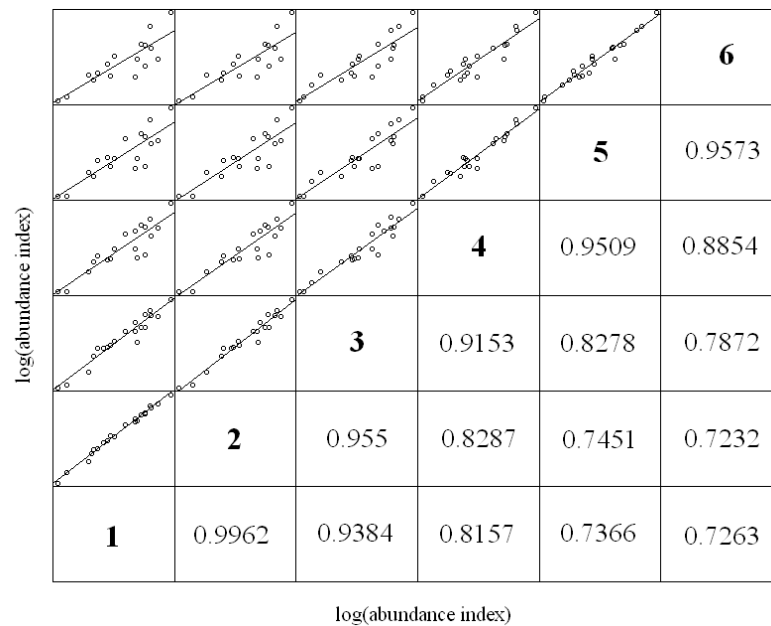


Figure 6.10: Haddock in Division VIa. Internal consistency within the Q1 Scottish VIa IBTS. The upper left part of the plots shows the relationship between log index-at-age within a cohort. Linear regression line shows the best fit to the log-transformed indices. The ages plotted on the x and y axes of each panel can be found by moving either vertically (for the x-axis) or horizontally (y-axis) to the main diagonal. The lower-right part of the plots shows the regression coefficient (r^2) for the two ages plotted in that panel.

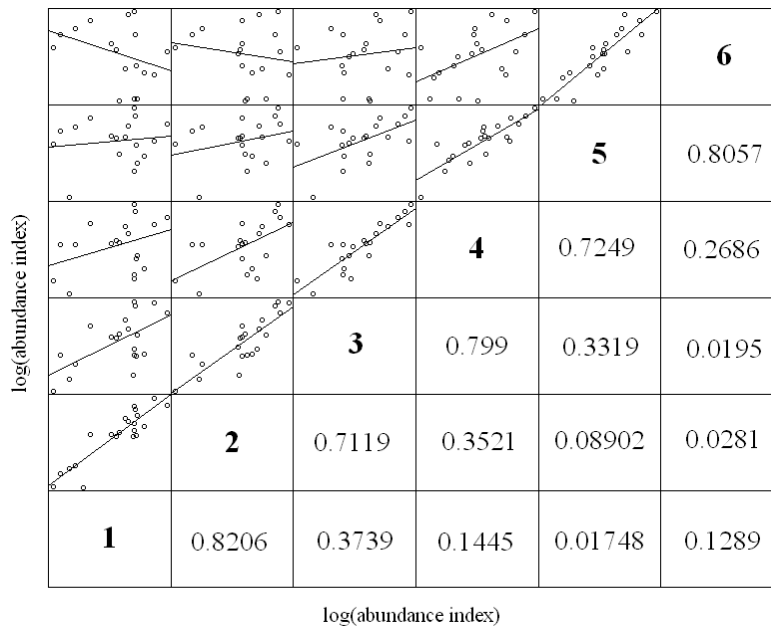


Figure 6.11: Whiting in Division VIa. Internal consistency within the Q1 Scottish VIa IBTS. The upper left part of the plots shows the relationship between log index-at-age within a cohort. Linear regression line shows the best fit to the log-transformed indices. The ages plotted on the x and y axes of each panel can be found by moving either vertically (for the x-axis) or horizontally (y-axis) to the main diagonal. The lower-right part of the plots shows the regression coefficient (r^2) for the two ages plotted in that panel.

7 Gadoid Abundance From The 2010 West Coast Industry Science Survey

P G Fernandes, R Watret, I Penny and F McIntyre

7.1 Introduction

The main species targeted by the demersal fisheries west of Scotland are cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*) and whiting (*Merlangius merlangus*). Many juveniles of these species are also discarded by the fisheries for the Norway lobster *Nephrops norvegicus*. The gadoid fisheries are located on the shelf (<200m) and are exploited throughout the year, with the highest landings usually made in February-April. In 2009, the annual landed value of the three species combined to the west of Scotland was £3,315,000 (Anon, 2010), so they are of significant commercial importance. Over the last century, their numbers have undergone significant fluctuations (Hislop, 1996). Currently the stocks are heavily exploited, with heavy restrictive measures in place to conserve the stocks, in particular for west coast cod which has been subject to a recovery plan since 2004 and, more recently, a long term management plan introduced in 2008. The Scottish fleet has undergone several changes in recent years either directly or indirectly prompted by measures designed to protect these vulnerable whitefish stocks.

Assessments of the status of two of these stocks (cod and haddock) are carried out each year by the International Council for the Exploration of the Sea (ICES) Working Group on Celtic Sea Ecosystems (WGCSE), where estimates of abundance are produced using catch-at-age data and survey data for ICES Division VIa. In the case of whiting the assessment has only produced estimates of relative abundance at age. The quality of these assessments has recently been under question as the landings data are thought to be unreliable, due, in part, to the highly restrictive TAC's and regulations operating within this division to conserve the vulnerable whitefish stocks. The unreliability of the catch at age data has resulted in the assessments being more reliant on survey data.

Surveys to estimate the relative abundance of these species have been carried out using the research vessel FRV Scotia since 1986. Currently, these are co-ordinated under the auspices of ICES as part of the International Bottom Trawl Survey (IBTS) series. These IBTS surveys take place in quarter one of each year. Since 1994 a second survey in quarter 4 has also been conducted annually. However, there has been criticism over the design of both Scottish VIa IBTS due to a perception that there is not enough effort placed in areas of known high gadoid densities. In 2010, Marine Scotland Science (MSS) conducted two west coast gadoid surveys: the Q1 Scottish VIa IBTS using the research vessel FRV 'Scotia'; and the Q1 Scottish VIa Industry Science Forum Gadoid Survey (hereafter referred to as the "industry science survey") which utilised two chartered vessels. The design for the latter survey was drafted by Marine Scotland scientists in consultation with fishing representatives, which resulted in the definition of new density strata based on their experience. The basis for this stratification was to allow for more samples per unit area to be taken from the high

density strata as compared to the lower density strata, to improve the precision of the survey.

Estimates of fish abundance from bottom trawl surveys can be based on a swept area model, whereby the density (numbers of fish per unit area) at a trawl location is represented by the number of fish caught in the trawl divided by the area swept by the trawl. The swept area is the product of the net width (wing spread – Figure 1) and tow distance (Walsh, 1996). These swept area techniques were originally developed to estimate biomass of unexploited stocks but are now being applied to fish stocks when the available data are insufficient to apply catch-at-age models (Somerton et al., 1999). However, the behaviour of the fish in relation to the gear has an affect on what metrics are used and also on what corrections need to be made to estimate the true density of fish in the area sampled. As such, absolute estimates of abundance from trawl surveys rely on having good information about the catchability of the net or the whole gear selectivity of the net.

The aim of the work reported here was to estimate the absolute abundance (total number and biomass) of the three gadoid species in ICES Division VIa, as opposed to an index of abundance which is normally produced from the west coast surveys. Estimates are provided for different assumptions about catchability using the wing spread as the swept area and different coefficients of catchability, including herding from the larger area swept by the doors. A catchability coefficient with a herding coefficient of one and a net retention coefficient of one, whereby all fish from the area swept by the doors are included in the density estimates, represents the minimum density sampled and these can be considered as the absolute minimum estimates of abundance. Estimates of sampling error are also included. The methods employed were devised in consultation with members of the local fishing industry. An industry science planning group was created consisting of several local fishermen and the scientists involved in the industry science survey. This group specified the net that was used and defined the strata on which the survey design was based. The survey was then carried out using chartered industry vessels with scientific staff on board to process the catch. This type of industry science collaboration is proving ever more useful to provide not only more credible results but also more effective ones in terms of the acceptance by the major stakeholder.

7.2 The 2010 Industry Science Survey Design

The survey area encompassed ICES Division VIa, i.e. the waters of the continental shelf to the west of the British Isles, west of longitude 4°W, north of latitude 54°30'N to a northerly limit of 60° 30'N. This area was further limited to areas where the depth was less than 300 m. Two regions were proposed as distinct areas to be surveyed by the two commercial vessels: northern areas of the west of Scotland and southern areas (see Figure 2a). The industry science survey took place 1-10 March 2010 with MFV Carina covering the northern part of the survey and MFV Genesis the southern part of the survey area. The industry science survey was implemented using a 'stratified' design where the survey area was divided into a small number of regions (strata) with a different amount of sampling effort per

unit area in each stratum. These strata were designed in consultation with fishermen with expertise of the area to determine strata with four levels of expected abundance: low, medium, high and very high (Figure 2b). A greater number of samples per unit area were taken in strata which were expected to have higher abundance. This improves the precision of the survey, as areas of high abundance are usually those of high variability which can be better sampled by taking more hauls in these areas. Compared to the areas of low abundance, eight times as much effort (samples per unit area) was allocated to the strata expected to have the very high abundances of gadoids, four times in areas of high abundances and twice the effort per unit area in areas of medium abundance. Areas unsuitable for fishing such as the rocky area west of the Hebrides were not sampled and were not included.

Survey gear

Two new trawls, accepted by the industry as effective, were purchased specifically for this survey so that the same trawl gear could be used for each charter vessel with the gear rigged in a consistent manner and no modifications were employed. The fishing gear must be standardised in this way in order for the survey results to be reliable, consistent and scientifically acceptable. The specification for the trawl design, specified by a sub-group of the industry science planning group, was as follows:

1. Mesh size in the upper wings and square to be 160 mm (6.25 inch).
2. To ensure all small (less than 20 cm) cod, haddock and whiting are retained the mesh size in the lower rings and belly must be no greater than 120 mm
3. High tenacity twine should be used in the upper/lower wings, square and the belly/top panel sections behind the 2nd tearing strip. Standard polyethylene twine to be used for the remainder of the tapered section.
4. Upper and lower wing tips to be constructed from double twine.
5. The extension to be constructed from 100 mm diamond mesh by 5 mm diameter high tenacity single twine by 150 meshes deep.
6. The intention is to fish the trawls with 20 mm nylon blinders supplied by Marine Scotland Science. However, codends will be required and should be constructed from 100 mm diamond mesh by 5 mm diameter high tenacity double twine by 50.5 mesh deep. The codends should be supplied with codlines and lifting beackets.
7. Both Headline and footrope to be wrapped with rope and the trawls rigged with selvage ropes.

8. Design must incorporate measures to give added strengthening to weak points around the mouth and belly of the trawl. This strengthening to be similar as that which would normally be built into commercial trawls (i.e. top and bottom guard meshes and tearing strips etc).
9. Tearing strips and guard meshes to be constructed from 6 mm diameter double high tenacity twine e.g. Carflex or similar grade of material.
10. Flotation to be from 200 mm diameter (8 inch) floats rated to max depth 1200 m x 4 hour duration.

Three types of acoustic trawl sensors were installed on each vessel: Scanmar door spread sensors, wing spread sensors and a bottom contact sensor. Measurements of trawl-door spread, wing spread and bottom contact were made at each haul. The measurements were then filtered to remove spurious values and averaged for each haul. The data were matched to positional information (from GPS) at the same time resolution. The start and end times of the trawl hauls were determined from observation of the Scanmar data. Distance along the trawl track was calculated by comparing the start and end positional co-ordinates.

Sample location and duration

Co-ordinates of the trawl locations were allocated and provided to the scientist in charge (SIC) prior to departure. At each location, the skipper, in consultation with the SIC, decided on the best way to fish the location taking into account the bottom type and weather conditions. The SIC ensured, where possible, that the actual location of fishing did not deviate by more than 5 nautical miles from the allocated location.

The trawl sample size is driven by the amount of time or tow duration. Scientifically, it is desirable to obtain as many samples as possible over the area and so a short tow duration is preferable. Trawl surveys usually employ half hour tows whereas commercial trawling typically lasts for 5 hours. However, the industry science group recognized the need to obtain representative samples of fish density, i.e. numbers of fish per unit area as opposed to total quantities of fish. In addition, with a small mesh blinder many smaller fish are caught and a long tow would result in a huge bulk, making the tows very difficult to process. Taking all of these factors into account, it was decided that tow duration should be one hour. The tow duration was actual "bottom time". This was estimated by the skipper according to his perception of when the gear arrived at and left the seabed using the Scanmar sensors and other information such as the amount of trawl warp deployed.

Sampling procedure

Each chartered vessel carried three scientific staff from MSS who were responsible for processing the catch. Samples were processed in accordance with MSS standard operating procedures for trawl surveys which have much in common with internationally agreed

protocols (ICES, 2006). Length frequencies were recorded. However, the otoliths were not taken for reading as an age length key was derived from the corresponding Q1 Scottish V1a IBTS which took place in the same area at the same time. The rest of the catch was processed by the crew so that it could be landed and used to offset the cost of the charter.

Data analysis

Numbers and weights for all cod, haddock and whiting were taken for each haul. To calculate the distance fished the latitudes and longitudes of the start and end of the haul were transformed to distance in metres using a Euclidean distance formula corrected for latitude. The swept area of the wings and doors were then calculated as the average of the individual wing/door spread measurements multiplied by the distance travelled. Two measures of area swept by the trawl gear were determined; the area swept between the wings of the net (v_1) and the area swept between the doors (v_1+v_2 , where v_2 was the area swept by the sweeps, i.e. between the doors and the wings, see Figure 1).

The proportion at age in each haul in the industry science survey was derived by applying a species-specific age-length key from the Q1 Scottish V1a IBTS to the lengths obtained in the industry science survey. Where an age was missing for a particular length class, the age of the next largest length class was chosen. The average fish density at age in each stratum $\hat{\rho}_{ans}$ (numbers at age per square kilometre) was then estimated as:

$$\hat{\rho}_{ans} = \frac{1}{m_s} \sum \frac{n_{ais}}{v_{1i} \hat{Q}_i} \quad (1)$$

where:

n_{ais} = number of fish at age caught in trawl i in stratum s

v_{1i} = the area swept by the net in trawl i (wing spread x distance) [km^2]

m_s = the number of trawls in stratum s

\hat{Q}_i = catchability estimate for trawl i

Wing spread is generally used as the metric to estimate the sampled width of the net. However, door spread may also be considered to be representative of the effective whole gear width because it is known, for example, that when sweep lines are increased, door spread increases and higher catches result (Walsh 1996). This is the result of the auditory and visual herding stimulus produced by the sand clouds created by the doors and sweep lines (Figure 1). Fish will enter the trawl path between the doors and of these a proportion will be herded by the sand clouds into the net. Using the wing spread to calculate estimates of abundance assumes that all fish herded between the doors are caught. A herding coefficient (h), equal to the actual proportion of fish in the sweep area (Figure 1) that are herded into the net, can be applied to adjust the density estimates. Additionally, once the

fish have been herded into the area where they are likely to be caught by the net, they may yet escape under the footrope. Another coefficient (e), equal to the proportion which is retained in the net (as oppose to escaping under the footrope) can also be applied. Both of these coefficients are combined in the whole gear catchability estimate Q , as follows (Somerton 1996):

$$\hat{Q}_i = e + eh \frac{v_{2i}}{v_{1i}} \quad (2)$$

where:

e = proportion in v_{1i} caught

v_{2i} = the area swept by the whole trawl (door spread x distance) minus that swept by the net (see Figure 1) [km²]

h = herding coefficient i.e. the proportion in v_2 herded into v_1 .

An h of 1 and an e of 1 assumes that all fish in the trawl path are caught in the net, this results in a minimum estimate because it implies that all the fish that were caught were present in the wider area swept by the doors (lowest assumed density) and no fish escaped under the footrope. However, estimates of abundance and biomass calculated using these values for cod, haddock and whiting are unlikely to reflect reality as horizontal herding by the trawl doors is known to occur. To account for this a herding coefficient of $h = 0.33$ was applied to the estimates calculated using wingspread and e was assumed to be 1. These values were chosen based on results from a previous study by Walsh (1992) which estimated an overall catchability coefficient of 0.33 for Atlantic cod and a range between 0.24 to 0.44 for all the groundfish caught in that survey (although it is not clear if by this it was meant that either e or h was 0.33). In another study (Reid *et al.* 2007) h was determined to be 0.02 in anglerfish (*Lophius piscatorius*) and we expect h to be higher for the gadoid fish because they are more active swimmers. There were no estimates of variance attributed to the values of h used here, so we have not propagated the variance associated with this component. The final estimates of precision are, therefore, likely to be underestimated. We used the delta method for approximating variance (e.g. Seber, 1982, pages 7-9), resulting in the following:

$$\hat{\text{var}}(\hat{\rho}_{ans}) = \frac{1}{m_s^2} \left\{ \sigma_{ans}^2 \sum_i \frac{1}{(v_{1i} \hat{Q}_i)^2} \right\} \quad (3)$$

where:

σ_{ans}^2 = the sample variance of numbers at age a in stratum s

The number of fish at age a in stratum s was then:

$$\hat{N}_{as} = A_s \hat{\rho}_{ans} \quad (4)$$

and the corresponding variance:

$$\hat{\text{var}}(\hat{N}_{as}) = A_s^2 \hat{\text{var}}(\hat{\rho}_{as}) \quad (5)$$

where:

A_s = area of stratum s [km^2].

The total number of fish at age a was then:

$$\hat{N}_a = \sum_s \hat{N}_{as} \quad (6)$$

and the total variance at age was:

$$\hat{\text{var}}(\hat{N}_a) = \sum_s \hat{\text{var}}(\hat{N}_{as}) \quad (7)$$

Survey precision was expressed as the relative standard error (RSE - Jessen, 1978), according to ICES (2004) as:

$$\text{RSE} = 100\% \times \text{standard error} / \text{estimate} \quad (8)$$

as well as 95% confidence intervals, assuming a normal distribution of estimates.

Weights at length were not taken on the west coast charter. Individual measurements of length and weight were made on the Q1 Scottish V1a IBTS and so species-specific length to whole-weight conversions were derived from these to estimate individual whole fish weights as:

$$\begin{aligned} w &= 0.0048 \times L^{3.2074} && \text{for cod} \\ w &= 0.0047 \times L^{3.196} && \text{for haddock} \\ w &= 0.0032 \times L^{3.2891} && \text{for whiting} \end{aligned}$$

where:

L = length of gutted fish [cm]

The total weight of fish caught in trawl i in stratum s (w_{is}) was then substituted for number (n_{is}) in Equation 1 to produce biomass density (biomass per square kilometre) and the same set of equations (3-7) were then applied to estimate total biomass at age. Total Stock Biomass (TSB) was then obtained by summing all ages.

Proportions at age mature for each species, as used by the ICES assessment working group (ICES, 2009), were as follows:

Age	1	2	3	4+
Cod	0	0.52	0.86	1
Haddock	0	0.57	1	1
Whiting	0	1	1	1

These were used to calculate total spawning biomass at age, and these were summed to give Spawning Stock Biomass (SSB). The same proportions mature at age were used for both males and females.

7.3 Distribution of Fish Densities

The sample locations for 2010 industry science survey are illustrated in Figure 3 (a-c). A total of 76 trawl haul stations were conducted. The raised numbers of fish per hour bottom contact trawling for each species are also shown in this figure, for the industry science survey (a-c) and the Q1 Scottish VIa IBTS (d-f).

Cod densities were very low in all areas in both surveys. The highest densities of cod on the industry science survey were detected within the windsock closed area, and to the north of Northern Ireland. Elsewhere densities were very low, with many zero values throughout the rest of the area.

In the case of haddock, the industry science survey recorded higher densities in the north-eastern part of the survey and the Minch, although reasonably high densities were found throughout the area. The Q1 Scottish VIa IBTS also recorded high densities in the Minch and generally throughout the area, although not so high in the far north east.

Whiting densities were higher in the inshore areas of the Minch on the industry science survey, but were more evenly distributed in the Q1 Scottish VIa IBTS. The catch rates and distribution of catches from the industry-science survey were similar to those from the Q1 Scottish VIa IBTS.

7.4 Estimates of TSB and SSB

Estimates of total stock biomass (TSB) and spawning stock biomass (SSB) from the industry science survey are presented by stratum in Table 1 for cod, haddock and whiting with 95% confidence limits (95% CL) and estimates of the relative standard error (RSE).

Table 7.1: Estimates from the industry science survey of the total stock biomass (TSB) and spawning stock biomass (SSB) in tonnes for cod, haddock and whiting, by stratum, for two values of the herding coefficient h ($h=1$ representing the minimum estimate of biomass, and $h=0.33$ representing the best estimate of biomass) with lower and upper 95% confidence limits (CL) and values of the relative standard error (RSE, see equation 8).

	Strata	Absolute minimum ($h=1$)				Best estimate ($h=0.33$)			
		Biomass (tonnes)	-95% CL	+95% CL	RSE	Biomass (tonnes)	-95% CL	+95% CL	RSE
Cod	Vhigh	752	0	2821	138	1590	0	6041	140
	High	46	0	106	65	95	0	219	65
	Medium	82	20	143	38	168	39	296	38
	Low	481	120	842	38	980	243	1718	38
	TSB	1361	0	3463	77	2833	0	7348	80
	SSB	1021	0	2600	77	2139	0	5532	79
Haddock	Vhigh	4287	485	8089	44	9125	946	17304	45
	High	4558	1653	7462	32	9388	3367	15408	32
	Medium	596	14	1178	49	1245	35	2456	49
	Low	1278	71	2484	47	2713	245	5180	46
	TSB	10718	5750	15687	23	22471	11950	32992	23
	SSB	9267	6086	12449	17	19461	12704	26218	17
Whiting	Vhigh	1230	539	1922	28	2625	1137	4112	28
	High	1102	560	1644	25	2275	1152	3399	25
	Medium	939	453	1425	26	1959	948	2970	26
	Low	1978	982	2975	25	4085	2048	6123	25
	TSB	5250	3835	6664	14	10944	8004	13884	13
	SSB	2234	1755	2714	11	4704	3698	5710	11

The absolute minimum TSB estimate of cod was 1361 tonnes (95% CL between 0 and 3463 tonnes). The minimum SSB estimate was 1021 tonnes (95% CL 0 to 2600 tonnes). The best estimate of TSB, based on $h=0.33$ and $e=1$, was 2833 tonnes (95% CL 0-7348 tonnes) and an SSB of 2139 tonnes (95% CL 0-5532 tonnes). These wide confidence limits are reflected in the high RSEs of between 77 and 80%.

The absolute minimum TSB estimate of haddock was 10718 tonnes (95% CL between 5750 and 15687 tonnes). The minimum SSB estimate was 9267 tonnes (95% CL 6086 to 12449 tonnes). The best estimate of TSB, based on $h=0.33$ and $e=1$, was 22471 tonnes (95% CL 11950-32992 tonnes) and an SSB of 19461 tonnes (95% CL 12704-26218 tonnes). The RSEs for haddock were reasonable, being between 17 and 23 %.

The absolute minimum TSB estimate of whiting was 5250 tonnes (95% CL between 3835 and 6664 tonnes). The minimum SSB estimate was 2234 tonnes (95% CL 1755 to 2714 tonnes). The best estimate of TSB, based on $h=0.33$ and $e=1$, was 10944 tonnes (95% CL 8004-13884) and an SSB of 4704 tonnes (95% CL 3698-5710 tonnes). The RSEs for whiting were good, being between 11 and 14 %.

The biomass estimates from the industry science survey are compared with the most recent assessment estimates for the three species in Table 7.2.

Table 2: Total Stock biomass (TSB) and Spawning Stock Biomass (SSB) estimates from the industry science survey compared with the estimates from the ICES assessment (ICES, 2010).

	TSB (tonnes)		SSB (tonnes)	
	Industry Science Survey Best Estimate (±95% CL)	2010 ICES Assessment	Industry Science Survey Best Estimate (±95% CL)	2010 ICES Assessment
Cod	2833 (0-7348)	10533 (8419-12647)	2139 (0-5532)	6227 (4385-8069)
Haddock	22471 (11950-32992)	19018 (8756-29280)	19461 (12704-26218)	13377 (9065-17689)
Whiting	10944 (8004-13884)	Unknown	4704 (3698-5710)	Unknown

The assessment's estimate of SSB for cod (6227 t) is much higher than the best estimate of SSB from the industry science survey of 2139 t, although the 95% CL do overlap (Table 2). The estimates of TSB are even further apart, and their confidence limits do not overlap (Table 2).

The estimates of haddock are a little closer, estimates of TSB from the assessment were 19018 t compared to an estimate of 22471 from the survey; whilst SSB where a little further apart (19461 t from the survey c.f. 13377 from the assessment).

There were no absolute estimates of biomass given for whiting in the latest ICES assessment.

Estimates of Numbers and Biomass at Age

Estimates of the numbers and biomass at age from the industry science survey are presented in Tables 3-5 for the three species, cod, haddock and whiting.

Table 3: Best estimates ($h=0.33$) of abundance (in millions) and biomass (in tonnes) by age of cod in ICES Division VIa from the industry science survey; RSE = relative standard error; -95% = lower 95% confidence limit; +95% = upper 95% confidence limit.

Age	Abundance	RSE	-95%	+95%	Biomass	RSE	-95%	+95%
0	0	0	0	0	0	0	0	0
1	2.322	78.2	0	5.951	296	72.9	0	728
2	1.008	36	0.282	1.734	797	35.1	238	1357
3	0.044	69.4	0	0.106	107	84.4	0	288
4	0.103	105.8	0	0.321	512	114.2	0	1682
5	0.132	137.6	0	0.494	1120	140.8	0	4275
6	0	0	0	0	0	0	0	0
7+	0	0	0	0	0	0	0	0
Total	3.609	51.9	0	7.351	2833	79.7	0	7348

Table 4: Best estimates ($h=0.33$) of abundance (in millions) and biomass (in tonnes) by age of haddock in ICES Division VIa from the industry science survey; RSE = relative standard error; -95% = lower 95% confidence limit; +95% = upper 95% confidence limit.

Age	Abundance	RSE	-95%	+95%	Biomass	RSE	-95%	+95%
0	0	0	0	0	0	0	0	0
1	42.200	37	11.280	73.120	2306	34	734	3878
2	7.465	27	3.491	11.440	1635	26	776	2494
3	8.463	26	4.122	12.803	2723	26	1307	4139
4	4.349	25	2.158	6.540	1536	25	756	2316
5	28.974	27	13.565	44.383	12101	27	5601	18601
6	1.469	26	0.700	2.239	872	26	413	1332
7	0.551	27	0.254	0.849	385	27	174	597
8	0.723	25	0.360	1.086	513	25	257	769
9	0.080	29	0.034	0.126	61	30	24	98
10	0.222	32	0.082	0.362	165	34	54	275
11	0.165	52	0.000	0.337	173	56	0	364
12	0.000	0	0.000	0.000	0	0	0	0
13+	0.000	0	0.000	0.000	0	0	0	0
Total	94.662	22	53.065	136.259	22471	23	11950	32992

Table 5: Best estimates ($h=0.33$) of abundance (in millions) and biomass (in tonnes) by age of whiting in ICES Division VIa from the industry science survey; RSE = relative standard error; -95% = lower 95% confidence limit; +95% = upper 95% confidence limit.

Age	Abundance	RSE	-95%	+95%	Biomass	RSE	-95%	+95%
0	0	0	0	0	0	0	0	0
1	126.621	30	51.220	202.022	6240	19	3874	8606
2	10.221	16	6.909	13.533	1387	16	945	1830
3	3.877	20	2.363	5.391	1266	20	752	1779
4	1.548	22	0.883	2.212	659	23	351	967
5	1.879	27	0.875	2.884	1104	30	437	1771
6	0.185	24	0.096	0.275	116	26	55	177
7	0.177	29	0.073	0.281	116	34	37	195
8	0.062	42	0.010	0.114	56	42	9	103
9	0.000	0	0.000	0.000	0	0	0	0
10+	0.000	0	0.000	0.000	0	0	0	0
Total	144.571	26	68.647	220.494	10944	13	8004	13884

The estimates of the abundance of cod are very imprecise (RSE ranges between 36 and 138%), but indicate a relatively large number of 5 year olds, which is consistent with the ICES assessment's estimates of the relatively high abundance of the 2005 year class of cod.

Similarly, the haddock estimates of 5 year olds are also high: this is also consistent with estimates from the ICES assessment of the strong 2005 year class of this species.

There are no absolute estimates of abundance at age with which to compare estimates of whiting, but the survey indicates a relatively high abundance of the 2005 year class (5 year olds). This feature of similar year class strengths is not uncommon in gadoid species.

Comparison of Numbers at Age between Surveys

A comparison of the numbers at age for the industry science survey and the Q1 Scottish VIa IBTS are shown in Figures 4-9. The age distributions showed similar patterns for the two surveys, except in the case of cod. Cod at age 1 were most abundant in the industry science survey whereas for the Q1 Scottish VIa IBTS cod at age 2 were the most abundant (Figure 4): this lead to a lack of significance in the correlation between the two survey estimates at age (Figure 5). In both surveys haddock at age 1 and age 5 were the most numerous (Figure 6) and there was a significant correlation between the estimates at age for the two surveys (Figure 7). For whiting the majority of fish were at age 1 in both surveys (Figure 8) and there was a significant correlation between the estimates of the two surveys (Figure 9).

Spatial Distribution of Numbers at Age

The spatial distribution of cod at age for the industry science survey is shown in Figure 10 and in Figure 11 for the Q1 Scottish VIa IBTS. This also shows where the industry science

survey caught most of its 1 year olds (off the northern coast of Northern Ireland) and where the IBTS caught most of its 2 year olds (inshore and to the North of the area); although the numbers caught in the IBTS were very low (maximum of 10 in a half hour tow). Both surveys found large proportions of age 2 fish to the north of the Hebrides.

The spatial distribution of haddock at age for the industry science survey is shown in Figure 12 and in Figure 13 for the Q1 Scottish VIa IBTS. Both surveys show that the young haddock were caught in the Minches and the older haddock (predominantly 5 year old) were caught along the shelf edge. Both surveys showed a very similar pattern in the distribution of numbers at age.

The spatial distribution of whiting at age for the industry science survey is shown in Figure 14 and in Figure 15 for the Q1 Scottish VIa IBTS. The spatial patterns of whiting at age are also consistent between the two surveys, with young whiting in the Minches and the southern part of the area, and older whiting in the north west along the shelf edge.

Discussion

Catchability

The absolute estimates reported here are sensitive to the estimates of catchability (Q). A herding coefficient of 1 assumes that all fish present in the path between the doors are caught. A net retention coefficient of 1 assumes that all the fish that were in the path of the net were caught. This means that all the fish caught were derived from the largest of the sampled areas which results in the lowest possible density from the various catchability options, and, therefore, the lowest possible [minimum] biomass. The lower confidence limit on these minimum estimates then represent what the minimum biomass could possibly be given the sampling process behind the survey. The minimum confidence limit values are, in the case of cod, extreme (zero), largely due to the highly variable nature of the cod estimates.

There is not much data available on the individual herding coefficients for the three species in response to the gear used. The estimates presented here use a herding coefficient of 0.33 and a footrope retention coefficient of 1 for all three species, based on the study by Walsh (1992). These values are reasonable assumptions in the absence of any further knowledge and estimates using these are considered to be the 'best' estimate but they should be treated with some caution. Net retention, for example, is almost certainly a function of fish size and so e would have to be applied to the size of fish to be properly applied. It would clearly be preferable to conduct some specific behavioural studies to examine the reactions to the trawl gear by the different species. Such studies would include catching bag experiments of the kind employed by Walsh (1992) and either video observations or sweep length alteration experiments as described by Somerton (1996).

There are other components of catchability that have not been considered here, in particular, the effect of vertical herding. Vertical herding increases the effective fishing height of a trawl, which occurs when fish dive into the path of the trawl in response to the sound produced by the vessel above (Somerton *et al.*, 1999). However, such vertical herding should be noticeable on echosounder records. Previous echosounder studies of gadoids in this area have indicated that there are very few fish in the water column at the kind of abundance levels that exist here.

Comparison of Biomass Estimates

Despite these reservations about catchability, there is some comfort to be gained from the comparison of the absolute estimates of haddock with the estimates of biomass from the ICES assessment of haddock (Table 2). The total stock biomass estimates are similar and both estimates are contained within both of the error bounds of the estimates. The spawning stock biomass estimates are less compatible, but both estimates still lie within each others' bounds of error.

In the case of whiting there is at present no absolute estimate of abundance from the assessment with which to compare.

The case of cod is less convincing: the survey estimates are lower than the assessment values. Although the bounds of error overlap in the case of SSB, neither estimate lies within neither bound of error, despite the survey error being very large (RSE of 80%).

Numbers at Age

The numbers at age compare very favourably with those calculated from the assessments (Tables 3-5; Figures 4, 6, and 8). In the case of cod, the industry science survey relative numbers at age are more similar to the relative assessment values than the IBTS indices. The industry science survey relative numbers at age are significantly correlated (correlation coefficient $r=0.97$, $p<0.05$) with the relative assessment numbers at age, whereas the relative IBTS index at age are not ($r=0.64$, $p=0.12$). This is rather unexpected given that the IBTS data are actually the primary data used to assess the stock.

In the case of haddock and whiting, the relative numbers at age are similar for all ages for all three indices.

The comparison between the industry science survey and the IBTS survey is fairly favourable, particularly for haddock and whiting. This is the case when looking at the absolute numbers per hour that were caught and their distribution (Figure 3), as well as the relative numbers at age (Figures 5, 7 and 9); and the spatial distribution of numbers at age (Figures 10-15). The only major difference was in the cod, where the industry science survey caught more 1 year olds and the IBTS caught more 2 year olds. However, in both

cases these estimates are based on very few samples, particularly in the case of IBTS, where the maximum number of cod caught in a half hour tow was 10.

Conclusions

The 2010 industry charter survey has provided absolute minimum estimates of abundance for the three main gadoid species in ICES division VIa. In the case of haddock the estimates are within the bounds of error of the estimates from the ICES assessments; in the case of cod, the estimates are lower than the ICES assessments and lower than the lower bounds of error in the ICES assessments. In the case of whiting there is no assessment to compare to, but the estimates provided here may be useful to evaluate any future assessment of this stock.

The industry science survey provided very similar results to the IBTS survey which was carried out at the same time. The age compositions were very similar in the case of haddock and whiting. In the case of cod, the age compositions were not significantly correlated: the industry science survey caught more 1 year olds, whilst the IBTS caught more 2 year olds. Both surveys caught very few cod, however, and the estimates of cod from the industry science survey were very imprecise.

The relative numbers and spatial distributions from the survey do not differ greatly from those provided by the IBTS survey for haddock and whiting. In the case of cod, both surveys suffer from the very low numbers encountered and the industry science survey estimates were imprecise. There is some implication that the survey may have performed slightly better than the IBTS in terms of the similarity to the assessment. With this in mind the IBTS survey has been re-designed to take into account the principles used and the results obtained in the industry science survey (a random stratified design, using appropriately designed strata).

7.5 References

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7.6 Figures

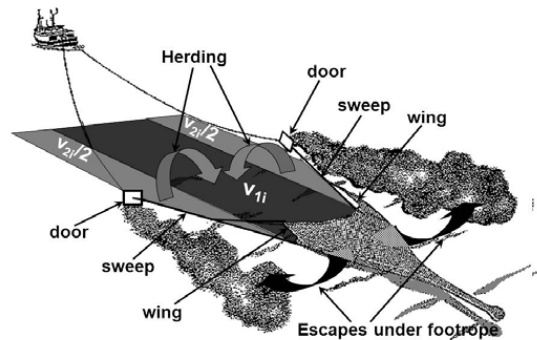


Figure 7.1: Illustration of a demersal trawl, showing the sampling areas swept by the whole gear, i.e. the doors and the wings. The area swept by the wings is indicated by v_{1i} and the area swept by the sweeps by v_{2i} ; the total area swept by the doors is therefore $v_{1i} + v_{2i}$. The fish density is dependent on the catchability of the trawl, which in this case is considered by movement of fish herded by the doors into the path of the net (grey arrows) and escapement of fish under the footrope (dark black arrows).

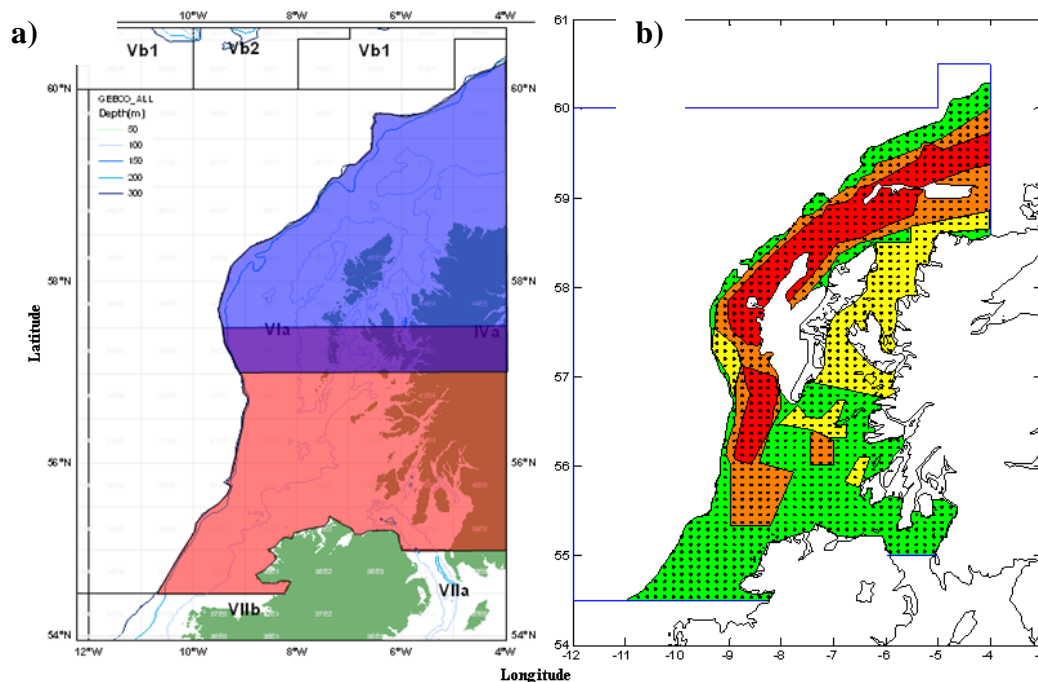


Figure 7.2: a) Map of the west of Scotland showing ICES Division VIa and the two areas surveyed by the industry science charter vessels: MFV Carina surveyed the northern part of the area (shaded blue) and MFV Genesis surveyed the southern part of the area (shaded red); the purple area was an area of overlap between the two. b) The strata identified by fishermen for the design of the west coast survey: Red = Very high abundance; Orange = High; Yellow = Medium; Green = Low; White = Rocky. Black dots represent potential sampling points from which a random selection was made within strata to provide the samples shown in Figure 7.3. The area shaded white to the west of the outer Hebrides is an area of rocky ground unsuitable for trawling: this was not sampled and not included as part of the survey area.

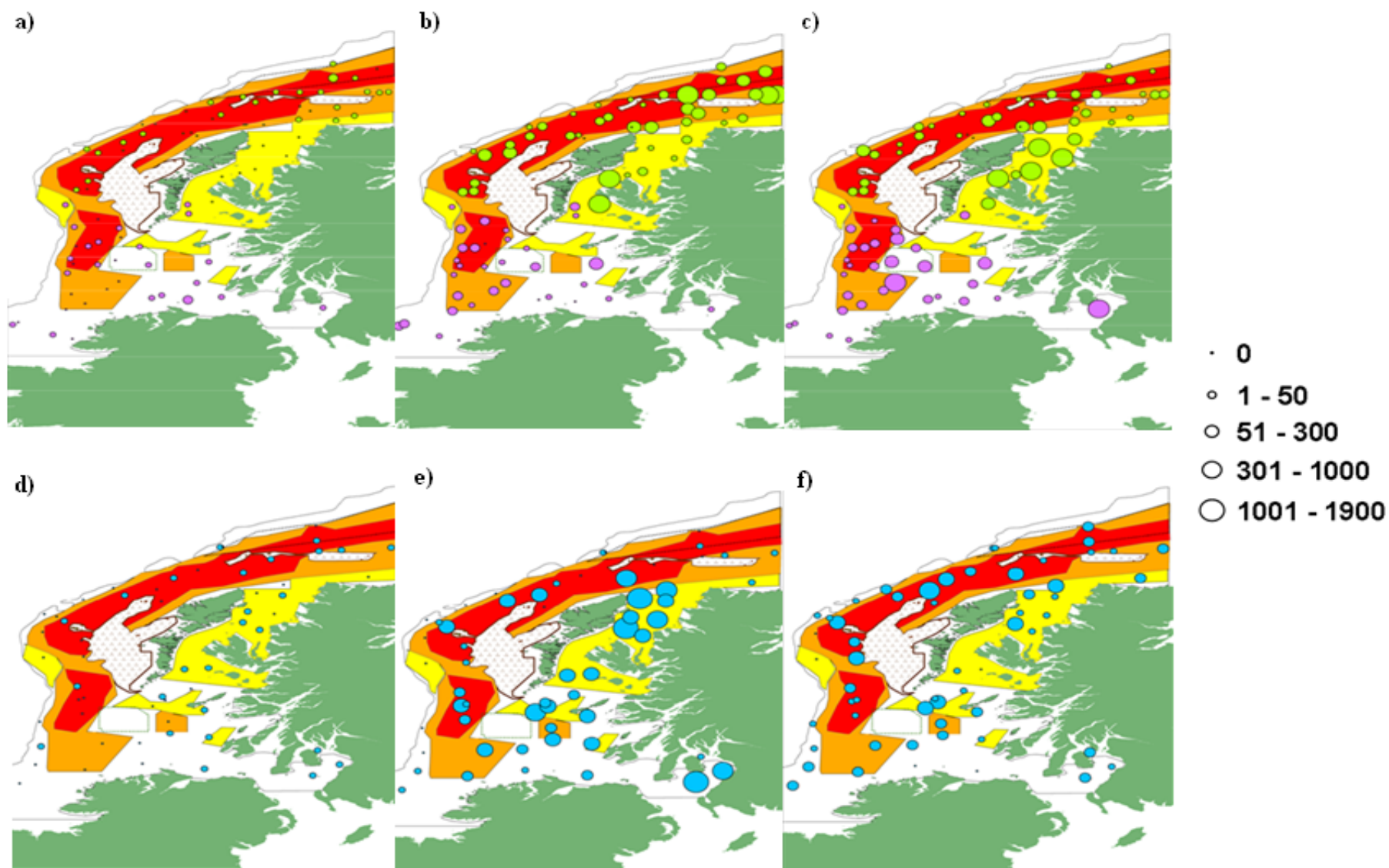


Figure 7.3: Maps of the seas to the west of Scotland with filled circles indicating the position of the samples taken. The area of the circle is proportional to the total numbers of fish caught per hour, according to the legend. Upper Panels - Results from the industry-science survey above, in purple (southern survey) and green (northern survey) for: a) cod; b) haddock; c) whiting; Lower Panels - Results from the IBTS survey (in blue) for: d) cod; e) haddock; and f) whiting. The shaded polygons indicate the strata definitions as described in Figure 7.2b.

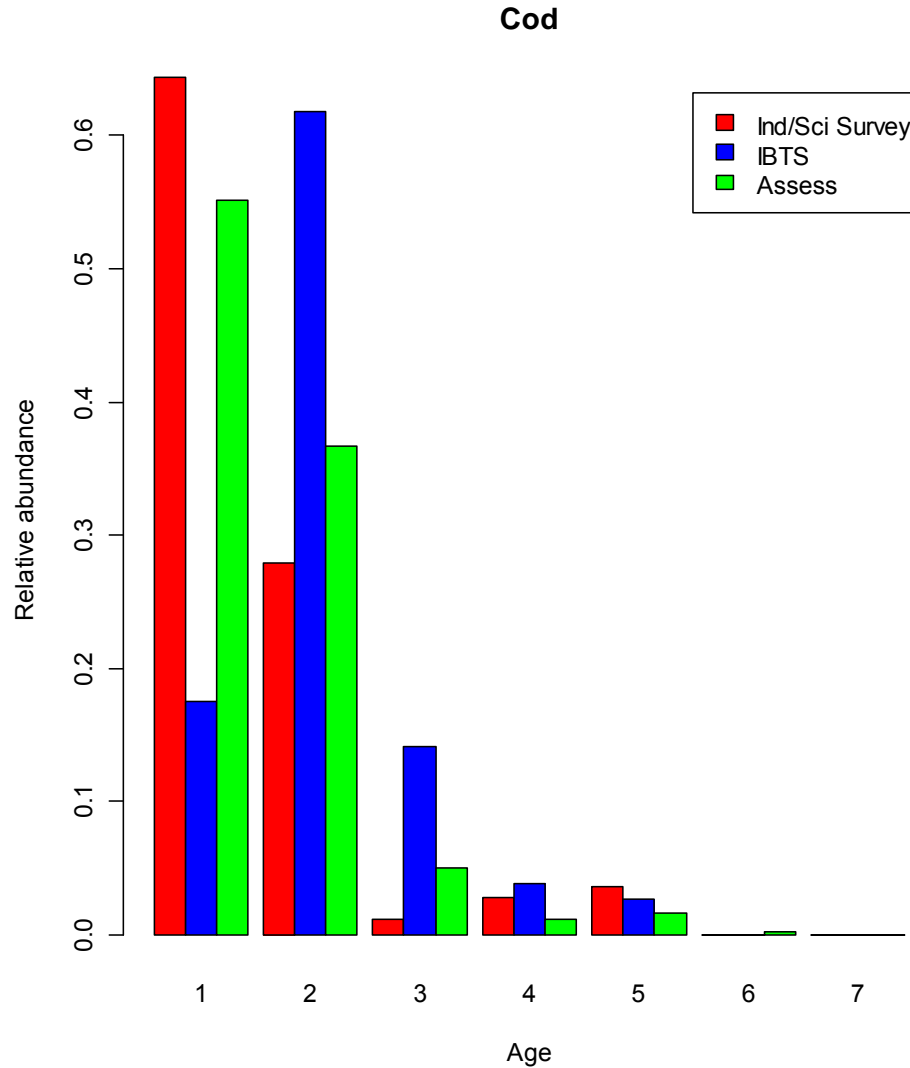


Figure 7.4: Estimates of the abundance at age of cod in 2010, on a relative scale, from the industry science survey (red bars), Q1 Scottish Vla IBTS survey (blue bars), and the 2010 ICES assessment (green bars).

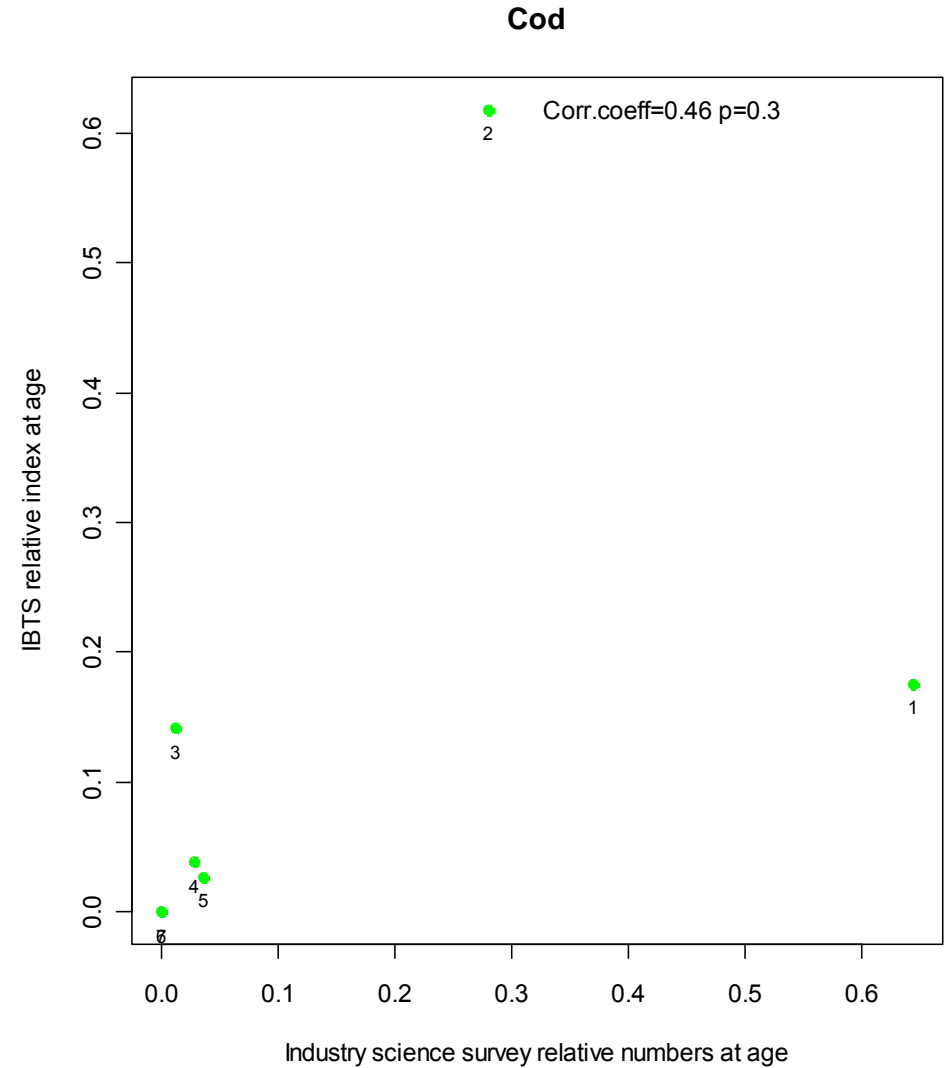


Figure 7.5: Comparison of the estimates of the abundance at age of cod in 2010, on a relative scale, from the industry science survey and Q1 Scottish Vla IBTS survey, with statistics of the correlation coefficient. Numbers at each point represent the age.

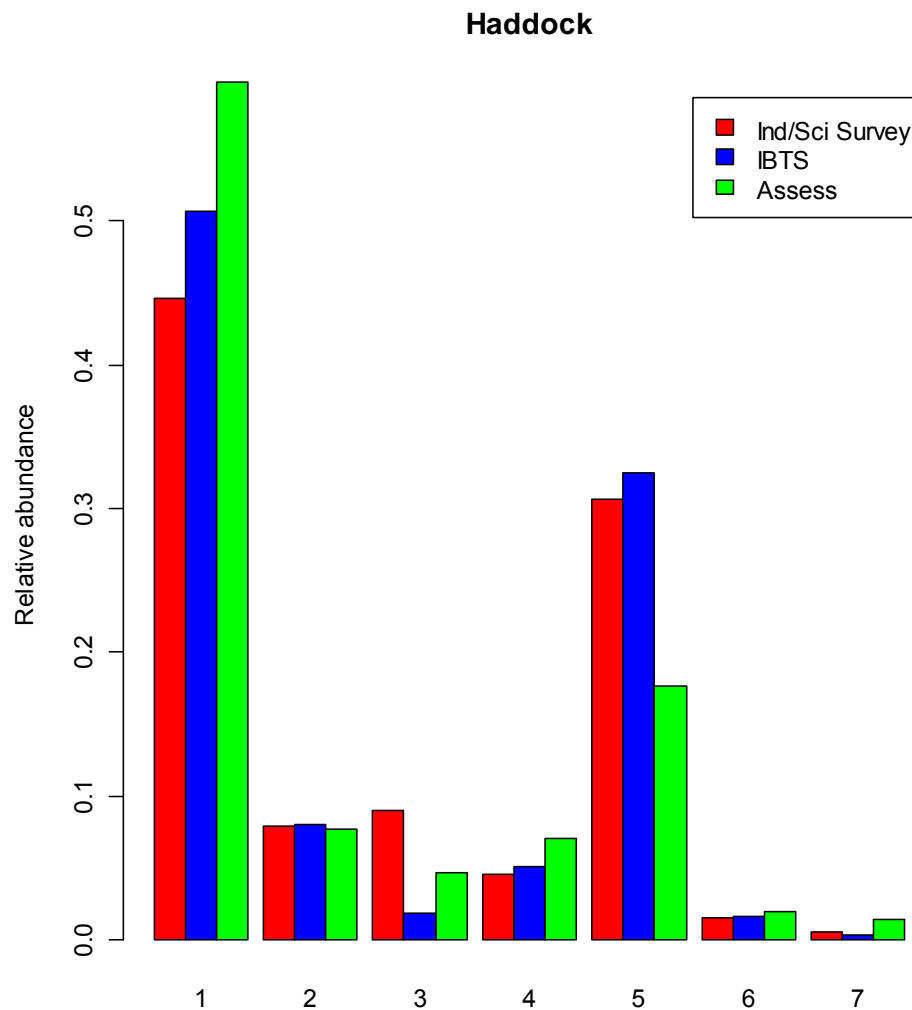


Figure 7.6: Estimates of the abundance at age of haddock in 2010, on a relative scale, from the industry science survey (red bars), Q1 Scottish Via IBTS survey (blue bars), and the 2010 ICES assessment (green bars).

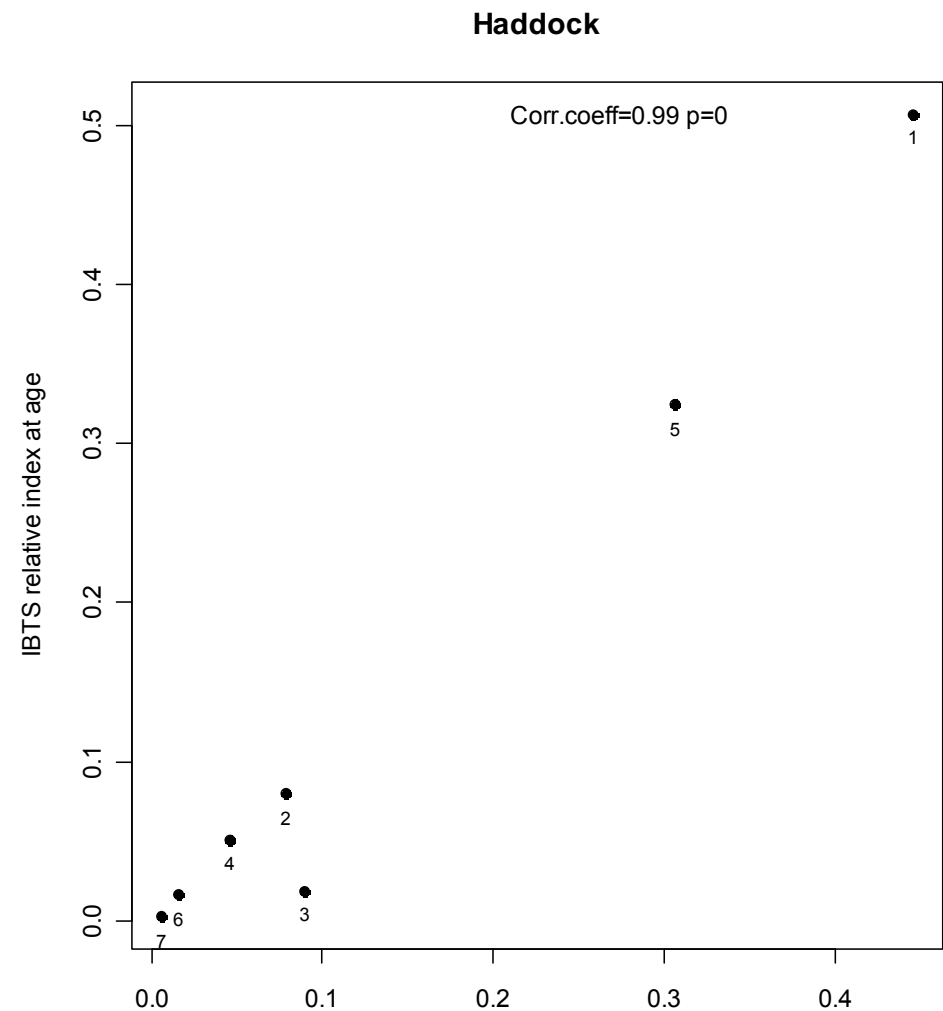


Figure 7.7: Comparison of the estimates of the abundance at age of haddock in 2010, on a relative scale, from the industry science survey and Q1 Scottish Via IBTS survey, with statistics of the correlation coefficient. Numbers at each point represent the age.

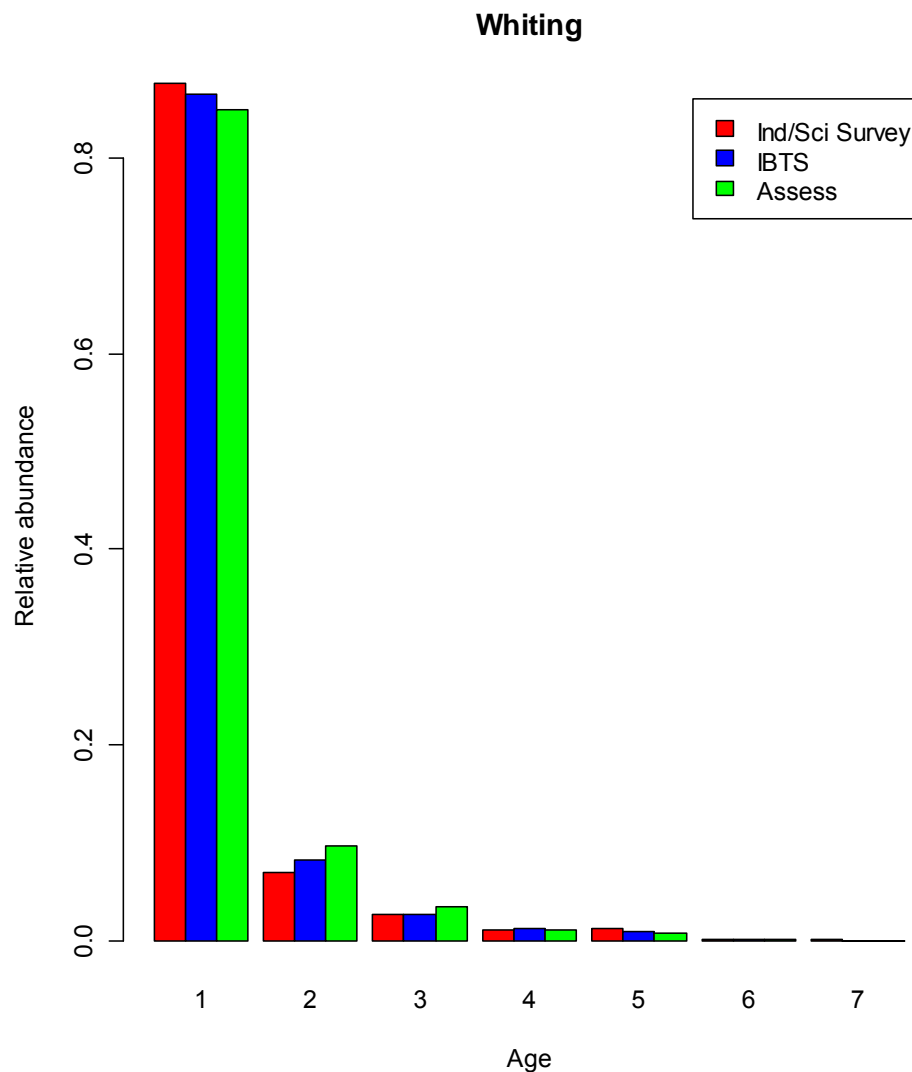


Figure 7.8: Estimates of the abundance at age of whiting in 2010, on a relative scale, from the industry science survey (red bars), Q1 Scottish V1a IBTS survey (blue bars), and the 2010 ICES assessment (green bars).

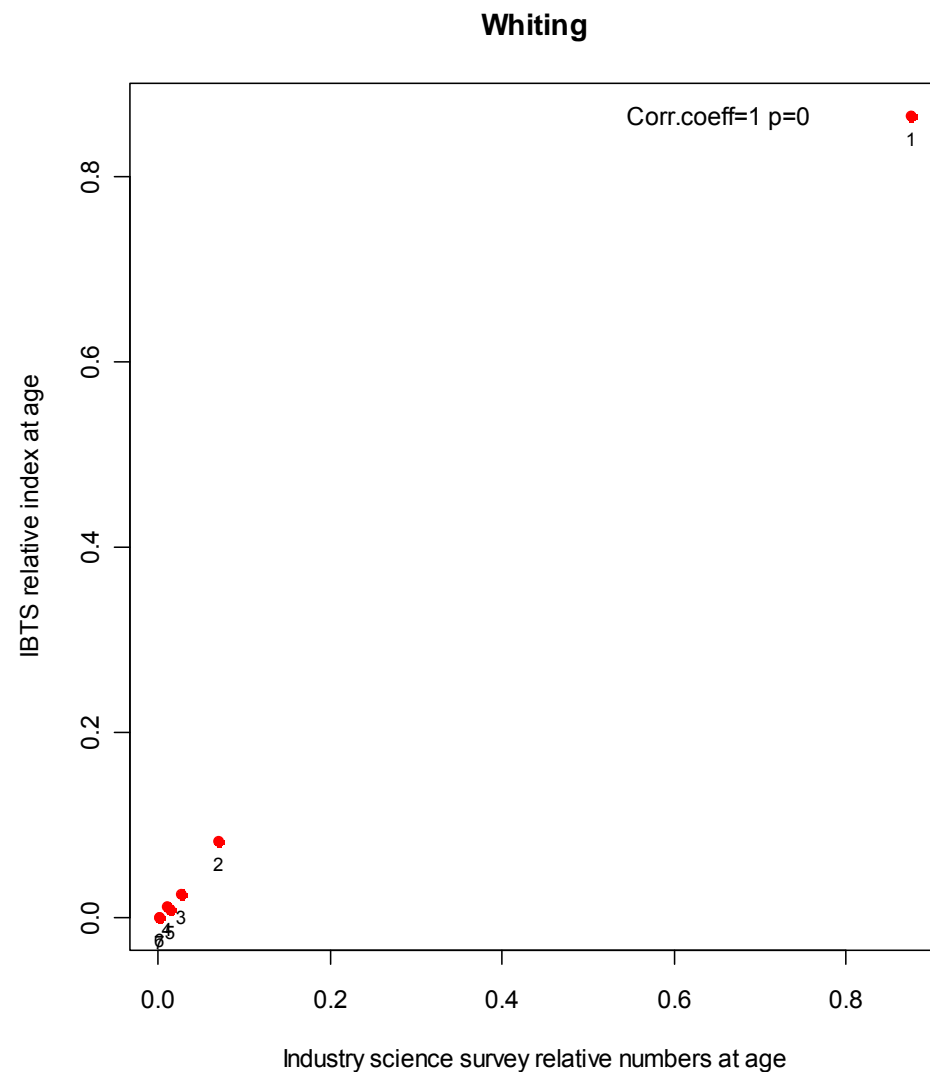


Figure 7.9: Comparison of the estimates of the abundance at age of whiting in 2010, on a relative scale, from the industry science survey and Q1 Scottish V1a IBTS survey, with statistics of the correlation coefficient. Numbers at each point represent the age.

Cod ind/sci survey

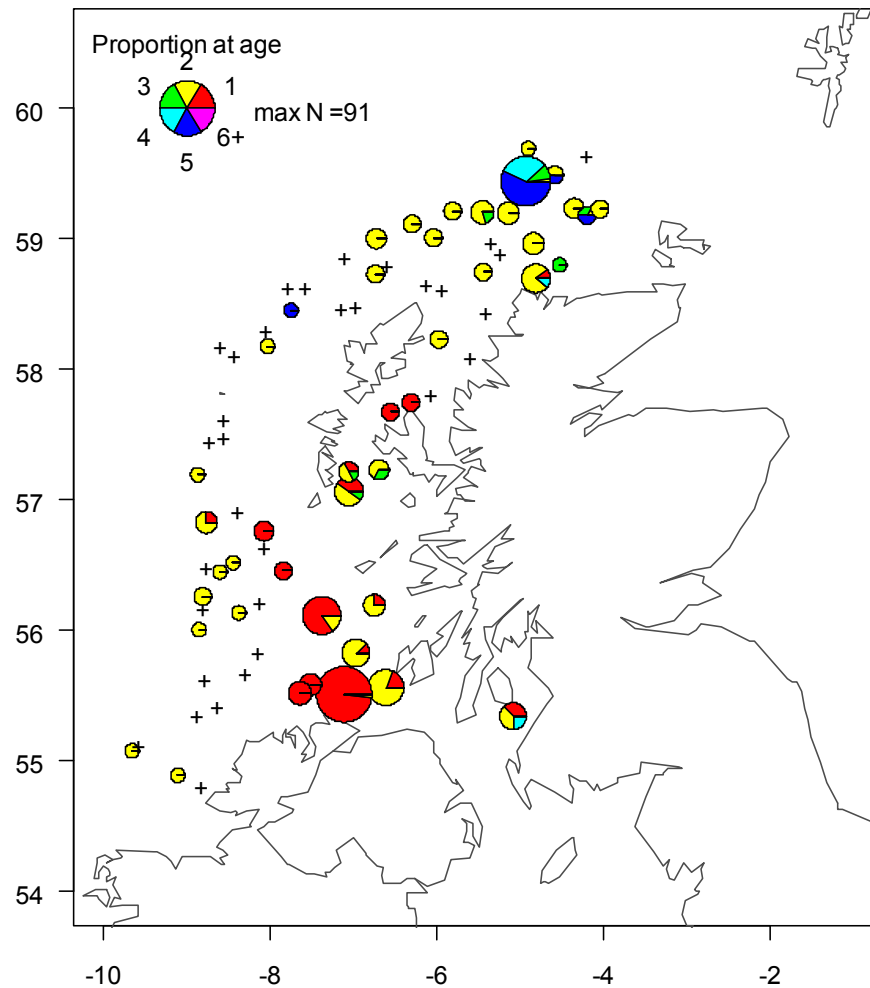


Figure 7.10: Map of the seas to the west of Scotland with pie charts indicating the position of the samples taken during the industry science survey (crosses indicate zero catches). The area of the pie chart is proportional to the total numbers of cod caught on a cube root scale relative to the maximum number caught as indicated (max N); and the pie segments correspond to the proportions at age according to the legend.

Cod IBTS

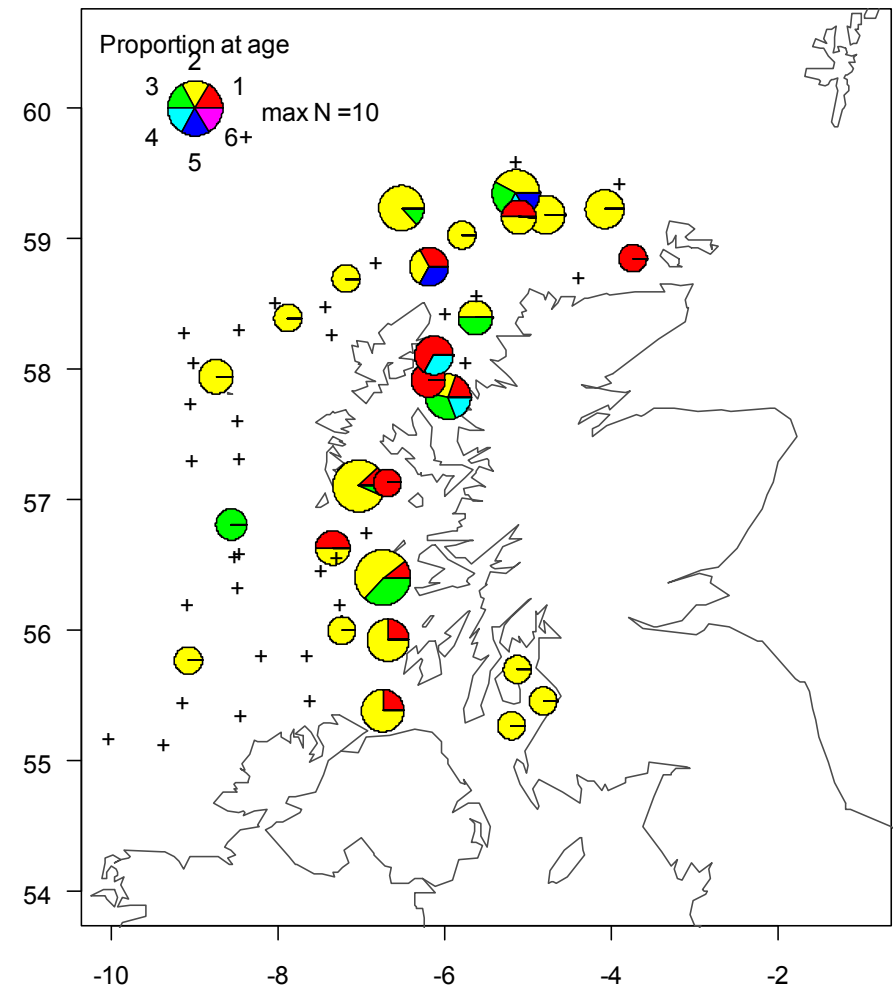


Figure 7.11 Map of the seas to the west of Scotland with pie charts indicating the position of the samples taken during the Q1 Scottish V1a IBTS (crosses indicate zero catches). The area of the pie chart is proportional to the total numbers of cod caught on a cube root scale relative to the maximum number caught as indicated (max N); and the pie segments correspond to the proportions at age according to the legend.

Haddock ind/sci survey

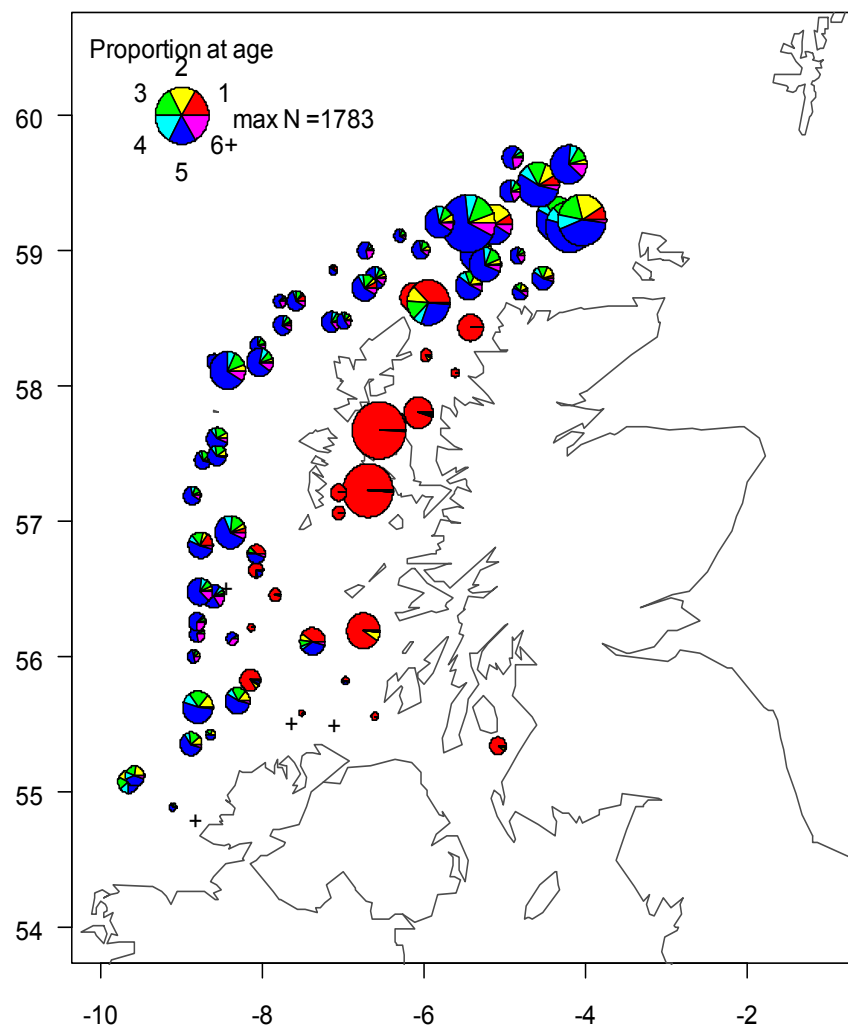


Figure 7.12 Map of the seas to the west of Scotland with pie charts indicating the position of the samples taken during the industry science survey (crosses indicate zero catches). The area of the pie chart is proportional to the total numbers of haddock caught on a cube root scale relative to the maximum number caught per tow as indicated (max N); and the pie segments correspond to the proportions at age according to the legend.

Haddock IBTS

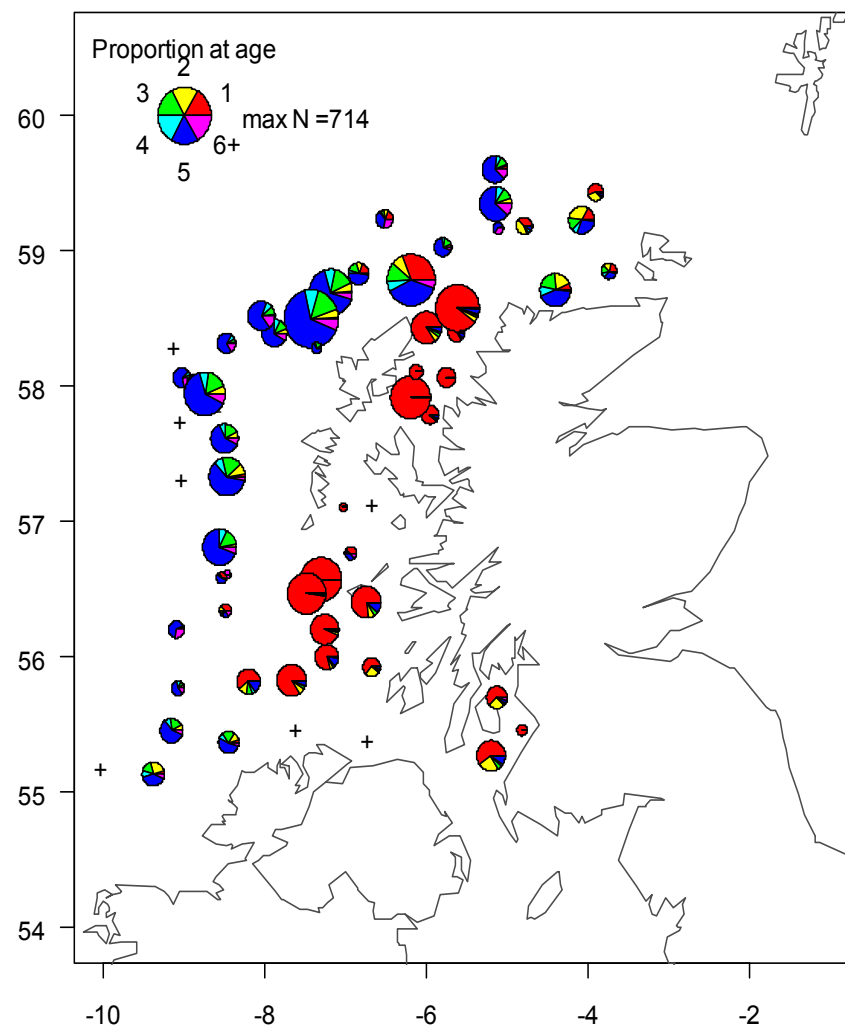


Figure 7.13 Map of the seas to the west of Scotland with pie charts indicating the position of the samples taken during the Q1 Scottish V1a IBTS (crosses indicate zero catches). The area of the pie chart is proportional to the total numbers of haddock caught on a cube root scale relative to the maximum number caught per half hour tow as indicated (max N); and the pie segments correspond to the proportions at age according to the legend.

Whiting ind/sci survey

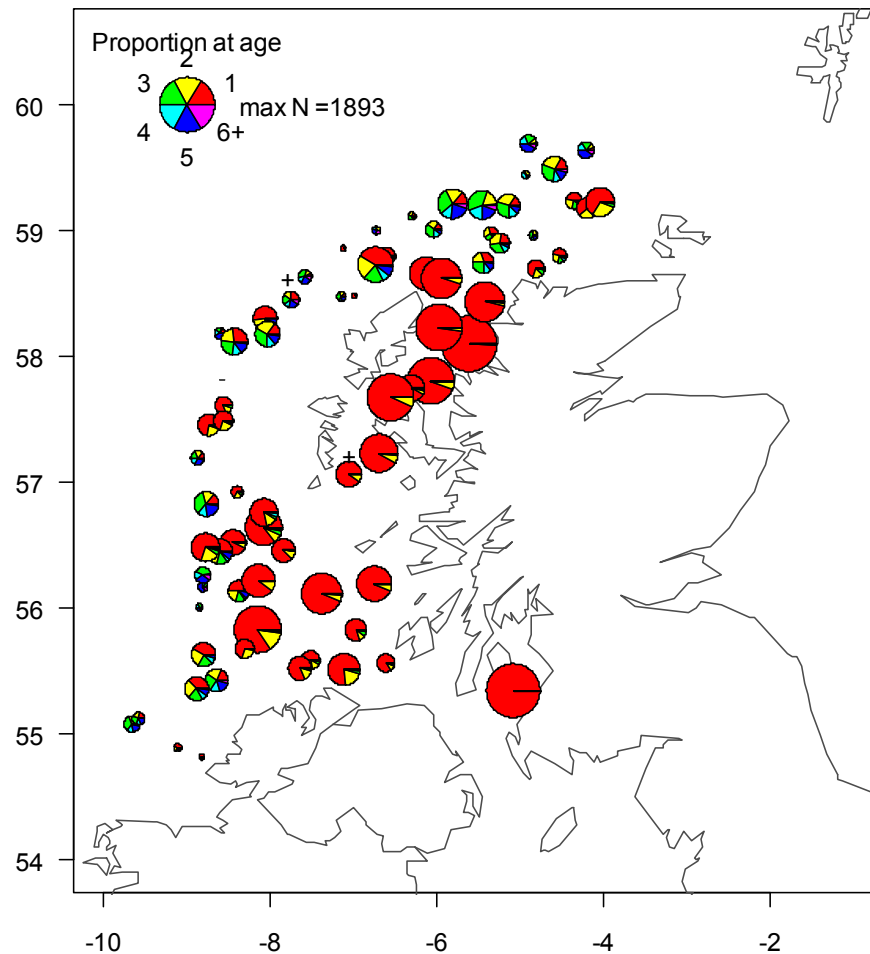


Figure 7.14 Map of the seas to the west of Scotland with pie charts indicating the position of the samples taken during the industry science survey (crosses indicate zero catches). The area of the pie chart is proportional to the total numbers of whiting caught on a cube root scale relative to the maximum number caught per tow as indicated (max N); and the pie segments correspond to the proportions at age according to the legend.

Whiting IBTS

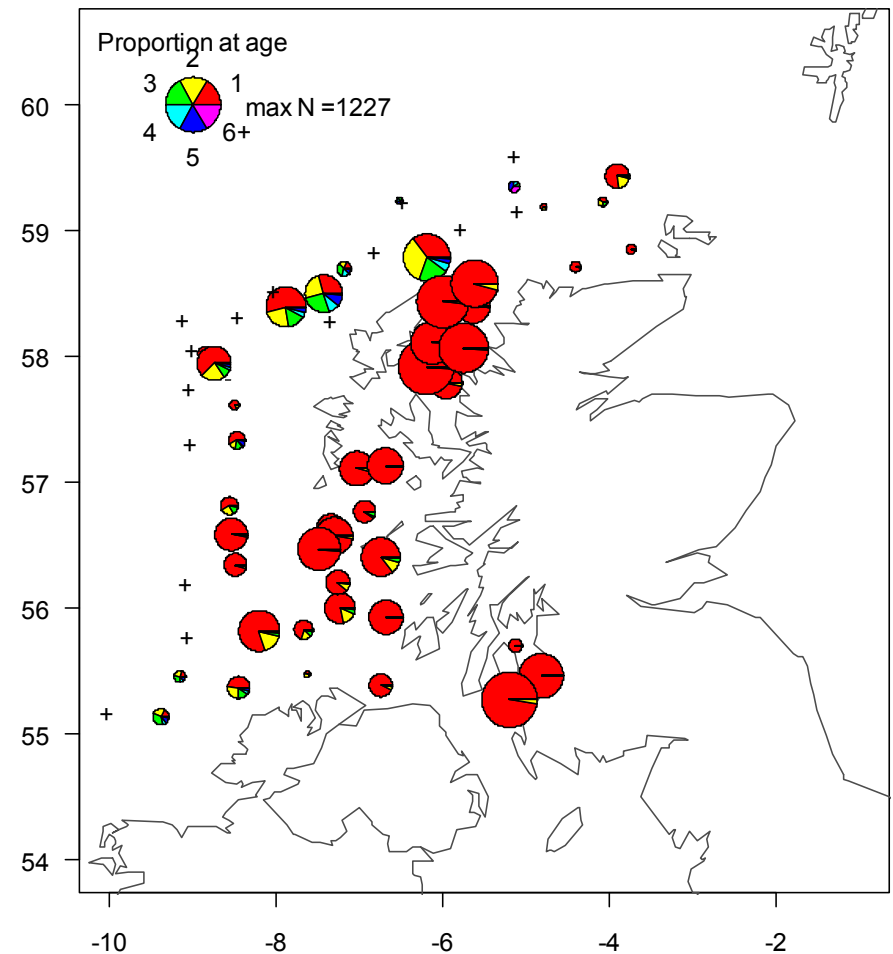


Figure 7.15 Map of the seas to the west of Scotland with pie charts indicating the position of the samples taken during the Q1 Scottish V1a IBTS (crosses indicate zero catches). The area of the pie chart is proportional to the total numbers of whiting caught on a cube root scale relative to the maximum number caught per half hour tow as indicated (max N); and the pie segments correspond to the proportions at age according to the legend.

8 Modelling the West Coast Ecosystem – Ecopath with Ecosim

G Tyldesley, S Magill, S Heymans and S J Holmes

8.1 Introduction

8.1.1 The State of the Stocks

The importance of the waters off the West Coast of Scotland to the Scottish fishing industry is undoubted. Cod, haddock and whiting contribute a considerable part of the value of Scottish demersal landings from this area. Other countries also have important fisheries on the West Coast with the bulk of the saithe, hake, ling, blue whiting and horse mackerel catches being landed outwith Scotland. Despite the economic importance of the fisheries here to a number of countries, the West Coast of Scotland is not studied to the same degree as the North Sea. As a result there is less certainty in the science behind the assessments of stock abundance and the health of the ecosystem is less well understood.

The aims of this project are twofold. Firstly we wish to determine whether the output of current stock assessment models for cod, haddock and whiting are compatible with our understanding of the ecosystem within which these species reside. Secondly, we will explore what interactions within the ecosystem could be leading to these three key demersal stocks behaving as they are. In order to address these issues we will use an ecotrophic food web model constructed with the software package Ecopath with Ecosim (EwE).

Cod and haddock on the West Coast (ICES area VIa) are both assessed annually (ICES 2009). The whiting stock was assessed annually until 2007 (WGNSDS 2007) at which point it was decided that the data was not sufficiently reliable to base an assessment on. Up until 2004 a full stock assessment was carried out (WGNSDS 2004) and between 2004 and 2007 the stock assessment was a survey based assessment (SURBA) resulting only in a time series of index of abundance as opposed to absolute abundance. For these three key species we plot a time series of the total stock biomass (TSB) relative to its 1985 value in Figure 8.1.

Cod biomass shows a steady downward trend over 25 years. The trend in haddock biomass exhibits a degree of fluctuation across the same period (Figure 8.1). Haddock is known to produce high variability in strength of year class. 1999 was a particularly strong year class on the West Coast and the total stock shows a resulting increase in 2000 and a steady decrease thereafter. Whiting, after a huge increase in biomass at the beginning of the 1990's shows steady decrease thereafter. The stock assessments show that all three species were at historically low levels and down to less than 40% of their 1985 values by 2007/2008. Since 2000 all three have been in decline.

The most obvious explanation for these declines is that the stocks are unable to sustain the fishing pressure that has been, and continues to be, applied to them. In Figure 8.2 we plot

the fishing mortality estimated by the stock assessments. This is taken to be the total mortality, Z , minus the natural mortality, M . In the stock assessments M is assumed to remain constant at 0.2 (WGNSSDS, 2004; WGNSSDS, 2007; ICES, 2009).

The fishing mortality on cod stays quite constant whereas the fishing mortalities on haddock and whiting fluctuate year on year without showing any clear trend (Figure 8.2). The fishing mortalities on whiting since 1997 have been mostly very high. Despite these fluctuations we could describe these mortalities as remaining high over all years.

Some of the questions which we seek to address with an EwE model are:

- Is increased seal predation such that gadoid stocks are unable to grow?
- Is the Nephrops fishery catching too many juvenile fish?
- What benefits to gadoid stocks would fishing at F_{MSY} result in?

One of the main hypotheses to explain decline in cod, haddock and whiting stocks is increased predation from grey seals (SCOS, 2008). It is well established that grey seal numbers around much of the British coastline, including the Inner and Outer Hebrides of Scotland, have been on the increase and fishermen argue that the increased predation associated with this rise in seal numbers is sufficient to suppress, in particular, the cod stock. EwE provides us with a good tool to explore this hypothesis and to investigate what impact this increase could have on other species within the ecosystem.

In recent years the Nephrops fishery has become an increasingly important fishery to the Scottish fleet. This fishery operates in areas that are important grounds for juvenile gadoids. Nephrops trawlers catch and discard some of these juvenile fish and it could be that this pressure on the juvenile fish is sufficient to suppress the population as a whole.

As well as top-down explanations for low levels of cod, haddock and whiting abundance there are concerns that bottom-up controls could also affect abundance (Cook and Heath, 2005; Brunel and Boucher, 2007; Heath and Lough, 2007). There are mechanisms which could link climate change and associated changes in ocean currents and nutrient levels to the performance of these stocks. EwE allows for the investigation of such hypotheses in a purely statistical as opposed to mechanistic sense. Levels of phytoplankton abundance (or any other model functional group) can be correlated with any driver which represents these climatic changes and the effects of this correlation can be investigated. Alternatively, a driver could be sought to give the best fit to biomass time series data for the higher trophic levels. The effects of this driver will then be propagated up the food chain and might explain gadoid stock performance.

In the North Sea, the suppression of the cod stock has been linked to herring preying on their eggs (Segers *et al.*, 2006). In order to investigate whether this is the case on the West Coast we would have to resolve the egg and larval stages of cod, haddock and whiting in the model. This entails extra work and for the moment lies outwith the scope of the investigation we will carry out.

Through constructing an EwE model, other possible explanations for the low abundance of cod, haddock and whiting could emerge and these may warrant further investigation.

8.1.2 Ecopath with Ecosim

Ecopath with Ecosim (EwE) is a quantitative tool used to analyse aquatic ecosystems. It combines software for ecosystem trophic mass balance analysis (Ecopath) with a dynamic modelling capability (Ecosim), and also includes a space-time dynamic routine (Ecospace) which can be used to explore past and future impacts of fishing on the environment (Christensen *et al.*, 2005).

The ecosystem as modelled is represented by functional groups (i), which can be composed of species, groups of species with ecological similarities or ontogenetic fractions of a species. Ecopath uses two equations to parameterize models (Christensen *et al.*, 2005) to describe;

- 1) the production of each group
- 2) the energy balance of each group

Production

The production of each group is estimated using the equation (Christensen *et al.*, 2005):

$$\text{Production} = \text{catch} + \text{predation mortality} + \text{biomass accumulation} + \text{net migration} + \text{other mortality} \quad (1)$$

or, more formally

$$P_i = Y_i + B_i \cdot M2_i + E_i + BA_i + P_i \cdot (1 - EE_i) \quad (2)$$

where P_i is the total production of group i , Y_i is the total fishery catch rate of i , B_i is the total biomass of the group i , $M2_i$ is the instantaneous predation rate for group i , E_i the net migration rate (emigration - immigration), BA_i is the biomass accumulation rate for i , and $P_i(1-EE_i)$ is the 'other mortality' rate for i (Christensen *et al.*, 2005). Equation (2) can be re-written as:

$$B_i \cdot (P/B)_i \cdot EE_i - \sum_{j=1}^n B_j \cdot (Q/B)_j \cdot DC_{ji} - Y_i - E_i - BA_i = 0 \quad (3)$$

where $(P/B)_i$ is the production/biomass ratio for i and under most conditions corresponds to the total mortality rate, Z , commonly estimated as part of fishery stock assessment. EE_i is the ecotrophic efficiency of group i , describing the proportion of the production that is utilised in the system, $(Q/B)_j$ is the consumption/biomass ratio of the predator j and DC_{ji} is the fraction of prey i in the average diet of predator j (Christensen *et al.*, 2005).

Energy Balance

The energy balance within each group is ensured when consumption by group i equals the production by i , plus respiration by i and unassimilated food by i . The units of the model are expressed in terms of nutrient or energy related currencies, and by a unit of surface.

Frequently biomass is expressed as t km^{-2} and production and consumption is expressed as $\text{t km}^{-2} \text{ yr}^{-1}$.

Ecosim is the dynamic expression of the ecosystem over time and is defined by a series of differential equations:

$$\frac{dB_i}{dt} = g_i \sum_j Q_{ji} - \sum_j Q_{ij} + I_i - (M_i + F_i + e_i) B_i \quad (4)$$

where dB_i/dt is the growth rate during time t of group i in terms of its biomass B_i , g_i is the net growth efficiency of group i , M_i is the non-predation 'other' mortality rate, F_i is the fishing mortality rate, e_i is the emigration and I_i is immigration rate (Christensen *et al.*, 2005).

The $\sum Q_{ji}$ expresses the total consumption by group i and is calculated based on the foraging arena concept, where B_i 's are divided into vulnerable and invulnerable components (Walters *et al.*, 1997). $\sum Q_{ij}$ indicates the predation by all predators of group i (Christensen *et al.*, 2005).

The transfer rate (v_{ij}) between the vulnerable and invulnerable components sets the top-down or bottom-up control of each interaction (Christensen *et al.*, 2005). For each predator prey interaction the consumption rate is calculated by:

$$C_{ij} = \frac{a_{ij} \cdot v_{ij} \cdot B_i \cdot P_j \cdot T_i \cdot T_j \cdot S_{ij} \cdot M_{ij} / D_j}{v_{ij} + v_{ij} \cdot T_i \cdot M_{ij} + a_{ij} \cdot M_{ij} \cdot P_j \cdot S_{ij} \cdot T_j / D_j} \quad (5)$$

where, a_{ij} is the effective search rate for predator i feeding on a prey j , v_{ij} is the base vulnerability expressing the rate with which prey move between being vulnerable and not-vulnerable, B_i is prey biomass, P_j is predator abundance, T_i represents prey relative feeding time, T_j predator relative feeding time, S_{ij} user-defined seasonal or long term forcing effects, M_{ij} mediation forcing effects, and D_j represents handling time as a limit to consumption rate (Christensen *et al.*, 2005).

Ecopath with Ecosim requires three of the following four data points for each group entered into the model:

- Biomass (B , t km^{-2}) for the year under consideration;
- Production/Biomass ratio (P/B , year^{-1});
- Consumption/Biomass ratio (Q/B , year^{-1});
- Ecotrophic Efficiency (proportion). This parameter indicates the unexplained mortality for each group and is often set to 95% when estimating the biomass.

In addition for each group the diet composition is required as a contribution (in mass) of the prey items in the diet of the group and for each fishery the group specific landings ($\text{t km}^{-2} \text{ year}^{-1}$) and discards ($\text{t km}^{-2} \text{ year}^{-1}$) are required. To run the dynamic simulations in Ecosim yearly estimates of biomass, fishing mortality, and catch by species and/or gear is required to drive the model.

8.2 The Model

8.2.1 Model Specification

This project focuses on cod, haddock and whiting. We seek to investigate what could be causing the decline in abundance of these species on the West Coast of Scotland that we highlighted above. In order to do this we intend to use the ecosystem modelling software Ecopath with Ecosim (EwE). Here we lay out the main criterion by which we will construct the model.

Given our focus, we choose to exclude areas of deep sea from the model. Analysis of the catch per unit effort (CPUE) of cod, haddock and whiting from the Scottish V1a International Bottom Trawl Survey (Sco.V1a.IBTS) reveals that abundance declines sharply at the shelf edge. We assume that ecosystem interactions outwith spatial areas in which cod, haddock and whiting are found in any abundance will have little effect on their populations. We choose our model area to be the shelf, which we define as all area above the 200m contour, of ICES area V1a. All species that are only found off the shelf are excluded. We calculate this area to be approximately $110,000 \text{ km}^2$.

Cod, haddock, whiting

In order to answer questions about cod, haddock and whiting, we choose to resolve their life history to a greater level than other species in the model. Within EwE it is possible to include any number of “stanzas” or life history stages for each species. As well as defining the weight and age at which the stanza is reached and the asymptotic weight, we need to provide parameters which define the biomass and consumption rate of the mature stage, as well as the total mortality, diet composition, consumption rate, the predators and fishing pressure for each stanza (Christensen *et al.*, 2005). In order to answer questions specific to fishing pressure and seal predation on these three species we decided that resolving each species into two stanzas, juveniles and adults, provides sufficient detail without requiring parameters beyond our knowledge. We state that maturity is reached at 24 months for all three species. Further parameterisation and justification is detailed below for each species.

Start Year – 1985

Inferences made from EwE model output are most reliable when the data used to calibrate the model include time series data, specifically indexes of biomass and values of catch. The stock assessments for cod and haddock run from 1978 to the present. The most recent

whiting assessment starts in 1985 and runs to 2007. These assessments provide time series of biomass and catch against which we can calibrate the model and time series of fishing mortality to drive it. As an alternative to calibrating the model to the stock assessment data we could calibrate the model against time series of CPUE from the Sco.Vla.IBTS. The first quarter Q1 Scottish Vla IBTS (Q1 Sco.Vla.IBTS) started in 1985 and is ongoing. This source of data also provides a relative biomass time series for non-assessed species. Time series of grey seal abundance, the growth of which is a potential cause of gadoid decline, is based on pup surveys which have been conducted annually in the Inner and Outer Hebrides since 1984.

Given the model dependence on time series data and the fact that 1985 is a year from which all the time series detailed run concurrently we choose 1985 for the start of the model.

Functional Groups

Our choice of model functional groups (Table 8.1) is based on those of a previous Ecopath model for the West coast of Scotland by Hagan and Pitcher (2005). We included extra groups where there were sufficient data to support them or if they were key species in answering the questions that we laid out earlier. Groups that were deemed to be less relevant were merged into larger functional groups.

Table 8.1: Functional groups in West Coast of Scotland Shelf EwE model. (* - multi-stanza groups). See Appendix A for a description of all species in each functional group.

Fish	Mammals and birds	Invertebrates
Cod *	Grey seals	Cephalopods
Haddock *	Harbour seals	Nephrops
Whiting *	Cetaceans	Other crustaceans
Anglerfish	Birds	Epifauna
Flatfish		Infauna
Pollock and saithe		Large zooplankton
Sharks		Small zooplankton
Rays and skates		Phytoplankton
Gurnards		Detritus
Poor cod		
Other demersal fish		
Sandeel		
Sprat		
Mackerel		
Horse mackerel		
Blue whiting		
Norway pout		
Herring		
Other pelagic fish		
Other benthopelagic fish		

8.2.2 General Parameterisation of Fish Functional Groups

Functional Group Composition

Most of the functional groups are made up of a number of individual species as detailed in the Appendix. In order to calculate the diet composition and the consumption and production values of these groups we need to define the proportion of the total biomass each species contributes to the group. Weighted averages can then be arrived at for these parameters. We use ratios of CPUE from the Q1.Sco.Vla.IBTS to define these proportions.

The functional groups should be defined such that the species composition of each group stays roughly constant over the period of study. Species which show divergent trends in CPUE should generally not be grouped together. However if species which do exhibit divergent trends in biomass are otherwise similar in terms of their interaction with the rest of the ecosystem (i.e. comparable diet composition and production and consumption values) the species could still be included in the same functional group. Given that we used this methodology for grouping species, we attribute any variation in the proportions of CPUE to noise and we take an average species composition over all years in the period of study.

Biomass

For many species of commercial importance, annual stock assessments are carried out or have been carried out for part of the period of study. These stock assessments are based on a combination of age-resolved commercial and survey data. The assessments are conducted over spatial areas that are deemed to enclose distinct populations. These areas do not necessarily correspond directly to the shelf of ICES area VIa (e.g. mackerel is assessed as a North East Atlantic stock (WGWIDE, 2009) and the recent saithe assessment includes ICES area VI with IV and IIIa (WGNSSK, 2009)). The assessments which do not correspond directly to our area of study require further analysis and we describe each case below.

For non-assessed species we use the method of Sparholt (1990) to raise CPUE from the Scottish VIa IBTS to absolute biomass. This method of calculating biomass for non-assessed species was used in a recent North Sea EwE model (Mackinson and Daskalov, 2007). The assertion made is that for each non-assessed species there is an assessed species which exhibits similar catchability. We term these species “reference species”. The absolute biomass ($Biomass_i$) of the non-assessed species, i , is then given by:

$$Biomass_i = Biomass_{ref} \frac{CPUE_i}{CPUE_{ref}} \quad (6)$$

Where $Biomass_{ref}$ is the absolute biomass of the reference species, $CPUE_i$ is the IBTS catch per unit effort of the non-assessed species of interest and $CPUE_{ref}$ is the IBTS catch per unit effort of the reference species.

Sparholt (1990) applied this approach to the fish species of the North Sea where there are more assessed species than on the West Coast of Scotland. Notably a plaice assessment is absent from the West Coast but it is assessed in the North Sea. Instead of setting the flatfish reference species to be some other assessed species on the West Coast we assert that the catchability of flatfish on the West Coast is more like that of plaice in the North Sea than it is to any other assessed species on the West Coast. Whilst making this assertion we note that the IBTS survey on the West Coast and in the North Sea use the same net (albeit with a different ground gear to cope with different terrain) and follow the same protocol. Due to the difference in surface area of the whole of the North Sea and the shelf of the West Coast we deal with biomass densities (tonnes km^{-2}) instead of absolute biomasses, ie.;

$$Biomass\ density_i = Biomass\ density_{ref} \frac{CPUE_i}{CPUE_{ref}} \quad (7)$$

While the gear and protocol of the North Sea and West Coast surveys are the same, the substrates over which the trawls are conducted are different and this will affect the catchability and thus our biomass estimates.

On the West Coast of Scotland the Q1.Sco.Vla.IBTS started in 1985. The fourth quarter survey (Q4 Sco.Vla.IBTS) did not start until 1990. We wish to establish absolute biomasses for 1985 so we use the first quarter survey. In order to restrict the calculation to the shelf we exclude all hauls at depths greater than 200 m. Patchiness in the spatial distribution of fish leads to CPUE indices being quite noisy. If the length distribution of the population of the reference species has remained constant the ratio $Biomass_{ref}/CPUE_{ref}$ should remain constant. By averaging this ratio over all years of the survey we hope to remove the noise from the CPUE. Given that we are trying to find the biomass for 1985 we only use $CPUE_i$ from 1985.

There are problems associated both with the original Sparholt (1990) method and with the use of a reference species from a different location. The original method takes no account of length structure of the fish assemblage. This is tackled by Fraser *et al.* (2007) and it would be interesting to see how an implementation of their method affects our biomass estimates and the resulting model balance. For migratory species, such as mackerel, CPUE will depend on the timing of the populations' presence in the area of interest with respect to the survey. If you only have CPUE from one particular time of year you will then bias your estimate of biomass. In the case of mackerel and some other migratory species we use other sources and methods to obtain an estimate of biomass (see sections on specific functional groups).

Production

Allen (1971) shows that, for a population of individuals growing according to the von Bertalanffy (1938) growth equation and experiencing mortality such that the population decreases exponentially, the ratio of production to biomass per unit time is equal to the total instantaneous mortality, Z . The instantaneous total mortality is the sum of the instantaneous fishing mortality F , the mortality due to predation $M2$ and all other mortality $M0$. M , which we refer to as natural mortality, is the sum of $M0$ and $M2$.

$$\frac{P}{B} = Z; \quad Z = F + M; \quad M = M0 + M2 \quad (8)$$

Pauly (1980) derives an empirical relationship between natural mortality (M), temperature (T) and the von Bertalanffy growth parameters, K and L^∞ as:

$$M = L_{\infty}^{0.279} K^{0.654} T^{0.463} \quad (9)$$

Assuming equilibrium, we define the harvest ratio as the total annual catch or yield, Y , divided by the fixed biomass, B . In order to estimate production we require a value for the fishing mortality, F . The harvest ratio is related to F by

$$\frac{Y}{B} = \frac{F}{F + M} (1 - e^{-(F+M)}) \quad (10)$$

Given a small total mortality this approximates to F .

For multi-stanza groups, Ecopath requires the biomass for a leading stanza, not for the group as a whole or for each of the stanzas. The calculation of biomass for the other stanzas is then carried out on the basis of the values of Z provided for each of the stanzas and the von Bertalanffy growth curvature parameter K (Christensen *et al.*, 2008). Parameterising the model in this manner is required in order that incompatible values of Z and B are avoided.

In order to calculate Z for each stanza we cannot use the method outlined above. Although catch data resolved by age-class allows F to be calculated for each stanza, the relation for the natural mortality for each fish functional group (Equation (9)) calculates the average natural mortality for the whole population and can not be used to calculate mortality by age class.

Using the values of total instantaneous mortality at age from the stock assessments and numbers at age for year classes 2 and above we can calculate a value of total instantaneous mortality for the adult stanza. The ICES stock assessments are for fish that have been recruited to the fishery. This is in the year after they are spawned. The breakdown of population size and values for Z are not calculated for year class 0 fish. In order to calculate a value of Z for the juvenile stanza we could assume that the value of Z in year class 0 is equal to that in year class 1. This is likely to be an underestimate and it would result in a lower value for the biomass of the juvenile stanza than expected.

An alternative solution is to estimate a value for Z for year class 0 from another source. In the Multi-Species Virtual Population Analysis (MSVPA) for the North Sea values of Z for year class 0 fish are provided (WKMSNS, 2002). We chose to average the year class 0 mortalities over the time-series of MSVPA output and calculate a juvenile stanza total mortality on the basis of these values.

Consumption

Consumption values are calculated in the life history routine in Fishbase (Froese and Pauly, 2000). This is an implementation of an empirical relation derived by Palomares and Pauly (1998).

Diet

There is no one source for diet information for fish species on the West Coast. However, much of the information available was compiled for the Hagan and Pitcher (2005) EwE model. The diets from this model were used where no further information was found.

For groups consisting of a number of species and where diet data was available for more than one of the species, composite diets were formed by creating weighted averages of the species diet compositions.

Fisheries Catch Data

The fishing yield or catch is made up of two components: landings and discards. There are a number of sources of landings data. STATLANT is an international database of landings data from the Northeast Atlantic for finfish, invertebrates and seaweeds. The total landings by weight are recorded annually for each species and are categorised by country into which the catch is landed and ICES area in which the catch was taken. There is no further information associated with each record. For many of the species these data are sufficient to parameterise the landings component of the catch.

For assessed species, ICES Working Groups use much of the data which goes into STATLANT. However, comparisons of assessment data with STATLANT data often reveal discrepancies. Further work, for instance adjustments for mis-reporting, are included in the landings data in the assessment. This data along with discard estimates form the “reported catch” and this is used to parameterise the stock assessment. Part of the output of the assessment is a prediction of what the total catch actually was. This assessment “predicted catch” is considered a more reliable estimate of catch and it is what we use as data in this work.

STATLANT contains data resolved to ICES area. The shelf edge cuts through ICES area VIa and we are only interested in landings from the shelf. For species which are restricted to the shelf, or for which the majority of the stock is on the shelf, we assume that the landings are only being taken from the shelf. Likewise for species which are known to be deep water species we can assume that the landings were taken from off the shelf. However for some species a portion the landings will have come from the shelf and some from off the shelf. There is no data within STATLANT to provide this information, only the country in which the catch was landed.

For the Scottish component of landings, Marine Scotland holds data which resolves landings to the level of statistical rectangle. This data results from raised values from their market sampling programme. By classifying each rectangle to be shelf, deep or part shelf (with a proportion associated), we are able to allocate all Scottish landings accordingly. For many species the Scottish component of the landings forms a large part of the total landings of the West Coast and for these species, where further data is unavailable, we assume that the distribution of landings of all countries follows that of Scotland. For mackerel and horse mackerel international landings data resolved to statistical rectangle sourced from the Stock Assessment Coordinator was used instead of Scottish data.

Discards can make up a large component of the total catch and hence should not be ignored when considering removals from the fishery. International data regarding discarding is not as readily available as landings data. For species other than assessed species we use the Scottish Multispecies Discards database which holds data on discarding of all species by the Scottish fleet.

The Fishing Fleets

In order to investigate whether sustained fishing pressure from particular gear types and/or changes in the behaviour of the fishing fleets may be driving the observed trends in stocks' biomass it is possible to include gear type in the model and to attribute all landings and discard data to a gear type.

The only fleet specific question that we wish to address in this work is with regard to the Nephrops trawl. As a result we chose not to explicitly resolve fleets within the model but instead to resolve the input data into either Nephrops fleet catch data or catch from other fleets. We use this information in the scenario testing section later in this report (Section 8.6).

The main data source we have resolving gear type for landings is held by Marine Scotland and results from their market sampling programme

8.2.3 Time-Series of Fitting Data

The sub-sections of Section 8.3 on each of the model groups highlight the time-series data that are available to drive the model and that which can be used to compare the model output against and fit it to. In some cases, in particular the species for which regular stock assessments take place, there are a number of possible drivers and sets of comparison data.

Catch and biomass data, the latter of which can be relative or absolute abundance, provide the comparison data or fitting targets. For assessed species there is a choice between using the CPUE index from the Scottish VIa IBTS or the stock assessment output to provide a time-series of biomass. For the assessed demersal stocks the trends in CPUE compare well with the assessment output but tend to include more annual fluctuation. Both time-series could be included but the CPUE index does not provide any extra information so is discarded. For the pelagic stocks, the Sco.VIa.IBTS does not sample the stocks well and the CPUE index does not map well onto the assessment output for the assessed species. For these species the assessment outputs with described adjustments are used. For some species (for instance anglerfish) there is a stock assessment which covers part of the period of study only. For these species we include both stock assessment output and the CPUE index. For non assessed species the CPUE index is included except for the Other Pelagic group for which no reliable data exists.

For most assessed species we use the time-series of catch estimated by the stock assessment as opposed to the reported catch. For the migratory pelagic species the stock assessment output is reported for the whole stock, not by area, so in these cases the reported catch is used. For all non-assessed species reported landings from STATLANT are used summed with discard estimates from the Scottish multi-species discard database.

8.2.4 Time-Series of Driving Data

In order to drive the model it is necessary to supply a time-series of “mortality” for each of the functional groups and this can be either total mortality or fishing mortality. These drivers are referred to in the EwE literature as mortality but they should not be confused with the instantaneous mortality which is the output of a stock assessment model but instead a rate of removal. For fishing mortality this is the catch divided by the biomass (C/B). Using instantaneous mortalities from a stock assessment will, in general, not allow both the fitting time series of catch and biomass to be reproduced by the model simultaneously. The driving data would be inconsistent with the fitting data.

For the assessed species we still present time series of F and Z (the instantaneous mortalities) for comparison to the time series of C/B but they are not used in the modelling.

For the non-assessed species, two ways of calculating catch/biomass were considered. The catch time-series was always taken to follow the international landings data (STATLANT) time-series summed with discard estimates where they existed. The biomass was either fixed at the Ecopath value across all years or it was fixed at the Ecopath value for 1985 after which it was taken to follow the CPUE (Q1 Sco.Vla.IBTS) time-series. For groups whose CPUE time series could be considered to be constant, but showing some random year-to-year fluctuation, the biomass is fixed at the Ecopath biomass value for the calculation of the time series of F . An example of such a group is the “Sharks” group. For groups which show a clear trend over the period of study, for instance the “Other Demersal” group, the biomass is linked to the trend in CPUE.

The justification for this decision is that the Ecosim model is unlikely to be able to follow the annual fluctuations that are seen in some of the CPUE time-series and the Ecosim output might be said to fit the data if the output just remains constant. By applying an F that is the result of assuming a constant biomass in the calculation, the Ecosim catch output is as required in the fitting data.

For the species whose CPUE indices show some trend this approach does not hold. If we are able to get the biomass to fit the data (i.e. follow the trend) a fishing mortality calculated on the basis of a constant biomass will not provide the correct catch output. The fishing mortality needs to be calculated on the basis of a biomass following the desired trend. If such a fishing mortality is supplied, the biomass will need to follow the trend before the Ecosim catch output will match the data, i.e. either both biomass and catch output time series will agree or both will disagree with the fitting data.

Fleet effort data can be used to drive Ecosim instead of fishing mortality and this perhaps provides the best route to exploration of how changes in fleet structure have impacted on an ecosystem. This approach was considered for this piece of work but was not pursued due to the difficulty in compiling complete time-series of effort data for the area of interest. This data has been compiled by STECF meeting since 2003. An alternative method of exploring

changes in the fleet structure, in order to study the impact of the Nephrops Trawl on the West Coast, is outlined in the scenario testing section.

8.2.5 Fitting Methods

For the Ecosim model to be useful it is important that it is capable of reproducing some of the trends that are apparent in the data or that reasons for any mismatch have been established. There are a number of ways that the fit of the output data of Ecosim to the sets of fitting data can be improved and these are discussed in this section.

Vulnerability Search

The way a particular predator/prey interaction responds to changes in predator and prey abundances is governed by a parameter referred to in the EwE literature (Christensen *et al.*, 2008) as “vulnerability”. A low vulnerability for a particular interaction dictates that changes in the predator abundance do not affect the rate at which that species consumes the prey species. The interaction is only governed by changes in prey abundance. This type of interaction is referred to as “bottom up”. A high vulnerability results in changes to the consumption rate by the predator being proportional to changes in biomass for that predator. This type of interaction is described as “top down”. The default vulnerability in Ecosim is 2 and this leads to consumption rate which does increase with predator biomass but the increase is capped.

Values for these parameters are impossible to derive from observation or experiment and in Ecosim models are generally treated as fitting parameters. They allow the modeller to propagate the effects of changes in predator or prey abundance through the ecosystem with greater or lesser impact. Adjustments can be made to the parameters either manually or using an automatic optimisation procedure contained within the EwE software.

There are as many vulnerability parameters as there are interactions within the ecosystem. Using all of the parameters as fitting parameters would result in a severely under-constrained problem. As a guide the EwE documentation recommends that no more parameters are adjusted than there are time-series to fit against. As, in essence, these parameters control how a predator behaves in a particular interaction it would not be unreasonable to expect that a predator would respond in each of its predator/prey interactions in a similar fashion as a result in changes to its or its prey's abundance. When using the automatic optimisation procedure it is therefore possible to treat all the vulnerabilities for a particular predator as one fitting parameter.

In order to establish which vulnerabilities the model is most sensitive to changes in, part of the EwE software package can make minor adjustments to each vulnerability parameter and see which produces the biggest changes in model output. This facility enables the modeller to focus attention on the most important interactions, and therefore the most sensitive vulnerability parameters.

Anomaly Search

One way in which environmental effects can be introduced into the model is for the phytoplankton abundance to be forced to follow a particular trend. If data exists that describe trends in primary production in a particular area then these data could be used to drive the model. If there is evidence that links some environmental phenomenon with primary production then a time series describing that phenomenon could be used. An alternative is to estimate a forcing function to drive primary production which improves the fit to the model by minimising the sum-of-squares. This time-series can be compared to others describing various environmental phenomenon and postulate that if it shows a similar trend to a particular phenomenon that this phenomenon is leading to changes that are being seen in the ecosystem.

As with the vulnerability search, an anomaly search can be conducted by hand, looking to see what the phytoplankton abundance needs to do in order to produce the time-series witnessed further up the trophic levels, or the EwE software can calculate the time-series which minimises the difference between Ecosim output and the fitting data.

An automatic vulnerability search can be conducted at the same time as an anomaly search. Again, some thought needs to be given to the number of parameters used to parameterise the vulnerability matrix and the anomaly time-series in order to avoid creating an over parameterised problem.

8.3 Functional Groups and Balancing Ecopath

NOTE: This section provides details on Ecopath parameters, balancing ecopath and Ecosim time series for each of the 33 functional groups of the model which may not be of interest to the general reader. Hence the section is shaded in order to indicate its technical nature. The general reader is advised to skip to Section 1.4

8.3.1 Cod

Atlantic cod, *Gadus morhua*, has a wide distribution in the north Atlantic stretching from the Celtic Sea to the Barents Sea and from the east coast of the USA and Canada to the western coast of Europe (Froese and Pauly, 2000). On the west coast of Scotland juveniles are frequently found in coastline shallow waters and sea lochs and generally spend the first winter close inshore (Hall *et al.*, 1990; Magill and Sayer 2004). As they reach maturity they move off into deeper water to join adult stocks, although Hawkins *et al.*, (1985) reported that cod up to 4 years old may remain localised in shallow Scottish sea lochs.

Ecopath Parameters

Due to the model focus we chose to split cod into juvenile and adult stanzas. We defined maturity as occurring at 24 months based on the maturity ogives used for the VIa cod stock assessment. On the basis of the ICES stock assessment, the Z value of the adult stanza is

calculated to be 1.17 year^{-1} . For the juvenile stanza we assume year class 0 fish experience mortality at a rate equal to an average of year class 0 mortalities from the North Sea MSVPA. We assume year class 1 fish to experience mortality as calculated in the ICES stock assessment. The juvenile stanza Z value is then calculated to be 2.21 year^{-1} . The biomass of the leading stanza, the adult stanza in this case, is calculated from numbers-at-age and weight-at-age in the ICES stock assessment to be 0.254 t km^{-2} . Using the stated Z values and a von Bertalanffy growth curvature parameter 0.18 year^{-1} from Fishbase (Froese and Pauly, 2000), EwE predicts a biomass for the juvenile stanza of 0.229 t km^{-2} .

Q/B ratio for mature cod was calculated from Fishbase (Froese and Pauly, 2000) at 2.58 year^{-1} . A value of 2.0 year^{-1} was used by Lees and Mackinson (2007) for Irish Sea cod and 3.8 year^{-1} for west coast of Scotland cod by Hagan and Pitcher (2005).

The diet matrix for mature cod was based on DuBuit (1989). This study made a quantitative assessment of the diet of cod on the west coast of Scotland and concluded that prawns and shrimps account for approximately 41% of the diet, with 'other pelagic' fish accounting for 11% and herring 9.2%. The diet for immature cod was based on the diet given in Lees and Mackinson (2007), which largely consisted of 'large zooplankton' (40%), 'small zooplankton' (25%) and 'infauna' (15%).

Balancing Ecopath

In the balancing process the Q/B ratio for mature cod was increased from 2.58 year^{-1} to 3.5 year^{-1} which is in line with the North Sea model (Mackinson and Daskalov 2007). The juvenile cod component of the diet of mature cod was reduced from 7% to 1% with the excess moved to herring. The juvenile haddock component of mature cod (1.5%) was moved to mature haddock.

Ecosim Time-Series

The biomass time-series output of Ecosim could be fitted to the stock assessment output, which provides a time-series of absolute biomass, or the CPUE from the Q1 Sco.Vla.IBTS, which gives a time-series of relative abundance (Figure 8.4). For the mature stanza, the stock assessment and CPUE index show downward trends. Examination of the Sco.Vla.IBTS data indicates that the large value for the CPUE in 2001 can be attributed to one out of many hauls which form part of each survey. It is an indication of the patchy spatial distribution of the population as opposed to an indication of an overall surge in biomass.

For the juvenile stanza, the trend in biomass indicated by the assessment is erratic showing annual variation in recruitment. This variation is also apparent in the CPUE but to a lesser degree. Both assessment and CPUE data could be included as fitting targets for both stanzas but the addition of CPUE time-series do not provide further elucidation of the pattern so only the assessment data is included.

Catch data is also taken from the stock assessment. Two time series of catch are provided in the stock assessment: the reported catch and the model estimated catch. The reported catch is widely accepted as being unreliable because it does not take into account black landings and misreporting (WGCSE 2009). The catch estimated by the stock assessment models is presented here (Figure 8.4) and it is the time-series of catch data to which the Ecosim is fitted.

In order to apply fishing mortality to the model, there are again options for sets of data with which to drive the model. For the adult stanza, Z and F from the stock assessment are shown (Figure 8.5) as well as stock assessment estimated catch/biomass. F shows no trend and remains between 1 and 0.7 throughout the time-series. The difference between Z and F is a constant, 0.2. Catch/biomass is somewhat lower than F , but also remains roughly constant between 0.4 and 0.6. For the juvenile stanza the stock assessment time-series of F and Z are shown as dotted lines. These represent the mortalities on the year-1 age class only; they exclude year-0 fish. The pink line represents the time series of Z acting on the whole group if a constant Z for the year-0 individuals is assumed; this constant value is taken from the North Sea MSVPA (WKMSNS 2002). The yellow line represents the result of a calculation of the fishing mortality acting on the whole stanza by scaling the total mortality acting on the whole stanza by the ratio of fishery removals to total removals.

8.3.2 Haddock

Haddock, *Melanogrammus aeglefinus*, is a highly commercial gadoid species. It is found up to a depth of 300m close to the bottom (Wheeler 1978). Juvenile haddock can be found in shallower water.

Ecopath parameters

Haddock is divided into juvenile and adult stanzas with the split between the two occurring at 24 months. The calculation of mortalities and biomasses for the stanzas follows the same form as for cod. The juvenile mortality is 1.67 year^{-1} and the adult mortality is 0.72 year^{-1} , calculated on the basis of values from the ICES stock assessment for VIa (WGCSE, 2009) and values for year 0 mortality from MSVPA for the North Sea (WKMSNS, 2002). The stock assessment also provides the adult biomass estimate of 0.836 t km^{-2} . Given these values and a von Bertalanffy growth curvature parameter of 0.12 from Fishbase (Froese and Pauly, 2000) EwE calculates a juvenile biomass of 0.163 t km^{-2} .

Q/B ratio for mature haddock was estimated from Fishbase (Froese and Pauly, 2000) at 4.96 year^{-1} . A lower value of 2.58 year^{-1} was used in the Irish Sea model (Lees and Mackinson, 2007) while Hagan and Pitcher (2005) used a value of 4.0 year^{-1} .

The diet for mature haddock was taken from Hagan and Pitcher (2005) which had been adapted from Richie (1937; 1938) and Jones (1954). 'Crustaceans' accounted for

approximately 36% of the diet and 'infauna' 11%. Approximately 50% of the diet of immature haddocks consisted of 'large zooplankton' and 15% of 'small zooplankton'.

Balancing Ecopath

No adjustments were made to the haddock functional group during balancing. However, there was insufficient production in the juvenile stanza to support the predation load and accordingly some of the juvenile haddock diet components of other fish were reduced. These included cod, pollock, monkfish, mackerel and horse mackerel.

Ecosim Time-Series

As with the cod functional group, there is a choice of CPUE or assessment output to fit to for the biomass time-series (Figure 8.6). The CPUE trends show much of the same pattern as the stock assessment and only assessment data are included as fitting targets.

The catch estimated by the stock assessment model, as opposed to reported catch, is used as the fitting time-series for the same reason as with the cod groups (Figure 8.6)

For the adult stanza, three choices of drivers are presented: the time-series of F or Z from the stock assessment (WGCSE, 2009) or the stock assessment estimated catch/biomass. F and Z from the stock assessment are separated by a constant 0.2 y^{-1} . The catch/biomass series shows the same trend but takes values between 0.1 y^{-1} and 0.3 y^{-1} smaller. For the juvenile stanza the assessment F and Z time-series for year-1 fish are shown (dotted lines) and the solid lines show the results of calculations for F and Z for the whole stanza following the same recipe as described for juvenile cod.

8.3.3 Whiting

Historically whiting, *Merlangius merlangus*, has been a key commercial species in the west coast of Scotland (Gordon and De Silva, 1980). The species is associated with shallow seas around the UK, particularly the juveniles that settle into shallow water and sea lochs close to the shore (Wheeler, 1978).

Ecopath Parameters

Whiting is also divided into juvenile and adult stanzas with the split occurring at 24 months. We calculate the juvenile mortality to be 1.73 year^{-1} and the adult mortality to be 1.30 year^{-1} on the basis of numbers at age values from the stock assessment and values for year 0 mortality from MSVPA for the North Sea (WKMSNS, 2002). The adult biomass, calculated from (WGNSSDS, 2004; WGNSSDS, 2007), is 0.265 t km^{-2} . Given these values and a von Bertalanffy growth curvature parameter of 0.34 from Fishbase (Froese and Pauly, 2000) EwE calculates the juvenile biomass to be 0.287 t km^{-2} .

Consumption ratio (Q/B) was calculated using Fishbase (Froese and Pauly, 2000) and estimated at 3.3 year^{-1} which is higher than the estimate (2.4 year^{-1}) used for the west coast of Scotland model (Hagan and Pitcher, 2005). Irish Sea whiting were estimated to have a consumption rate of 2.97 year^{-1} (Lees and Mackinson, 2007).

The diet composition for mature whiting was taken from DuBuit (1991) study of the food of whiting in Scottish waters. Herring accounted for 34% of the diet, while mackerel accounted for 12% and 'other pelagics' 12%. Quantitative data on the diet of immature whiting was not available. It was assumed that the diet would be similar to that of juvenile cod and haddock. Therefore a composite diet was constructed based on that for juvenile cod and haddock.

Balancing Ecopath

In the balancing of the model the diet composition of some species were changed in order that the consumption of juvenile whiting was lowered slightly. The Q/B ratio was increased to 4.5 year^{-1} in order to decrease the predicted P/Q ratio.

Ecosim Time-Series

The whiting stock assessment was discontinued after 2007. From 2004 to 2007 the stock assessment was based only on survey data (SURBA) and provided an index of abundance instead of an absolute abundance. In order to derive a time series of absolute abundance, the survey based index was scaled to the absolute abundance estimated by the 2004 stock assessment (assessment model run [c] without catch data between 1995 and 2003 (WGNSSDS, 2004)). Given that the 2007 assessment is based on the survey data it is unsurprising that there is a match in trend between the CPUE index and that of the assessment (Figure 8.8). As with cod and haddock, the assessment trend is used as the set of fitting data for biomass for both juvenile and adult whiting.

The assessment estimated catch is calculated based on SURBA indices and scaled by the same factor as the biomass time series.

The Z and F time-series from the survey based assessment are presented for both stanzas. For the juvenile stanza Z and F are calculated to include year-0 fish not accounted for by the assessment using a constant year-0 Z from MSVPA for the North Sea (WKMSNS, 2002). This follows the same recipe as for cod and haddock. Catch/biomass is presented for the adult stanza. As with cod and haddock, this follows the same trend as F but takes lower values.

8.3.4 Pollack (and Saithe)

This group is composed of the sympatric gadoid species pollock, *Pollachius pollachius*, and saithe, *Pollachius virens*. Pollock are often associated with harder rocky ground and can be found from surface waters to a depth of 200 m (Wheeler, 1978). Saithe are associated with surface and mid water where they can aggregate into dense shoals (Wheeler, 1978). Both

species are abundant in the waters around the UK (Gordon and De Silva, 1980). Biomass of the group is dominated by saithe (97%), as indicated by CPUE data from the Q1 Sco.VIa.IBTS off the west coast of Scotland. Consumption and diet matrix parameters were weighted accordingly but the biomass value was taken solely from the saithe stock assessment, the details of which are given below.

Ecopath Parameters

Raising the total CPUE of the group using the method of Sparholt (1990) with the reference species of cod and haddock gives a biomass of 0.077 t km^{-2} .

A saithe stock assessment is carried out annually for the combined ICES areas VI, IV and IIIa. Until 1999 separate assessments were carried out for ICES areas VI and for IV and IIIa but was discontinued as there were no biological grounds for treating the stocks separately. For the years in which we have separate assessments as well as the combined assessment we split the combined assessment in the ratio of the separate stock assessments. After 1999, we assume that the ratio of the split stays constant at the 1999 level. This gives us a complete time series of biomass for the whole of ICES area VI. In order for this to be useful we need to scale this assessment to the shelf of area VIa and in the absence of further data we set the scaling factor at 0.5. A biomass value of 0.44 t km^{-2} was calculated and this was used ahead of the value raised from CPUE data.

Production rates (P/B) for this group was estimated using F and M (natural mortality) for both species then weighting the calculation appropriately. This gave a value of 0.45 year^{-1} for P/B for these two species. This is comparable with estimates used in the Irish Sea (Lees and Mackinson, 2007) of 0.49 year^{-1} for 'other large gadoids'.

Consumption ratio (Q/B) was calculated using Froese and Pauly (2000) and estimated at 2.3 year^{-1} for pollock and 4.76 year^{-1} for saithe. This gave a weighted average of 4.69 year^{-1} . Lees and Mackinson (2007) used a composite 'other large gadoids' group that was largely pollock and used a rate of 1.95 year^{-1} .

Diet for the groups was based on the diet for adult saithe from the North Sea as given in Mackinson (2001). Large zooplankton, such as Euphausiids were 22% of the diet with 20% Norway pout and 15% herring.

Balancing Ecopath

Adjustments to the pollock/saithe diet composition were made to reduce its predation load on juvenile cod and haddock. The cod component was reduced from 5% to 3% and the haddock from 3.7% to 0.1%. The excess was moved to herring.

Initially the P/Q ratio was set to 0.3 year^{-1} in order to estimate P/B which was unlikely to be accurate. This was later adjusted to 0.2 year^{-1} as the P/B was too high and the EE too low.

Ecosim time-series

The biomass time-series results from a combination of stock assessments as described above in the Ecopath parameters section. The catch time-series is taken from STATLANT data for the whole of VIa that has been scaled to the shelf using Scottish landings data resolved to ICES statistical rectangle level. A time series of CPUE from the Q1 Sco.VIa.IBTS is included in Figure 8.10 below but the stock assessment is used in preference for fitting data.

The driving data for the group includes time series of F and Z from the stock assessment. These are taken for the stock assessment for the West Coast combined with the North Sea (WGNSSK, 2009) and are assumed to apply across the area. A time series of catch/biomass for the West Coast is also included. This shows a similar trend to F from the stock assessment.

8.3.5 Monkfish

Two species of monkfish are caught off the west coast of Scotland, the monkfish, *Lophius piscatorius* and the black bellied angler fish, *Lophius budegassa*. *L. piscatorius* is the larger (can grow up to 2 m) and more common of the two and can be found from shallow water down on to the continental slope (Wheeler, 1978). *L. budegassa* is less common and generally found between depths of 100 and 300 m. CPUE from the Q1 Sco.VIa.IBTS indicated that the biomass of the group is dominated by *L. piscatorius*, which accounts for 93% of the total biomass for the group. Ecopath parameters were weighted accordingly.

Ecopath Parameters

Sparholt (1990) and Mackinson (2007) used cod, haddock, whiting and saithe as reference species to raise CPUE to absolute biomass. For the West of Scotland we use cod and haddock and obtain a value of 0.063 t km^{-2} .

Due to the increased importance of monkfish to the West of Scotland fishery, a monkfish survey has been designed and carried out since 2005 with the aim of establishing monkfish abundance (Fernandes, 2009). This gives a time series of minimum estimates of absolute biomass in ICES area VIa for 2005-2009. A crude extrapolation of these values based on the time series of CPUE from the Q1 Sco.VIa.IBTS yields a biomass of 0.14 t km^{-2} .

Production rate (P/B) for this species was estimated at 0.73 year^{-1} from the sum of estimates of natural mortality (Pauly 1980) and fishing mortality from catch and biomass estimates. Howell *et al.* (2009) modelled the deep sea fisheries off the west coast of Scotland and used a P/B of 0.11 year^{-1} , while Lees and Mackinson (2007) estimated a value of 1.25 year^{-1} .

Q/B ratios for the two species was estimated from Fishbase (Froese and Pauly, 2000) at 1.7 year^{-1} for monkfish and 1.9 year^{-1} for the black bellied anglerfish. This gave a weighted

average of 1.71 year^{-1} for the group. This is slightly higher than the rate used by Howell *et al.* (2009) of 1.3 year^{-1} but lower than the 1.99 year^{-1} used by Lees and Mackinson (2007).

Laurenson and Priede (2005) studied the diet of monkfish in the waters of the Shetland Isles (UK). The diet is largely piscivorous, being dominated by immature cod (17%), pollock (14%), Norway pout (12%) and sandeel (10%).

Balancing Ecopath

The cannibalism of monkfish was reduced from 6.4% to 1% with the excess moved to herring. The juvenile haddock diet component was moved to mature haddock.

Instead of using the biomass estimate suggested above, the model was used to calculate a biomass on the basis of an *EE* of 0.95. This resulted in a value of 0.217 t km^{-2} which was felt to be reasonable given the uncertainties in the estimation.

P/Q was initially fixed at 0.3 year^{-1} as the *P/B* ratio was likely to be inaccurate. This was later reduced to 0.28 year^{-1} in order to stop the group going extinct in the Ecosim simulation.

Ecosim Time-Series

Two fitting time-series for biomass are given: the CPUE index from the Q1 Sco.VIa.IBTS, which covers the entire period of study, and the assessed biomass from the monkfish survey which started in 2005 (Fernandes, 2009). It is noticeable that for the period of overlap between the two time series they do not show the same trend (Figure 8.11). More weight should be placed on the stock assessment. It is doubtful whether effort should be made to fit the individual peaks of the CPUE series, if indeed it would be possible.

The catch time-series is taken from STATLANT values that have been scaled to the shelf using Scottish landings data resolved to the ICES statistical rectangle level, together with Scottish discard estimates from the Scottish multi-species discard database.

In order to drive this functional group a fishing mortality derived from catch/biomass is applied. Due to the fluctuating nature of the CPUE time series, a constant biomass (at the balanced Ecopath level) is used for this calculation. This decision is discussed further in section 0.

8.3.6 Flatfish

The flatfish group is made up of a large number of species. The amount that each species contributes to the group is calculated from the proportions of the CPUE from the Q1 Sco.VIa.IBTS. These proportions have remained reasonably constant across the period of study (Figure 8.12). Averaging the ratios over all years indicate that lemon sole, *Microstimus kitt*, plaice, *Pleuronectes platessa*, dab, *Limanda limanda* and megrim, *Lepidorhombus*

whiffiagonis account for over 80% of the total biomass. The averaged value for each species is used for the weighting of consumption and natural mortality parameters.

Ecopath Parameters

With a reference species of plaice from the North Sea the CPUE is raised to give a biomass of 1.36 t km^{-2} using the method of Sparholt.

The consumption rate (Q/B) for each species was calculated from Fishbase (Froese and Pauly, 2000). By using a weighted average the consumption rate for the group was estimated at 3.8 year^{-1} . The diet composition for the group was based on the diet given by Lees and Mackison (2007) in an Ecopath model of the Irish Sea for 'medium flatfish'. 'Other small fish' accounted for 20% of the diet, while epifauna accounted for 15% and infauna at 11%. Crustaceans and gurnards also appeared in the diet at approximately 10% each.

Balancing Ecopath

Adjustments were made to the diet of flatfish in order to reduce the predation load on gurnards, monkfish, and other benthopelagic fish. The gurnard component is reduced from 10.4% to 0.4% with the excess to Nephrops and zooplankton. The monkfish component is reduced from 10% to 1% with the excess to infauna. The other benthopelagic fish component of the diet was reduced to 1% from 20% with the excess to epifauna.

P/B was estimated from a P/Q ratio of 0.3. The EE was fixed at 0.95, once adjustments had been made to the predators of flatfish and the biomass was estimated. This resulted in a biomass of 1.51 t km^{-2} , a small increase on the Sparholt raised estimate.

Ecosim Time-Series

The only indication of trends in biomass available for the flatfish group for the area of study is the CPUE index from the Q1 Sco.Vla.IBTS. This shows an increase over the period of study (Figure 8.13) The catch time-series is taken from STATLANT data scaled to the shelf by Scottish landings data resolved to the ICES statistical rectangle level as well as discard data from the Scottish multi-species discard database.

The catch, which decreases over the period of study, is very low in comparison to the biomass calculated for Ecopath. A time-series of catch/biomass (F) is calculated to drive the group but, given the small catch, this is low in comparison to the predation load. Any changes in the flatfish biomass output of Ecosim will have to come from changes in predator or prey biomasses.

8.3.7 Rays and Skates

Within the functional group Rays and Skates are a large number of species. There is considerable annual variation in the composition of the group as derived from CPUE with the most abundant species in any year being any one of the homelyn ray, *Raja montagui*, the cuckoo ray, *Leucoraja circularis*, the thornback ray, *Raja clavata* or the blue skate, *Dipturus batis*. Despite this annual variation no trend is apparent and we average over all years to define the composition.

Ecopath parameters

With a reference species of plaice from the North Sea the CPUE is raised using the method of Sparholt to give a biomass of 0.68 t km^{-2} .

P/B estimates were calculated for each species in the ray group and a weighted average derived for the group estimate at 0.23 year^{-1} . Hagan and Pitcher (2005) previously used a value 0.4 year^{-1} for the west coast of Scotland, while a value of 1.6 year^{-1} was used in the Irish Sea (Lees and Mackinson, 2007).

Consumption ratio calculated using Fishbase (Froese and Pauly, 2000) for the six species ranged from 1.1 year^{-1} for the largest species common skate, to 3.1 year^{-1} for the Homelyn ray. The weighted average for the group was calculated at 2.2 year^{-1} . The diet composition is taken from Hagan and Pitcher (2005) and was adapted from a diet for rays from the old North Sea Model (Mackinson, 2001). Approximately 40 % of the diet consisted of crustaceans, while epifauna accounted for 14%, Nephrops 13% and herring 13%.

Balancing Ecopath

In order to reduce the predation load on crustaceans, the crustacean component of rays and skates diet was reduced to 30% with the excess moved to epifauna.

Ecosim time-series

The CPUE from the Q1 Sco.Vla.IBTS is used as an index of abundance of the rays and skates groups for fitting in Ecosim. This shows a rising trend and is displayed in Figure 8.15. The catch data is composed of landings and discards data. The discards data come from the Scottish Multi-species discard database and are only applicable to the Scottish fleet. The landings are taken from STATLANT and scaled to the shelf using Scottish landings data resolved to ICES statistical rectangle level.

In order to drive the model a time series of catch/biomass is calculated. The biomass is taken to follow the trend of the CPUE, fixed in 1985 at the balanced Ecopath level.

8.3.8 Sharks

The two sharks which dominate this group in terms of biomass are the small spotted catshark, *Scyliorhinus canicula* and the piked dogfish, *Squalus acanthias*. Between them these account for well over 90% of the CPUE (Figure 8.16). There is a clear trend from piked dogfish being the most abundant species at the start of the period of study to the small spotted catshark accounting for almost all of the CPUE in recent years. Despite this trend we group these species together on the grounds that their diets and consumption rates are not dissimilar.

Ecopath Parameters

Using reference species of cod and haddock we obtain a value of 0.79 t km^{-2} for the raised biomass using the method of Sparholt.

The production rate for the shark group was derived from a weighted average of production estimates from the 4 species in the group. This returned a value of 0.34 year^{-1} . A value of 0.16 year^{-1} was used for the North Sea (Mackinson, 2001). This value was subsequently used by Hagan and Pitcher (2005) for the west coast of Scotland. Lees and Mackinson (2007) used 0.32 year^{-1} to represent the *P/B* of a 'large shark' group in the Irish Sea.

The lesser spotted dogfish had a *Q/B* ratio of 2.9 year^{-1} calculated from Fishbase (Froese and Pauly, 2000) while spurdog had a value of 4.8 year^{-1} . The weighted group average was estimated at 3.4 year^{-1} . The shark diet was based on the diet used by Mackinson (2001) in the 1880 North Sea Ecopath model. It is largely based on the diet for lesser spotted dogfish. The diet was dominated by herring (32%). Cephalopods accounted for 15% of the diet while epifauna and crustaceans both accounted for approximately 11%. Sprats were also found in the diet (10%).

Balancing Ecopath

No adjustments were made to the shark functional group during the balancing process.

Ecosim Time-Series

The CPUE index time-series from the Q1 Sco.VIa.IBTS shows no overall trend but has some year to year variation (Figure 8.17). The catch data consists only of landings data – no discard data was available. The catch was taken from STALANT and scaled to the shelf area using Scottish landings data resolves to ICES statistical rectangle level. It shows a clear decrease over the period of study.

The *F* used to drive the group was calculated from catch/biomass, the biomass being the fixed Ecopath biomass.

8.3.9 Poor Cod

Poor cod, *Trisopterus minutus*, is a small benthopelagic gadoid found in coastal waters of the north east Atlantic (Wheeler, 1978). It can be found in schools close to the sea bed.

Ecopath parameters

There is no assessment carried out for poor cod. The reference species used by Sparholt (1990) for raising CPUE was Norway pout. As there is no assessment for Norway pout on the West Coast we use North Sea Norway pout as the reference species. This results in a biomass estimate of 0.052 t km^{-2} .

Poor cod features in neither the STATLANT international landings database nor the Scottish Multispecies Discard database. The production rate was therefore estimated from Fishbase (Froese and Pauly, 2000) purely on the basis of natural mortality and was calculated to be 0.32 year^{-1} . The consumption rate of poor cod was estimated from Fishbase (Froese and Pauly, 2000) at 3.9 year^{-1} . The diet was based on Armstrong (1982). Crustaceans accounted for approximately 57% of the diet while infauna represented 16%. 'Other benthopelagic fish' accounted for 11%, while 10% came from large zooplankton.

Balancing Ecopath

In order to reduce the predation load on the crustacean group, the crustacean component of poor cod diet was reduced to 30% with the excess to epifauna.

As there is no catch data for poor cod it was not possible to calculate an accurate estimate of P/B . P/Q was fixed at 0.3 and P/B was estimated from this value.

Once adjustments had been made to the predators of poor cod, the biomass was estimated by the model using an EE of 0.95. This resulted in a biomass of 0.067 t km^{-2} , not much more than the original estimate given above.

Ecosim time-series

The only time-series data available for this group is the CPUE from the Q1 Sco.VIa.IBTS (Figure 8.18). No catch data was found for Poor cod. Therefore there is no driver acting directly on this group. The CPUE index shows a number of years later in the time series with high abundances. The only way that this trend could be replicated in the model would be through changes in predator of prey abundances.

8.3.10 Gurnards

A number of species of gurnards are found in the waters off the West Coast of Scotland. However, only the grey gurnard, *Eutrigla gurnardus* and red gurnard, *Aspitrigla cuculus*

contribute significantly to this group with approximately 78% of the CPUE attributed to the grey gurnard. The proportions of the CPUE of these two species show little variation over the period of study (Figure 8.19). Both species are usually associated with sandy bottoms, but are regularly found on rocky and muddy ground. The red gurnard is generally less abundant but in some locations it can be very abundant (S Magill pers. comm.).

Ecopath parameters

Using reference species of cod and haddock the CPUE is raised to a biomass of 0.13 t km^{-2} using the Sparholt method.

The production rate (P/B) was estimated from values of F , from catch and biomass estimates, and M from Fishbase (Froese and Pauly, 2000). A value of 1.23 year^{-1} was calculated for gurnards, which is almost half the value used in the Irish Sea model of 2.21 year^{-1} (Lees and Mackinson, 2007).

A consumption ratio (using Froese and Pauly, 2000) was estimated from Fishbase (Froese and Pauly, 2000) at 4.1 year^{-1} for grey gurnard and 4.2 year^{-1} for red gurnard, giving a weighted average of 4.12 year^{-1} for the group. This similar to the estimate used in the Irish Sea of 4.104 year^{-1} (Lees and Mackinson, 2007) and 4.67 year^{-1} used by Hagan and Pitcher (2005). The diet matrix for gurnards was based on the gurnard diet used in the West coast of Scotland Ecopath model (Hagan and Pitcher, 2005). This diet was based on an MSVPA diet by Mackinson (2001) and largely consisted of crustaceans (37%) and large zooplankton (22%).

Balancing Ecopath

No adjustments were made directly to the gurnard group. The consumption by other species was reduced by making adjustments to the diets of the other pelagic group and the flatfish group.

Ecosim Time-Series

The CPUE index from the Q1 Sco.VIa.IBTS is used to give a trend in relative abundance for fitting Ecosim. The trend shows an increase over the period of study (Figure 8.20). Catch data, taken from STATLANT (landings), and scaled to the shelf using Scottish data resolved to ICES statistical rectangle shows a decrease over the same period. There were no discards of gurnards recorded in the Scottish multi-species discard database. F was calculated from catch/biomass, the biomass following the CPUE index fixed at the Ecopath value in 1985.

8.3.11 Other Demersals

This functional group includes a number of large demersal fishes with hake, *Merluccius merluccius*, making up about 60% of the group by biomass. John Dory, *Zeus faber*, blue mouth, *Helicolenus dactylopterus*, and ling, *Molva molva*, make up most of the rest of the biomass, with blue-mouth featuring more strongly in the years since 2000 (Figure 8.21).

Ecopath Parameters

None of the species included in this group are assessed so we use the method of Sparholt with reference species of cod and haddock to raise the Q1 Sco.VIa.IBTS CPUE to absolute abundance. We calculate a value of 0.052 t km⁻² for the biomass of this group.

Production rate (P/B) was calculated by obtaining a value for F and M (calculated from Fishbase – Froese and Pauly, 2000) for each species in the group and then calculating a weighted average using the biomass ratio from CPUE of the Q1 Sco.VIa.IBTS surveys. The value for the group was returned at 4.22 year⁻¹. This high value was driven by a high F value of 4.04 year⁻¹ which was the result of the low biomass estimate. Consumption ratios calculated from Fishbase (Froese and Pauly, 2000) ranged from 1.6 year⁻¹ for conger eel to 3.7 year⁻¹ for blue-mouth. The weighted group average was estimated at 2.4 year⁻¹.

A composite diet for the group was constructed using diet information for hake, bluemouth, John Dory and ling. The diet for hake (which accounted for 50% of the diet) was based on the diet used by Howell *et al.* (2009), which was developed from Sánchez and Olaso (2004) and Guichet (1995). Similarly the diet for ling and bluemouth were based on the diets used in Howell *et al.* (2009). The John Dory diet was adapted from the Western English Channel model (Araújo *et al.*, 2005). The resultant overall diet for the group was largely piscivorous with 56 % of the diet from the 'other benthic-pelagic fish' group. Approximately 11% of the diet was 'epifauna', 8% 'other pelagics' and 7% from blue whiting.

Balancing Ecopath

For the other demersals group it was noted that the biomass estimated by raising the Q1 Sco.VIa.IBTS CPUE was insufficient to support either the fishery or the predation load (the fishing mortality being much higher than the predation mortality). The estimate of fishing mortality could have been too high as a result of landings that have been categorised as landings from the shelf when they could have been taken from off the shelf. Nonetheless the EE was fixed at 0.95 and a biomass value to support these predation and fishing mortalities was calculated. This yields a value of 1.1 t km⁻².

The P/B value for the group was estimated by fixing P/Q at 0.2. Changes to the diet of the other demersal group included part of the diet component on mackerel (2%) being moved to the rays and skates group. The other benthic-pelagic fish component was reduced to 10%

from 56% with the excess split between herring (16.3%), blue whiting (10%), mackerel (10%) and horse mackerel (10%).

Ecosim parameters

Given the absence of a stock assessment for any of the species in this group the CPUE index from the Q1 Sco.VIa.IBTS was used as an indicator of biomass trend and this was used as the fitting data for other demersals. The series shows a marked increase towards then end of period, but much of this can be explained by Blue-mouth incursions onto the shelf in the latter years (Figure 8.22). The catch data is made up of landings and discards data. The discard data was from the Scottish multi-species discard database and the landings came from STATLANT data.

F was calculated from catch/biomass, the biomass being the CPUE index fixed at the Ecopath value in 1985.

8.3.12 Other Benthopelagic Fish

Ecopath parameters

The bulk of the Q1 Sco.VIa.IBTS CPUE of this group, and hence the group composition, is from argentinines and boarfish (Figure 8.23). We use a reference species of North Sea Norway pout to raise the CPUE of all the species in this group and obtain a biomass estimate of 0.16 t km^{-2} .

Production rate was calculated by obtaining a value for F and M (calculated from Fishbase – Froese and Pauly, 2000) for each species in the group and then calculating a weighted average using the biomass ratio from CPUE of the Scottish VIa IBTS surveys. The value for the group was 0.37 year^{-1} . Consumption rate (calculated from Fishbase – Froese and Pauly, 2000) ranged from 10.17 year^{-1} for Atlantic argentine, to 4.5 year^{-1} for four-bearded rockling. The weighted average for the group estimated consumption ratio at 5.27 year^{-1} .

Balancing Ecopath

Cannibalism made up 26.7% of the other benthopelagic fish diet and this was reduced to 5% with the excess split between infauna and epifauna. The predation load through the other demersal group was also reduced.

Despite changes to the diets of the consumers of this group, other small fish did not balance. The original biomass is likely to be inaccurate as most of the species in this group are benthopelagic species and will be poorly sampled by the Scottish VIa IBTS. We therefore fix the EE at 0.95 and estimate the biomass. The biomass required increases from the original estimate of 0.16 t km^{-2} to 0.5 t km^{-2} .

The P/B value is estimated by fixing the P/Q ratio at 0.3.

Ecosim Time-Series

The catch data time-series for this group results from summing the reported international landings (STATLANT) and discards (Scottish Multi-Species Discard database) for each of the species in the group. The CPUE index from the Q1 Sco.VIa.IBTS is used as an indication of biomass trends (Figure 8.24).

The fishing mortality applied to the group is calculated from catch/biomass with the biomass fixed at the Ecopath value.

8.3.13 Herring

Traditionally, herring, *Clupea herengus*, has been a key fishery species in the waters around Scotland, particularly during the 19th and early 20th century (Smylie, 2004). However the fishery has seen a dramatic decline, largely as a result of overfishing. Herring are distributed throughout the north east Atlantic and from the English Channel to as far north as the Arctic Ocean. Most herring populations spawn in autumn. Larvae from the spawning grounds off the west and north coast of Scotland are known to feed into the nursery grounds in the North Sea (MSS, 2009). Some of the larvae remain on the west coast where the juveniles can be found in inshore waters and sea lochs. Eventually the juveniles will join the adult stocks which are found in the deeper waters of the continental shelf.

Ecopath Parameters

A stock assessment for herring is carried out annually by ICES for the West Coast of Scotland (HAWG, 2009). This stock assessment comes in two parts. The assessment for VIa North includes all of ICES area VIa as far South as 56° latitude and also includes the Clyde. VIa South which makes up the rest of VIa is assessed in conjunction with areas VIIb,c which lie to the West of Ireland. In order to arrive at an estimate of herring biomass for area VIa we include all of the biomass from the VIa North assessment. The VIa South and VIIb,c assessment needs to be split before inclusion. A graphical representation of the results of the July 2008 acoustic survey presented in (HAWG, 2009) indicate the majority of this stock being found in VIa South. The distribution of landings from VIa South and VIIb,c, also presented in (HAWG, 2009), indicate that most of the landings were taken from VIa South. We choose to include 80% of the VIa South and VIIb,c biomass in the estimate for total VIa biomass and this results in a biomass estimate of 5.95 t km⁻².

The P/B ratio (calculated from Fishbase – Froese and Pauly, 2000) was estimated at 0.59 year⁻¹. The Q/B ratio for herring (calculated from Fishbase – Froese and Pauly, 2000) was estimated at 10.1 year⁻¹. This was greater than the value of 7.0 year⁻¹ used by Hagan and Pitcher (2005).

The diet for herring was adapted from Mackinson (2001). Herring are planktivores and this is reflected in the diet matrix for the species where the diet is dominated by large zooplankton (84%). Herring are also known to feed on crustaceans such as small shrimps and this accounted for 15% of the diet.

Balancing Ecopath

The only change to the diet of herring was to reduce the crustacean component to 1% and to move the excess to large zooplankton.

Initially the P/Q ratio was fixed at 0.3 and P/B was estimated from this. Instead, it was decided to fix P/B at 1.5 year^{-1} in line with Mackinson (2007).

Much of the production by herring was unaccounted for in the model leading to an unrealistically low EE . The herring components of a number of species were increased to take account of this. These included mackerel, blue whiting, other pelagic fish, cephalopods, cod, Pollock and large demersals.

Ecosim time-series

As herring is a pelagic species an index of abundance from the CPUE from the Q1 Sco.VIa.IBTS is unreliable. It is presented here for completeness (Figure 8.25). As a set of fitting data for the biomass of herring, stock assessment output was used. The VIaN assessment was combined with the output of the VIaS assessment as described in the Ecopath parameters section. The biomass shows a decline over the period of study. The catch data used is the output of the stock assessment model. This too shows a decline but as can be seen in the plot it forms a small part of the total estimated biomass.

8.3.14 Norway Pout

Norway pout, *Trisopterus esmarkii*, is a small benthopelagic gadoid, largely found over muddy seabed between 100 and 200 m, although the species is caught in numbers in shallower water.

Ecopath parameters

The Norway pout stock is not assessed on the West Coast of Scotland, unlike in the North Sea. Sparholt (1990) used Norway pout as a reference species for raising a number of other benthopelagic species including poor cod, blue whiting, argentinies and various Sebastes. In the absence of an assessment for Norway pout we choose to raise CPUE from the Q1 Sco.VIa.IBTS, by the same amount that the North Sea IBTS CPUE would be raised by in order to meet the North Sea MSVPA stock assessment for Norway pout. We obtain a biomass value of 0.56 t km^{-2} .

The P/B ratio was estimated from F , from catch and biomass data, and M (calculated from Fishbase – Froese and Pauly, 2000) at 0.78 year^{-1} . The consumption rate for Norway pout (calculated from Fishbase – Froese and Pauly, 2000) was estimated at 5.6 year^{-1} . This is comparable to the value used in Lees and Mackinson (2007), who placed Norway pout in a composite ‘small gadoid’ group, where Norway pout accounted for 28 % of that group.

The diet for Norway pout is largely planktivorous (Mackinson, 2001), with large zooplankton accounting for 75%, while crustaceans accounted for 19%.

Balancing Ecopath

In order to make the Norway pout group balance its biomass was estimated by the model with an EE of 0.95. The implied biomass was 0.833 t km^{-2} which is about 50% higher than the estimate from Sparholt raising. Given the uncertainty in this estimate this seems reasonable.

The P/B value was estimated by fixing P/Q at 0.3.

Ecosim time-series

The time series data available for Norway Pout are restricted to CPUE trends from the Q1 Sco.VIa.IBTS and catch data from STALANT. The CPUE trend shows that Norway Pout became relatively abundant for a period of years from the late 90's (Figure 8.26). The low catches, in comparison to the estimated biomass, indicate that the model is unlikely to be able to replicate this purely with the application of a fishing driver. Predation load or the abundance of prey will have to change.

A fishing driver of catch/biomass is included in the driving data for Norway pout.

8.3.15 Sprat

The sprat, *Sprattus sprattus*, is small pelagic clupeoid fish (Wheeler, 1978). They are abundant in inshore waters and estuaries particularly during the summer months when they spend their time in shallower water (Wheeler, 1978; Whitehead *et al.*, 1986).

Ecopath parameters

The sprat stock is not assessed on the West of Scotland. As it is a pelagic species, it is not well sampled by the Scottish VIa IBTS. Using the method of Sparholt with herring as a reference species returns a biomass estimate of 0.002 t km^{-2} for the biomass of sprat on the West Coast shelf in 1985. This estimate is an order of magnitude less than the official landings from the West Coast of Scotland. The time series of CPUE of sprat from the Q1 Sco.VIa.IBTS fluctuates hugely from year to year with over two orders of magnitude separating the smallest and largest values. The 1985 value is the minimum value observed.

Assuming that fluctuations in the CPUE time series represent noise in the sampling process instead of fluctuations in the population, we average CPUE over all years and calculate an alternative value of 0.1 t km^{-2} for the biomass which we use as the initial EwE estimate.

The P/B ratio was estimated from F , from catch and biomass estimates, and M (calculated from Fishbase - Froese and Pauly, 2000) at 15.1 year^{-1} . This high value was driven by a large fishing mortality (14.2 year^{-1}). The consumption rate for sprat (calculated from Fishbase - Froese and Pauly, 2000) was estimated at 8.4 year^{-1} . This compares with a value of 8.1 year^{-1} used by Mackinson (2001) in the North Sea, and Hagan and Pitcher's (2005) value of 8.5 year^{-1} for the west coast of Scotland.

As a result of a lack of quantitative data the sprat diet was based on the planktivorous diet used for this species in Hagan and Pitcher (2005). Approximately 77% of the diet consisted of large zooplankton and 23% crustaceans.

Balancing Ecopath

Sprat biomass is insufficient to support the predation mortality on it. It is assumed that the biomass estimate is inaccurate and a value of EE of 0.95 is used in Ecopath to estimate the biomass of sprat to be 1.4 t km^{-2} . This value is considerably more than the initial estimate and is about double the value from Mackinson (2007).

P/Q was initially set at 0.3 and then lowered to 0.25 in order to lower the implied P/B value.

Ecosim Time-Series

The time series data available for sprat are Q1 Sco.VIa.IBTS data and catch data from STATLANT. As noted above sprat are not well sampled by the Scottish VIa IBTS. Nevertheless the time series is included in the fitting data (Figure 8.27). In order to calculate a value for F to drive the model catch/biomass is calculated with a fixed biomass at the Ecopath value.

8.3.16 Sandeel

Four species of sandeel are prevalent in the waters off the west coast of Scotland – sandeel, *Ammodytes marinus*, smoothed sandeel, *Gymnammodytes semisquamatus*, greater sandeel, *Hyperoplus lanceolatus*, and Corbins sandeel, *Hyperoplus immaculatus*. They are known to burrow in shell and sand beds but can also be found schooling in the water column (Wheeler, 1978). The species of this group are a major prey item for many fish species as well as sea-birds such as puffins (Lythgoe and Lythgoe, 1991). Using the ratios of CPUE in the Q1 Sco.VIa.IBTS for these species we define the group composition to be 36% *A. marinus*, 35% greater sandeel and 26% Corbins sandeel. Ecopath parameters were weighted accordingly.

Ecopath parameters

All species of sandeel are poorly sampled by the Scottish V1a IBTS. The West Coast of Scotland stock assessment for sandeels was discontinued in 1996. Using the West Coast assessment (ICES 1994) we calculated a biomass of 0.90 t km^{-2} . Due to the limited nature of the fishery, spatially and in terms of under exploitation of the stock, the Industrial Fisheries Working Group noted that there were large uncertainties involved in this estimate.

The P/B ratio was estimated from F , calculated from catch and biomass estimates, and M (calculated from Fishbase - Froese and Pauly, 2000) at 0.91 year^{-1} . This value is lower than that used in the Irish sea (Lees and Mackinson 2007) of 1.53 year^{-1} . Consumption rates (calculated from Fishbase – Froese and Pauly, 2000) for the four species ranged from 3.8 year^{-1} for Corbins sandeel to 18.2 year^{-1} for smoothed sandeel. The weighted average for the group was estimated at 6.09 year^{-1} . A value of 5.02 year^{-1} was used in the Irish Sea (Lees and Mackinson, 2007). However, our estimate is lower than the value of 10.25 year^{-1} used for sandeel in the west coast of Scotland (Hagan and Pitcher, 2005) and the North Sea (Mackinson, 2001).

Diet for the sandeel group was based on information given by Bauchot (1987) which indicates that the diet is largely planktivorous. Large zooplankton accounted for 60% of the diet, while crustaceans accounted for around 40%.

Balancing Ecopath

The biomass of sandeels was estimated by the model using an EE of 0.95. This results in an estimate of 1.3 t km^{-2} which is less than 50% greater than the initial estimate from the stock assessment.

The P/Q ratio is set at 0.3 and a P/B value of 1.8 year^{-1} is estimated by the model.

Ecosim Time-Series

As noted above, sandeel are not well sampled by the Scottish V1a IBTS and hence the CPUE time series for this group is not used. The discontinued stock assessment provides a series of biomass values up to 1996 and this partial time series is used as a fitting target (Figure 8.28). Catch data from STATLANT is used but as the fishery is not exploited much these values are low.

A time series of F to drive this model group is calculated from catch/biomass with the biomass being the fixed Ecopath biomass.

8.3.17 Mackerel

Mackerel, *Scomber scombrus*, is a highly commercial species with an annual value of £64 million to the Scottish fishing fleet (MSS, 2009). It is a migratory fish and individuals only spend part of the year in ICES area VIa. Following spawning the fish migrate towards their main feeding grounds in the Norwegian Sea.

Ecopath parameters

An annual stock assessment is carried out but this is for the whole of the North East Atlantic population. In order to allocate a biomass to the shelf of ICES area VIa for 1985 we need to consider seasonal changes in the spatial distribution of the stock. There are multiple types of data source that could be considered for studying temporal variation in the spatial distribution of mackerel including acoustic surveys, egg surveys and trawl surveys (WGWISE, 2009). Each of these sources of data provides an incomplete picture by not providing complete temporal or spatial coverage. We use spatially resolved landings data to indicate the location of the stock. During each quarter, we assume that the whole stock is distributed between ICES areas in proportion to the amount of landings from these areas. During each quarter within area VIa, we again assume that the stock is distributed according to distribution of landings. Landings data resolved to statistical rectangle level provided this distribution. In combination these data provide a quarterly time series of the proportion of the stock on the shelf of area VIa and applying this to the stock assessment provides a biomass estimate. We calculated a value of 4.19 t km^{-2} for the biomass.

The P/B was calculated from natural mortality (calculated from Fishbase - Froese and Pauly, 2000) and fishing mortality to be 0.81 year^{-1} . Hagan and Pitcher (2005) used a value of 1.09 year^{-1} to achieve mass balance. Lees and Mackinson (2007) used a lower value of 0.41 year^{-1} . The consumption ratio for mackerel was estimated at 4.4 year^{-1} (from Fishbase - Froese and Pauly, 2000). This was higher than that used by Hagan and Pitcher (2005) at 3.95 year^{-1} , and significantly higher than that used in the Irish Sea of 1.73 year^{-1} (Lees and Mackinson, 2007).

The diet was based on Mackinson (2001) as applied to the North Sea Ecopath model. Mackerel is predominantly a planktivore and this was reflected in a diet matrix dominated by large zooplankton (60%). Crustaceans accounted for 18% and small zooplankton accounted for 10%.

Balancing Ecopath

In order to reduce the predation mortality on juvenile haddock, mackerel's predation on this group was reduced to 0.1% from 1% and the excess was shifted to large zooplankton. Similarly the crustacean diet component was reduced to 1% with the excess to large zooplankton and to herring.

Ecosim time-series

Instead of using the CPUE time series from the Scottish V1a IBTS, which does not sample the stock well, as an indication of trends in biomass we use trends from the stock assessment for the North East Atlantic population (Figure 8.29). In order to take account of some of the annual variation in movements of the stock we use spatially resolved catch data as described in the Ecopath parameters section to scale the total biomass in each year. We perform a similar calculation on the model estimated catch data from the assessment. We note that the biomass shows a downward trend over the period of study.

In order to drive the model we include time-series of catch/biomass (Figure 8.29) using the two series calculated above and the series of F and Z from the stock assessment. These are all shown in the figure below.

Given the wide ranging movements of this stock it is worth noting that changes in biomass may have more to do with the environment, ecosystem or fishery that the stock experiences on these migrations. It could be the ecosystem modelled in this case will not be able to account for the observed trends.

8.3.18 Horse mackerel

Horse mackerel, *Trachurus trachurus*, is a migratory, pelagic fish. The western population spawns in the Bay of Biscay and West of the British Isles early in the year and migrates to feed in the Norwegian and North Seas in the latter half of the year (Froese and Pauly, 2000). The western horse mackerel stock is assessed by ICES.

Ecopath parameters

In order to obtain a biomass for the shelf area off the West Coast of Scotland, we use the western stock assessment (WGWIDE, 2009) and estimate the portion of time that this stock spends in the area of interest. We choose to do this by approximating the distribution of the stock by the distribution of landings as with the mackerel estimate. We calculate a biomass value of 4.73 t km^{-2} .

P/B was estimated (calculated from Fishbase - Froese and Pauly, 2000) at 0.28 year^{-1} which was lower than the value 0.7 year^{-1} used for the North Sea (Mackinson, 2001). The Q/B ratio was estimated (from Fishbase - Froese and Pauly, 2000) at 3.7 year^{-1} . This value was higher than the value of 2.9 year^{-1} used by Hagan and Pitcher (2005).

Horse mackerel feed on zooplankton, shrimps and prawns (Dahl, 1987; Olaso *et al.*, 1999). The diet was based on that used by Mackinson (2001). Approximately 63% of the diet consisted of large zooplankton and small zooplankton at 21%.

Balancing Ecopath

In order to reduce the predation mortality on juvenile haddock, horse mackerel's diet component was reduced to 0.1% from 1% and the excess was shifted to large zooplankton. Similarly the crustacean diet component was reduced to 1% with the excess to large zooplankton and to herring.

Initially the P/Q ratio was fixed at 0.3, but in order to increase the EE this was then fixed at 0.2 from which the P/B value was estimated.

Ecosim Time-Series

The time-series of biomass from the Western Stock assessment is used as fitting data for Ecosim. These values are scaled from the western stock estimates to VIa estimates as described above in the Ecopath parameter section. Catch data is taken from reported international landings. A time-series of CPUE from the Q1 Sco.VIa.IBTS is shown in Figure 8.30. This does not give a good indication of changes in stock abundance so is not used in the fitting process.

A time series of F is calculated from the catch and biomass values depicted.

8.3.19 Blue Whiting

Blue whiting, *Micromesistius poutassou*, is a pelagic gadoid species that is found on the continental shelf and slope occurring at depths of 150 – 3000m (Froese and Pauly, 2000). They are a migratory species and the main stock to the west of Europe pass through the waters of ICES area VIa as part of their annual migration towards the Norwegian Sea (Reid, 2001).

Ecopath Parameters

An ICES stock assessment is carried out for this blue whiting western stock (WGWIDE, 2009). In order to arrive at a biomass value for ICES area VIa this stock assessment was scaled by an estimate of how much of the year the population spent in the area. In 2009, significant landings were only taken from VIa in the second quarter of the year, amounting to approximately 50% of that quarter's landings. We therefore assume that blue whiting are only present in VIa for 25% of the year and that 50% of the stock are there in that quarter. The population follows the shelf edge so we further assume that 50% of the population present in the area interact with the shelf ecosystem and therefore should contribute to the model dynamics. With this as justification we scale the total western stock biomass by $0.25 \times 0.5 \times 0.5 = 0.0625$ and apply this proportion to the entire time series of biomass. This results in a biomass estimate for 1985 of 1.78 t km^{-2} .

Production was estimated at 0.59 year^{-1} , as calculated from M (from Fishbase - Froese and Pauly, 2000) and F . F was approximated by $\text{catch}/\text{biomass}=0.22\text{year}^{-1}$ for the whole of the western stock. The consumption rate for blue whiting was estimated from Fishbase (Froese and Pauly, 2000) to be 9.06 year^{-1} . This is comparable to the Q/B estimate used for the ICES VIa deepwater fisheries model (Howell, Heymans *et al.* 2009).

The diet of blue whiting is largely based on pelagic crustaceans and large zooplankton such as euphausiids (Bergstad 1991; Mackinson 2001) although a shift in diet from juveniles to adults has been identified (Bergstad 1991). Smaller individuals tend to feed on large zooplankton, mysids, euphausiids and crustacean's larvae while the diet shifts in larger individuals to include some piscivory. The diet used in the model is based on that used by Mackinson (2001) with 25% of the diet consisting of crustaceans, 35% from large zooplankton and 15% from the 'other pelagics' fish group.

Balancing Ecopath

The consumption of crustaceans by blue whiting was too high. This diet component was reduced to 1% with the excess to large zooplankton. The sandeel component of the diet is reduced from 10% to 1% in order to reduce the predation load on sandeels. The excess is split between herring and mackerel. Cannibalism was reduced from 5% to 2% in order for the group to balance.

Initially P/Q was fixed at 0.3 but this led to a very high P/B value. The P/B value was fixed at 1.5 year^{-1} and the P/Q ratio was set at 0.25. This led to a consumption (Q/B) estimate of 6 year^{-1} .

Ecosim time-series

As described above in the Ecopath parameter section, in order to obtain a fitting time series for the biomass we use the stock assessment for the Western stock and scale it to the shelf of area VIa. The result of this scaling is shown in Figure 8.31. Catch data is taken from STATLANT and in the absence of spatially resolved catch data an assumed 50% of the catch is assumed to be from the shelf.

Two time series of F are supplied (Figure 8.31b). The first results from $\text{catch}/\text{biomass}$ as calculated above. The second is the mean F (for ages 3-7) from the stock assessment. The $\text{catch}/\text{biomass}$ time-series was used to drive the model.

As with other migratory species included in the model, much of what drives the blue whiting biomass may be occurring outside of VIa and hence is not included in this model.

8.3.20 Other Pelagics

The 'other pelagics' group includes pelagic species caught in the Scottish VIa IBTS survey that are otherwise uncategorised. These species include Alice shad, *Alosa alosa*, twaite shad, *Alosa fallax*, sardines, *Sardina pilchardus*, anchovy, *Engraulis encrasicolus*, and pearlsides, *Maurolicus muelleri*.

Ecopath Parameters

Production was estimated by calculating M (from Fishbase - Froese and Pauly, 2000) and F for each species then devising a weighted value for the group (see above). A value of 6.4 year^{-1} was estimated. Consumption rate (estimated from Fishbase - Froese and Pauly, 2000) for the species in this group ranged from 2.8 year^{-1} for Alice shad to 15.5 year^{-1} for pearlsides. The consumption rate for anchovies was 9.13 year^{-1} . The weighted average for the group was estimated at 10.76 year^{-1} .

The diet for the 'other pelagics' group was based on Mackinson (2001). Approximately 37% of the diet consisted of epifauna and 14% infauna, while 10% came from Nephrops and crustaceans.

Balancing Ecopath

A number of adjustments were made to the other pelagic diet. The crustacean and Nephrops components were reduced to 1% and 0.1% respectively from 10% with the excess split between large zooplankton and cephalopods. The juvenile haddock component was reduced to 0.1% and the gurnard component was reduced to 2% from 5% with the excess moved to herring.

Further adjustments to the parameterisation of this group were that the Q/B was reduced from 10.76 year^{-1} to 6 year^{-1} , the EE was fixed at 0.95 from which an implied biomass was calculated and the P/Q ratio was fixed at 0.3.

Ecosim Time-Series

For the other pelagic group no reliable time-series data exists. Both the catch data from STATLANT and the CPUE index from the Scottish VIa IBTS fluctuate dramatically from year to year. No merit is seen in including them as driving or fitting data so they are omitted. The fishing mortality is assumed to remain constant at the, very low, Ecopath level.

8.3.21 Grey Seals

Biomass

A pup count of grey seals (dashed line in Figure 8.32), *Halichoerus grypus*, is carried out annually (SCOS, 2008). It is believed that this count is accurate to within $\pm 13\%$. This pup count has been linked to total population size through the use of various models the outputs of which vary considerably. Two of these models, the selection of which was based on parsimony (SCOS, 2008), were used to create an averaged posterior mean estimate for the population size in 2007. This estimate was presented with large confidence limits (factor of 2 on either side) to convey the uncertainty in the modelling process. With the information available in the report it was not possible to calculate such an average for 1985. Time series of numbers for the Inner and Outer Hebrides resulting from both models are presented and instead we calculate a naïve average of the two models giving a value of 38.8 thousand individuals on the West Coast for 1985. Tires and Pauly (1998) estimate the mean weight of a male grey seal to be 168kg and that of a female to be 152 kg. We assume a sex ratio of 1:1 and set the mean weight of a grey seal to be 160 kg. This gives a biomass density of 0.056 t km^{-2} .

In order to provide a time-series of grey seal biomass to fit the Ecosim model to, this calculation is carried out over the whole period of interest and the results are presented in Figure 8.32.

Consumption

In SCOS (2008) it is stated that the food requirements of grey seals will depend on the oiliness of the prey. Values of 7 kg of cod for an average seal or 4 kg of sandeel are offered as daily consumption estimates. Through analysis of scat samples Hammond and Harris (2006) estimated that grey seals in the Hebrides consumed on average 4.99 kg of fish per day in 2002 and that this had not changed from 1985. This value implies an annual consumption/biomass ratio of 11.4 year^{-1} . (Trites and Pauly, 1999) equate the mean daily ration of a marine mammal as a percentage of body weight to:

$$R=0.1W^{0.8}$$

This results in a consumption/biomass ratio of 21.2 year^{-1} . We use the former value which is based on local data.

Production

The production rate for the group was calculated by the model by fixing the P/Q ratio at 0.01. This yields an estimate of $P/B=0.101 \text{ year}^{-1}$ which is comparable to the values used in the North Sea model (0.09 year^{-1} - Mackinson and Daskalov, 2007) and Irish Sea model (0.06 year^{-1} - Lees and Mackinson, 2007) for their seals composite functional group.

Diet Composition

Hammond and Harris (2006) estimated the diet composition of grey seals through the analysis of scat samples in the Hebrides. They re-worked samples collected in 1985 and analysed samples from 2002. In 1985 45% of the diet was made up by sandeels and about 10% each of ling, cod and various flatfish. In 2002 only 30% of the diet was made up by sandeels. The herring component of the diet increased from about 5% to 15%. Four functional groups which did not appear in the 1985 diet featured in the 2002 diet: other benthopelagic feeders, poor cod, blue whiting and sprat. We use the diet composition calculated for 1985 with very small percentages for the for the four extra functional groups in order that these might feature more strongly in the diet as the modelled ecosystem structure changes during the Ecosim simulation.

8.3.22 Harbour Seals

Biomass

Numbers of harbour seals are estimated from counts at haul out sites. The Sea Mammal Research Unit (SMRU) considered it impractical to carry out a survey of the entire coast annually, so they aimed to complete the survey every five years (SCOS 2008). This data extends back as far as 1992 for the Outer Hebrides and 1994 for the Scottish Highlands. We assume that the Highlands' count is for the West Coast of the Highlands and that from 1985 to 1992/1994 numbers stayed constant. It is estimated that 30-40% of the seals are not counted during the surveys due to seals being in the water and estimates are revised up to account for this. We arrive at a number of 18,100 harbour seals on the West Coast of Scotland, and assuming an average weight of 63.6kg (Trites and Pauly, 1998), this gives a biomass density of 0.010 t km^{-2} .

The resulting time-series of biomass for this calculation is shown in Figure 8.32 and this is used as a fitting target for Ecosim.

Consumption

SCOS (2008) comments that 3 kg of fatty fish or 5 kg of gadoids are considered a likely daily ration. A 4 kg daily ration results in an annual consumption/biomass ratio of 23.0 year^{-1} .

Trites's formula (Trites and Pauly, 1999) results in an annual ratio of 10.1 year^{-1} . We use the latter value.

Production

The production rate for the group was calculated by the model by fixing the P/Q ratio at 0.01. This yields an estimate of $P/B=0.101 \text{ year}^{-1}$ which is comparable to the values used in the North Sea model (0.09 year^{-1} - Mackinson and Daskalov, 2007) and Irish Sea model (0.06 year^{-1} - Lees and Mackinson, 2007) for their seals composite functional group.

Diet composition

The diet composition of harbour seals is based on values taken from (Pierce and Santos, 2003). Five faecal sampling periods during the period 1993 to 1994 on Mull and Skye were used to determine diet composition. Haddock, whiting, herring and horse mackerel were found to make up the bulk of the diet with sandeels only contributing a small portion of the diet, unlike in other areas of Scotland.

8.3.23 Cetaceans

The production rate for this group was set at $P/B = 0.02 \text{ year}^{-1}$ which is the value used for both the toothed and baleen whale cetacean groups in the North Sea and Irish Sea models (Lees and Mackinson, 2007; Mackinson and Daskalov, 2007). For the consumption rate, an average the values from the North Sea model were averaged to give $Q/B=14 \text{ year}^{-1}$.

The diet, including both piscivorous and planktivorous components, was taken from the previous West Coast of Scotland model (Hagan and Pitcher, 2005).

The biomass for this group was estimated at 0.014 t km^{-2} by the model using an Ecotrophic Efficiency of 0.1.

No changes were made to the group's parameters during balancing.

8.3.24 Birds

Values from (Barrett *et al.*, 2006) imply that the make-up of this group by biomass should be 36% auks, 30% petrels, 22% pelecaniformes and 12% gulls.

Biomass

(Barrett *et al.*, 2006) estimates the total biomass of seabirds in ICES Region 5 (encompasses areas Vibe, VI and VII) to be 9500 tonnes. From data in (WGSE, 2002) we calculate that VIa contributes 40% of the biomass of breeding (does not include immature)

birds to that of Region 5. Applying this percentage to the (Barrett, Chapdelaine *et al.* 2006) estimate gives a biomass density of 0.034 t km^{-2} . The JNCC states (Mitchell *et al.*, 2002) that numbers of breeding birds in Britain has risen from 6 million, in 1985-1988 to 8 million in 1998 to 2002. We assume that this trend is applicable to seabirds on the West of Scotland and reduce this biomass density to 0.025 t km^{-2} for a 1985 estimate.

Consumption

From values in Barrett *et al.* (2006) we estimate an annual consumption/biomass ratio 90.5 year^{-1} if seasonal movements of birds are not taken into account or 97.6 year^{-1} if they are. From (WGSE, 2002) we estimate a ratio of 83.1 year^{-1} . We use an average of these values giving a consumption of 90 year^{-1} .

Production

The production rate for the seabird group is set at $P/B=0.4$ which was taken from (Trites and Pauly, 1999).

Diet Composition

The diet of the seabird group was based on the (Hagan and Pitcher, 2005) Ecopath model. The diet of components of seabirds for each of the multi-stanza gadoid groups was assumed to be from the immature stanzas as it was felt that this was a more likely predator/prey interaction.

Ecopath Balancing

The other small fish component of the diet of seabirds was reduced from 9% to 3% with the excess split between blue whiting and mackerel.

8.3.25 Nephrops

Ecopath Parameters

The Norway lobster, *Nephrops norvegicus*, is assessed in three Functional Units on the West Coast of Scotland; the Firth of Clyde and the North and South Minches (WGCSE 2009). Although Nephrops do occur elsewhere within ICES area VIa, including on the shelf edge, the vast majority of the biomass is believed to be accounted for by these areas with less than 3% of the landings since 1985 coming from elsewhere. The assessment has been carried out since 1995 using an Underwater TV Survey. The density of burrows is calculated and this is scaled to give an abundance estimate. We have used the mean landings weights provided together with the assessment to calculate a biomass of 0.47 t km⁻² for 1995 and have assumed that the population remained constant at this level before then.

A consumption rate, $Q/B=4.876 \text{ year}^{-1}$, and production rate, $P/B=0.73 \text{ year}^{-1}$, were taken from (Lees and Mackinson, 2007).

Balancing Ecopath

In order to balance the model the biomass was estimated by fixing the *EE* at 0.95. This resulted in a biomass value of 0.796 t km⁻², approximately 70% higher than the initial estimate. Given the uncertainty in the initial estimate this does not seem unreasonable.

Ecosim Time-Series

The biomass time-series for Nephrops, only partially extends over the period of interest (Figure 8.33). It is calculated by summing the stock assessments for the three Functional Units on the West Coast. Given the required increase in biomass in the Ecopath balancing this time series is used to indicate relative abundance changes only. The catch data is taken from a combination of STATLANT data and stock assessment data. Discard estimates are provided in the stock assessment and these are included. Prior to the start of the

assessment a constant discard rate is assumed, at the average value for the assessed years.

A time series of F is calculated from catch/biomass using the balanced Ecopath biomass in all years.

8.3.26 *Other Crustaceans*

Ecopath parameters

This group was formed from an amalgamation of the prawns and shrimps group and crabs and lobsters group of the original West Coast model (Hagan and Pitcher, 2005). The diet was constructed from the diet breakdown of both these groups in the West Coast model. The consumption and production rates for this group were based on parameters from the Irish Sea model (Lees and Mackinson 2007) resulting values for this composite group of $P/B = 0.871 \text{ year}^{-1}$ and $Q/B = 5.81 \text{ year}^{-1}$.

Balancing Ecopath

In order to reduce the crustacean biomass necessary to support the predation load, the diets of many of the predators of crustaceans were altered. These changes are described in the sections for each species.

Cannibalism by crustaceans was reduced to 1% from 13% again to lower the implied biomass of the crustacean group. The Nephrops component of the diet of crustaceans was reduced from 5.24% to 0.1% with the excess to large zooplankton. The small benthic-pelagic fish component of the diet of crustaceans was reduced to 0.1% from 5.14% with the excess to infauna. Poor cod formed 0.1% of the diet of crustaceans and yet this accounted for 40% of the consumption of poor cod. This was reduced to 0.01% and the excess moved to infauna.

8.3.27 *Epifauna and Infauna*

The parameterisation for the Epifauna and Infauna functional groups were taken directly from the original West Coast model (Hagan and Pitcher, 2005).

In order to make the Epifauna group balance the cannibalism was reduced to 20% from 37.5% and the excess moved to detritus.

8.3.28 *Squids*

The consumption ($Q/B=15 \text{ year}^{-1}$) and production ($P/B=1.981 \text{ year}^{-1}$) values for the squid functional group were taken from the Irish Sea Ecopath model (Lees and Mackinson, 2007).

The diet was based on the previous West Coast of Scotland model (Hagan and Pitcher, 2005).

A biomass was calculated by the model by setting $EE=0.95$. In order to balance the model the other small fish component of the diet of squid was reduced from 1.8% to 0.5% with the excess split between herring and blue whiting. The predation mortality of juvenile haddock needed to be reduced in order that the group balanced. The cephalopods diet component was reduced to 0.05% and the excess moved to herring

8.3.29 Zooplankton (Large and Small)

Two zooplankton functional groups were included in the model as in the previous West Coast model (Hagan and Pitcher, 2005). The parameterisation for both groups was taken directly from that model.

8.3.30 Phytoplankton

The parameterisation for the phytoplankton group was initially taken from the previous West Coast model (Hagan and Pitcher, 2005). The biomass estimate of 80 t km^{-2} was felt to be too large when compared with other models (7.5 t km^{-2} - Mackinson and Daskalov, 2007, and 10 t km^{-2} - Lees and Mackinson, 2007) so this was instead estimated by the model with an Ecotrophic Efficiency of 0.95. The biomass required to satisfy consumption was still not as low as these other models at 32 t km^{-2} .

8.3.31 Detritus

The biomass of the detritus group was set at 100 t km^{-2} as in the previous West Coast model. This value is comparable to the sum of the two detritus groups in the Mackinson and Daskalov (2007) and Lees and Mackinson (2007) models.

8.4 Ecosim Model Fitting

8.4.1 Initial Fitting Results

Using the model parameterised and then balanced in Section 8.3 as the starting point, we drive it with fishing mortality time-series and compare the output with our sets of fitting data (biomass and catch).

In Figure 8.34 the solid black lines show the model output for the mature haddock stanza. The increases and decreases in biomass match with the data, insofar as the timing matching up, but the amplitude of the variation is not replicated and hence the biomass and catch are always overestimated by the model. The dashed and dotted black lines show the results of fitting this base model to the data using the automatic fitting procedures within EwE. The dashed line shows the results of fitting vulnerability parameters only and the dotted line is

the result of fitting vulnerability parameters and a primary production forcing function or “anomaly”.

For mature haddock the vulnerability parameters are automatically adjusted to be very high and an improved fit is the result with the biomass lower than without any fitting. Forcing the primary production to follow a fitted anomaly produces little improvement to the haddock model output but over all model functional groups it produces a quantitative improvement in the sum of squares error. Despite these improvements in model fit the mature haddock biomass is still overestimated and the size of fluctuations seen in the data is not reproduced. Noticeably the initial steep decline in haddock biomass is not reproduced.

8.4.2 Biomass Accumulation Adjustment

In Section 8.3 the model was set up so that it was in equilibrium. The assumption was that the biomass of all species did not alter through 1985. The stock assessment for haddock (WGCSE, 2009) shows that haddock spawning stock biomass peaked in 1982 and was in decline over the following 10 years. The implication being that the haddock population in 1985 was not in equilibrium. This can be handled in Ecopath by allocating a non-zero value for the Biomass Accumulation parameter. A negative value for the parameter implies that the biomass decreases during the year. We set the Biomass Accumulation for haddock to be the difference in biomass between the 1985 and 1986 values.

Given that we are doing this for haddock, for consistency we consider the other two model focal species of this project. Figure 8.4 shows that cod biomass has been in decline for most of the period of study. The decrease from 1985 to 1986 is steep and not representative of the rate of decline over the whole period, indeed the biomass recovers in 1987. Instead of using the decrease in the first year, we choose to set the Biomass Accumulation for cod to the average over all the years of the study. Figure 8.4 shows that for most of the first 6 years the mature whiting biomass is in decline before increasing rapidly. We choose to set the Biomass Accumulation to the decrease in biomass observed in the first year as with haddock. For the other model functional groups we choose to leave the Biomass Accumulation at zero.

As a result of these adjustments the Ecopath model becomes unbalanced. This is perhaps counterintuitive as in order to achieve a decrease in biomass across the year there needs to be an increase in mortality. If the predation and fishing loads stay constant an increase in “other mortality”, or Ecotrophic Efficiency, is required and the group should become “more” balanced. For multi-stanza groups this dynamic is complicated and all of the juvenile stanzas in this case become unbalanced. In order to balance the model again the relevant diet components of each of the predators of the juvenile stanzas were reduced by the same required amount and in each case the excess was redistributed evenly across their other prey items.

The time series of fishing mortalities were then applied to this new model and the result for haddock is shown by the solid green line in Figure 8.34. The difference between this and the solid black line, representing the model without the Biomass Accumulation terms, is stark. Now, there is insufficient production in the haddock group to account for the mortality applied and the biomass shows a steep decline which slows but never recovers. The two periods of increase in biomass observed in the data and in the earlier model output are no longer apparent.

Fitting the model to the data with vulnerability adjustments and a primary production anomaly results in the periods of decrease and increase returning (dashed and dotted green lines in Figure 8.34). The overall fit to the haddock data is perhaps an improvement on the model without the Biomass Accumulation terms in. A comparison between the sum-of-squares errors between the two model set-ups shows little change. The vulnerability parameters for haddock which govern how it interacts with other model species go from being very high to very small in the model with Biomass Accumulation terms. This is significant because these parameters govern how the functional groups respond in the scenarios described in Section 8.6.

8.4.3 Forcing Recruitment

By setting up the model so that it is not initially in equilibrium we were able to reproduce the steep decline in haddock biomass that is witnessed in the initial years of the model. Adjustments needed to be made to the haddock vulnerability settings in order to reproduce the increases and decreases which followed.

The sharp increases that are observed in haddock biomass are associated with large year classes and the mechanisms which lead to these are not well understood but spatial mechanisms have been proposed. If we use an ecosystem approach to produce a result which is actually caused by something outwith the scope of the model then the way the model responds in the scenario tests may be affected in a fallacious manner.

An alternative is to accept that we are unable to reproduce the dynamics that are seen and instead force the egg production to be sufficient to produce the observed recruitment. This was investigated in two different ways in EwE. The first method was to import the biomass data for the juvenile haddock stanza as forcing data instead of fitting data. Whilst this produced the exact pattern of juvenile biomass, little impact was observed on the adult stanza but it is unclear to the authors why this is so. EwE can be used to model stocked populations referred to as Hatchery Populations in the EwE User's Guide. A forcing function can be supplied which drives the size of the population at the start of the juvenile stanza relative to the size calculated in Ecopath.

As an alternative to forcing the juvenile stanza biomass we used a time-series based on numbers-at-age-zero back calculated from the stock assessment numbers-at-age-one and

an assumed mortality rate to force the population size at the start of the juvenile stanza. As a result the juvenile biomass does not necessarily match up exactly with the juvenile biomass data but, unlike with forcing the juvenile biomass directly, the effects of variable recruitment are propagated through to the adult stanza.

The result of this exercise for haddock is depicted by the solid black line in Figure 8.35 and for whiting in Figure 8.36. By annually forcing the initial size of the juvenile stanza and with no adjustments to the vulnerability parameters we are able to produce more of the dynamics that are seen in the data. Altering vulnerability parameters and fitting a primary production anomaly (dashed and dotted lines) produces a quantitative improvement in model fit but qualitatively for the mature haddock biomass output it is hard to argue that the model output is an improvement. For the middle years of the run the output is closer to the data but towards the end the model further overestimates biomass.

While it is possible to produce an arguably better fit to the data by forcing haddock and whiting recruitment we choose not to use this approach for the “what if” scenarios investigated in the next section. Some of the scenarios investigated involve forecasts and we do not know how recruitment will behave in the future. The scenarios which do not involve forecasts do involve making adjustments to the populations as they were and there is no reason to expect recruitment to have behaved in the same manner if the conditions were different.

8.5 Baseline Model Results

Having investigated how further adjustments to the model affect the fit of the model output to data we present the results from the model with Biomass Accumulation terms included for the three main species of interest and we go on to use this version of the model in the scenario testing section.

For cod the biomass accumulation rate was set at the average rate of decline over all years. Without fitting the model by adjusting the vulnerability parameters, this initial rate of decline does not continue and the biomass, and hence the catch, are overestimated (Figure 8.37 – solid line). Adjustments to the vulnerability parameters for cod to be very high so that it becomes a top down predator results in a much better fit (dashed line). The decline in modelled mature cod biomass is also reflected in the juvenile stanza. Using a forcing function for primary production makes little difference to the quality of fit.

For haddock (Figure 8.38), as discussed above, the first few years of the unfitted model simulation produce a good fit to the mature haddock data. In order to fit the later data better adjustments are required to the vulnerability settings. The variation in immature haddock biomass is never well fitted by the model.

For whiting (Figure 8.39) we are unable to reproduce the sharp increase in biomass noticed at the beginning of the 1990's. Automatic adjustments to fitting parameters are insufficient to produce this increase. As with haddock, using a hatchery approach to drive the egg production for whiting was investigated to see whether the pattern in mature whiting biomass could be explained by patterns in recruitment that were not possible to otherwise model. This was not as successful as for haddock and the peak in whiting biomass was not achieved.

It is well established that grey seal numbers have increased in recent years all around the British Isles including on the West coast of Scotland (SCOS, 2008). The Ecosim model reproduces an increasing grey seal biomass. The default model, without any changes to vulnerability settings or environmental forcing, produces an approximate 40% increase in grey seal biomass from 1985 to 2008 (Figure 8.40). This increase is not as large as that in the single time-series we used as fitting data but it does fall within the range of model outputs for population trends presented by SCOS (2008). In the cases where vulnerability parameters and a time series of primary production were estimated to produce the best fit to the time series data for all species the grey seal biomass climbs to a similar level to before but slightly more quickly. In all cases the predation loads on cod and, to a lesser extent, haddock increase through the simulation.

As well as looking at using the automatic optimisation procedure in Ecosim to produce vulnerability estimates for a number of the interactions, a search was also conducted for a primary production driver concurrently with the vulnerability search described above. The model output resulting from this exercise is shown by the dotted lines in the figures. The resulting forcing function for primary production is shown in Figure 8.41. There is little obvious graphical improvement to the three species above but the log-likelihood sum-of-squares, which measures the difference between the model outputs and fitting datasets, does show a decrease. While the haddock biomass is still too high the downward trend in biomass in recent years is reproduced as a result of using this function to force primary production.

8.6 Ecosim Scenario Testing

In order to use the model developed to explore possible explanations for the current state of the gadoid stocks of interest and how certain management decisions may affect the ecosystem a number of retrospective scenarios were designed. The way the model is set up has a substantial impact on how the model responds to changes in external driving and for this reason the output is presented for each scenario with two of different set-ups: firstly the model with default vulnerability parameters and no primary production and secondly the model with vulnerability and primary production values fitted. The results for each of the scenarios are presented with the default model in black and the fitted model in green.

8.6.1 *Grey Seal Population Control*

In order to investigate how much impact the increase in grey seal biomass has had on the ecosystem, two scenarios were investigated and the results for cod are shown in Figure 8.42. In the first, mortality was imposed on the grey seal population sufficient to prevent the population biomass from rising above the 1985 value for the duration of the simulation (dashed lines). In the second, a high mortality was imposed so as to remove all predation from grey seals from the model (dotted lines). Both of these scenarios were applied to the default (black) and fitted (green) model set-ups.

In both model set-ups the cod stock shows slightly higher biomass trajectories for the first scenario where seal biomass was kept constant. Referring back to the grey seal biomass plot (Figure 8.40) we note that the modelled biomass did not climb to the level as described by the data and that it did not start to increase until about 1995. This partly explains the small size of the response of cod biomass to the changes in seal biomass. A curb to the decline in cod biomass is achieved in the second scenario. The biomass starts to decrease in the second half of the simulation which appears to be because the larger cod population depletes some of its preys' biomasses.

In the case of haddock and whiting, curbs placed on the grey seal population have insufficient impact to their biomasses for an affect to be noticed graphically.

In summary, if the grey seal predation on cod was held at its 1985 level there would be a very small increase of cod abundance and catches in 2008 compared to what actually was observed. However, if grey seal predation on cod was completely removed, the model estimates that in 2008 cod abundance and catch would have been approximately four times as large as they actually were in that year, although these predicted larger cod catches and abundances were still lower than cod catches and abundances in 1985. The model predicts that cod biomass would have increased substantially in the 1990s if all grey seal predation was removed, but also indicates a rapid decline to the enhanced 2008 levels presumably owing to the observed fishing pressure time series which is used in the model coupled with a consumption of cod prey.

8.6.2 *Nephrops Trawl Selectivity Measures*

In order to investigate the effects of the Nephrops trawl fleet on the West coast of Scotland ecosystem the fishing mortality on all species except for Nephrops was reduced in order to simulate a clean Nephrops fishery.

In order to parameterise this change in selectivity the landings of all species as well as the discards of cod, haddock and whiting by Scottish Nephrops trawlers were removed from the catch data used to calculate the time series of fishing mortality for every fish species in the model. The results of applying these revised fishing mortalities to all fish stocks are shown

for cod, haddock and whiting in Figure 8.43-45 for both the fitted (green) and the unfitted (black) models.

The results show that removal of the landings by Scottish Nephrops trawlers makes little difference to the fishing mortality for mature haddock and whiting. The small changes in their biomass and catch trajectories are as a result of changes in fishing mortality on the juvenile stanzas and on their prey and predators. The fishing mortality for cod is reduced and this leads to small increases in biomass and overall catch in the 1990s when compared to the scenario with a Nephrops fishery which has by-catches of fish. However, these increases disappear in recent years.

8.6.3 *Fishing at Maximum Sustainable Yield*

A goal of future fisheries management is to operate all fisheries by fishing at maximum sustainable yield. Here we investigate the impact to the gadoid stocks of a change to fishing in such a manner.

Values for F_{MSY} are calculated during stock assessment and are supplied for cod (0.19 year^{-1}) and haddock (0.3 year^{-1}) in the ICES advice sheets (ICES 2010). For the purposes of this forecast F_{MSY} for whiting was assumed to take the same value as for haddock. These values of F_{MSY} are instantaneous mortality rates. To use the values in EwE we convert them to harvest rates giving values of 0.17 year^{-1} (cod) and 0.25 year^{-1} (haddock and whiting). These rates are applied from 2009 for a period of 15 years, while the fishing mortality for all other species remains at the level calculated for 2008.

The results of this exercise are shown for cod, haddock and whiting in Figure 8.46-48. As an illustration of how the model responds differently to changes in the driving data depending on how it is initially set up we show the results of applying this scenario both to the initial model which was set up in equilibrium (black) and perhaps the more plausible model which was adjusted so that the biomass of the three gadoid species was in decline (green). For cod and haddock a dotted line representing B_{pa} , biomass for a precautionary approach, is included (ICES, 2010).

If a change to fishing at F_{MSY} is implemented in the model the biomass of mature cod quickly starts recovering. In the case of equilibrium model the stock recovers more slowly and it starts to plateau below B_{pa} , whereas the biomass accumulation adjusted model continues to rise beyond B_{pa} but this takes over 20 years (not graphed). Despite the decrease in fishing mortality the catches also recover but not to previous, unsustainable levels.

In the case of haddock the biomass and catch both increase on the imposition of F_{MSY} . There is a big difference in how the model responds in the two model set ups. These different rates of recovery result from differences in vulnerability parameters in the two models. In the case of the model initially in equilibrium the vulnerability needed to be

adjusted to drive down the initial biomass. With the model altered to be initially out of equilibrium, automatically reproducing the initial decline in haddock biomass, the vulnerability instead needed to be adjusted to produce the spike in biomass resulting from years of high recruitment. The way this model set up responds does appear more plausible.

For the whiting stock the biomass increases on imposition of F_{MSY} but the catch remains at roughly the same level. The fact that the model does not reproduce the steep decline in whiting biomass that is apparent in the data indicates that the modelled whiting is able to support the current fishery. It is unsurprising then that this decrease in F to F_{MSY} produces a reduced catch.

8.7 Discussion

This investigation set out to use a particular approach to broaden our understanding of cod, haddock and whiting population dynamics on the West Coast of Scotland. The first goal was to reproduce the biomass and catch time-series from the stock assessments. Then we looked at how the model responded to various hindcast and forecast scenarios which were designed to explore hypotheses for poor gadoid performance on the West Coast.

In order to do this we have parameterised an Ecopath with Ecosim model, building on previous work on an EwE model for the West Coast (Hagan and Pitcher 2005). Much of the diet information used to parameterise that model was used in this work. Whereas the previous work used the model to estimate biomass values, here more effort was made to make local estimates of biomass for a large number of the functional groups using survey data and stock assessment values. In balancing the Ecopath model many of these estimates needed to be altered but the authors tried to keep the adjustments within reasonable bounds of the local estimates. The main difference between this investigation and the previous one was the use of time series data, again derived from surveys and stock assessments, to drive and fit the Ecosim model.

8.7.1 Modelling Conclusions and Further Work

The default setup of an Ecopath model is for each functional group to be in equilibrium with respect to production and mortality. In Ecosim, changes in biomass for a particular functional group result from changes in mortality or prey biomass. Using this default we were unable to reproduce some of the key features of the biomass trends, notably the initial steep decline in mature haddock biomass. In order to reproduce this decline in Ecosim it was necessary to assume that the mortality was initially greater than the production for the haddock functional group, or in the terms of Ecopath, that there was a negative Biomass Accumulation. Making this assumption for cod and whiting produces different fits without clear improvements. It should be noted that further consideration could be applied to all functional groups within the model in this regard but this is left for future work.

As well as some improvements in the fit to data being achieved through the use of biomass accumulation terms in the initial model setup, noticeable gains were achieved through the use of an automatic fitting routine within Ecosim which adjusts vulnerability parameters. These parameters govern how changes in functional groups' biomasses affect the interactions between them with the two extremes being described as "top down" and "bottom up" (Christensen *et al.*, 2005). The automatic adjustments to the parameters depend on how the data differs from model output and in turn on choices made about how the Ecopath model is set up (such as the inclusion of biomass accumulation terms or not).

In order to draw conclusions from the fitted values it is therefore necessary to have a degree of certainty, obtained from both well founded choices in the initial model set up and indication that these choices above all others lead to an improvement in model fit. While some improvement in model fit was observed it is felt that a more exhaustive exploration of justifiable set-ups and a sensitivity analysis on the model outcome is necessary before direct conclusions can be drawn from the fitted vulnerability values. As is noted in the scenario testing section the values chosen for these parameters can play a large roll in any hindcasting or forecasting.

The other avenue that was explored to produce a better fit to the data was the use of a driving function for primary production. Again, an automatic fitting routine within the EwE software was used to find a time series which results in the best fit to data for all functional groups. The result was a quantitative improvement in the fit but the qualitative improvements were small. A cursory comparison was made between the fitted forcing function and ocean colour index time series as well as North Atlantic Oscillation time series, but no correlation was observed. More work could be done in this area. It was unclear whether the automatic fitting procedure was just fitting to noise in the fitting dataset as opposed to more general trends observed. Perhaps an investigation into how using smoothed sets of fitting data affects the fitted forcing function could provide further insight.

The resulting fits to the biomass and catch data for the mature stanzas of cod, haddock and whiting show some merit. The broad trends are picked out but the amplitude in the variation, particularly for haddock and whiting, is not as high as observed in the stock assessments. Looking at the juvenile stanzas for haddock and whiting suggests a reason for this as the model seems to be unable to reproduce the annual variation in recruitment. An investigation into forcing recruitment was carried out with good results but this was not carried forward for the scenario tests as it is not known how recruitment would have behaved in the light of the different conditions of the scenarios.

The scenarios constructed help further understand the population dynamics and provided some interesting results. The investigation into controlling the grey seal population suggests that while grey seals provide a considerable source of mortality to cod the rise in their population over recent decades has not been the reason for the decline in the cod stock. The rise in grey seal biomass from the fitted EwE model was at the lower end of that

estimated by the assessment (SCOS, 2008). Forcing the grey seal biomass to follow a higher trajectory might have resulted in more of the decline in the cod biomass being explained by the rise in the grey seal population. This was not investigated in the current work.

The investigation into the impact of the Nephrops trawl fleet on the three gadoid stocks shows that there is insufficient bycatch by the fleet to have a large impact on the stocks. Moving to a clean Nephrops trawl results in very little change to the haddock and whiting biomass trajectories whilst a small increase is observed in the cod biomass trajectories.

The final scenario investigated involved making a forecast of biomass and catch trajectories on the basis that the stocks are fished at maximum sustainable yield (F_{MSY}). The construction of this scenario involves making a number of assumptions. In the stock assessments F_{MSY} values have been estimated for assessed stocks and here an assumption is made that these values of F_{MSY} can be achieved in the context of a mixed fishery. For the other functional groups we assumed that F remained constant at the most recent value. This is unlikely to be the case but more work is necessary to estimate the impact that changing to F_{MSY} for some stocks would have on effort and in turn on the fishing mortality on other stocks.

The forecasts are carried out on two model variants and for the haddock forecast in particular there are wildly different outcomes. The way the model responds depends on the fitted vulnerability parameters. Given the more plausible model set up we see cod, haddock and whiting biomasses increasing. Cod biomass eventually returns to historically high values and the catch rises but does not achieve the previously observed levels. Haddock biomass returns more quickly to historically high values and the catch shows a limited increase. For whiting the catch shows very little recovery despite an increase in biomass.

8.7.2 General Comments

For a first time user building an intuition about what is important in the construction of an Ecopath model and sourcing data to parameterise it takes time and guidance from experienced users is certainly useful on both counts. The Ecopath documentation and software is clearly well thought out. However, the representation of “ F ” as a fishing mortality when it is meant as a harvest rate is misleading. The documentation is unclear on this detail and perhaps inconsistent.

This project has focussed on three particular species and as a result these three fish species were each resolved into two age classes or stanzas. Care was taken in parameterising these three species as well as those functional groups which were proximate to them in the food web. Parameterising the other groups, which do not impact much on the species of interest, still takes time and thought. Part of the strength of the modelling approach adopted

is that effects can propagate through the food web to produce results which might not otherwise have been predicted or explained

The parameterisation of the whole food web is also one of the disadvantages of using the adopted approach to answer the questions we set out to answer. Parameterisation of parts of the food web remote to our focus not only takes time but it diverts attention away from these questions. Instead of thinking about mechanisms which could lead to patterns in observed data and tailoring a model to test them, it is easy to adopt the mentality of hoping that the model will somehow be able to replicate the data and then trying to derive an explanation from the model behaviour. While this is not necessarily a bad approach, confidence that the data underpinning the model is representative of a reality, or a good understanding of how sensitive the model is to the parameters which govern the size of groups and flows between them, seems to be crucial. This sensitivity has not been accounted for in this work but has been investigated elsewhere (Essington, 2007).

In order to test the scenarios developed in Section 8.6 it was necessary to have a fitted Ecosim model. Fitting the model and implementing the scenarios have been the main goal. In doing so the construction and balancing of the Ecopath model had to be done fairly quickly. Re-visiting this part of the process would be informative and perhaps more time should be spent exploring the inevitable inconsistencies in the original data set and the variety of ways in which these can be resolved. An exploration of how the resulting variety in Ecopath models goes on to perform in Ecosim simulations would be interesting and could give a better idea about how reliable the Ecosim predictions are.

If multispecies models are to become a greater source of advice for fisheries management then it seems sensible to invest in making it easier for people constructing the models to quickly access and compile the data necessary. Fishbase (Froese and Pauly, 2000) is a great source of biological data although further data mining is still necessary. Fishery specific data is held in more disparate databases. For assessed species this data is compiled for the assessment but for others the various discard and landings databases need to be consulted. Furthermore, for non-assessed species derivation of survey indices needs to be carried out. If this data was held centrally and openly accessible, with relevant indices calculated this job could be quicker and calculations would not have to be repeated.

8.8 References

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8.9 Figures

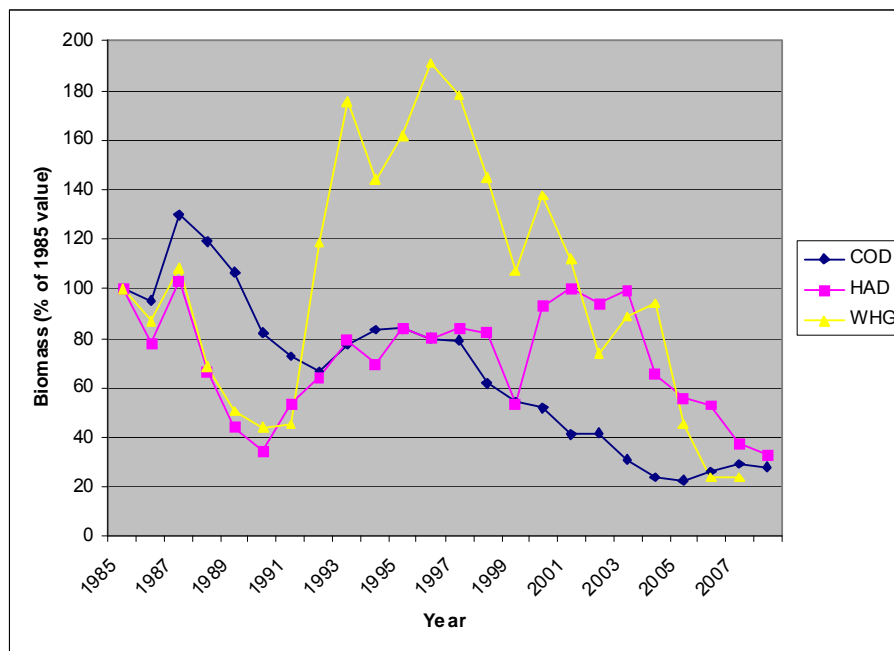


Figure 8.1: Total stock biomass for cod (COD), haddock (HAD) and whiting (WHG) relative to the 1985 value. Values are taken from WGCSE (2009) and WGN SDS (2007).

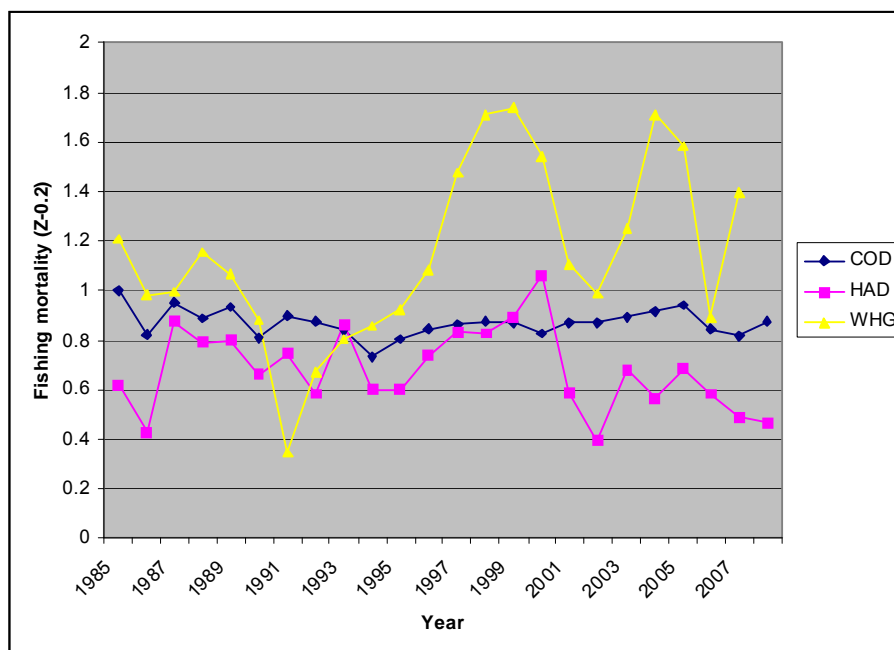


Figure 8.2: Instantaneous fishing mortality as estimated in the stock assessments for cod (COD), haddock (HAD) and whiting (WHG). This is the total mortality minus a fixed natural mortality of 0.2.

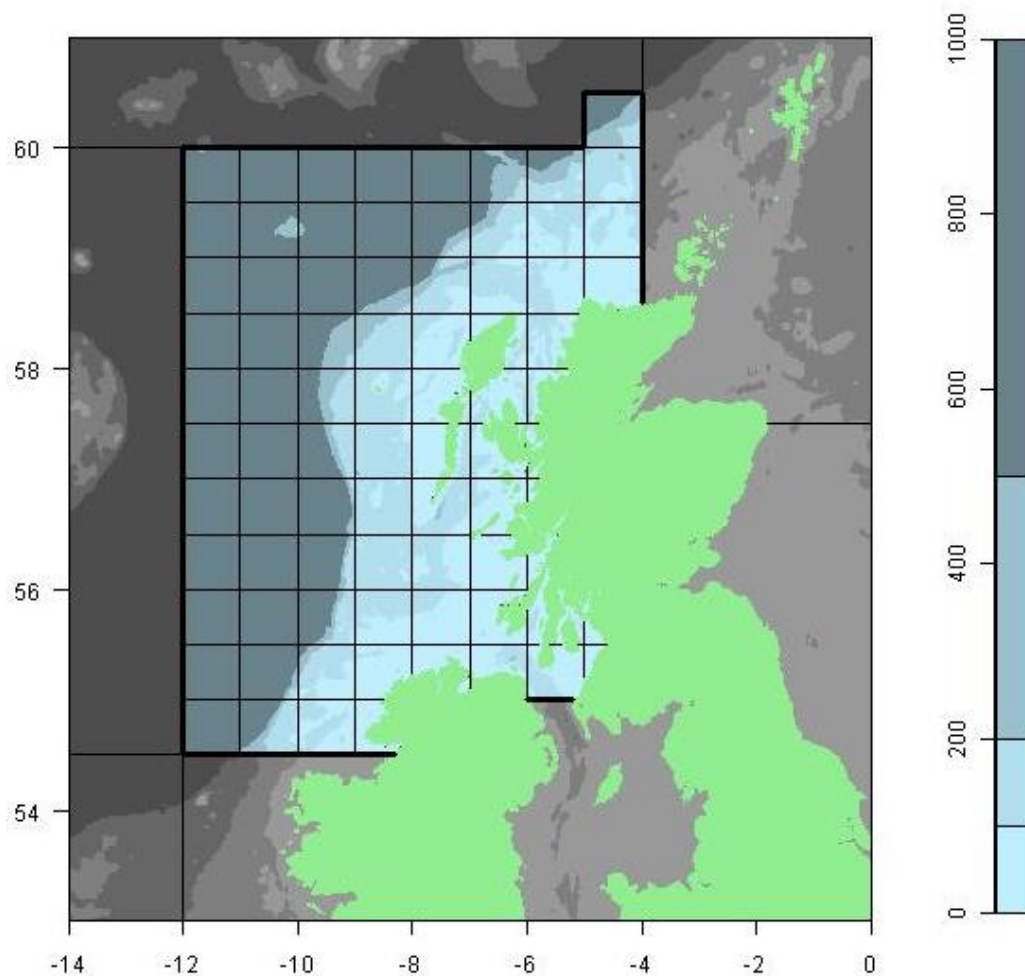


Figure 8.3: ICES area VIa with the shelf edge showing. EwE model represents the shelf area within VIa.

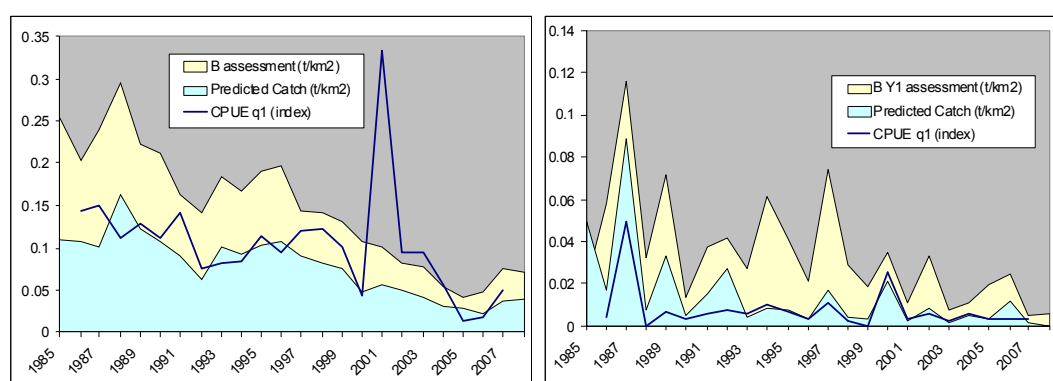


Figure 8.4: Fitting time series for mature (left) and juvenile (right) cod. Biomass and catch are stock assessment output. CPUE from Q1 Sco.VIa.IBTS gives an index of relative abundance.

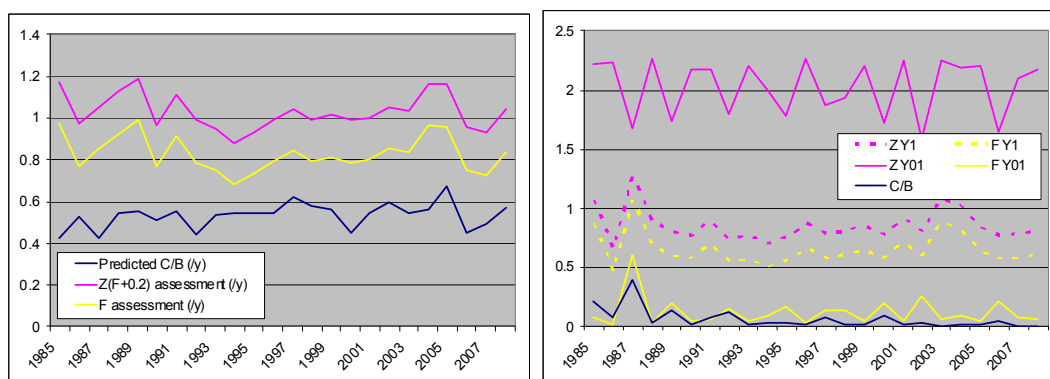


Figure 8.5: Driving data for juvenile (right) and adult (left) cod. Stock assessment values for F and Z are presented as well as time series of catch/biomass.

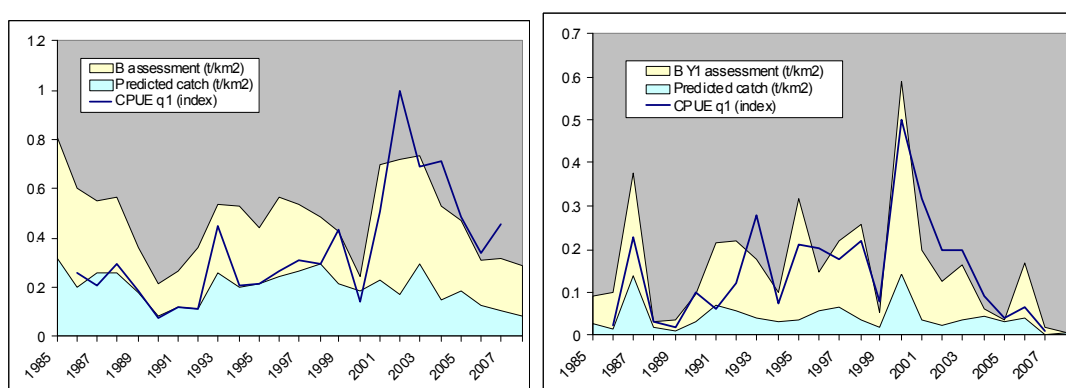


Figure 8.6: Fitting time series for mature and juvenile haddock. Biomass and catch are stock assessment output. CPUE from Q1 Sco.Vla.IBTS gives an index of relative abundance.

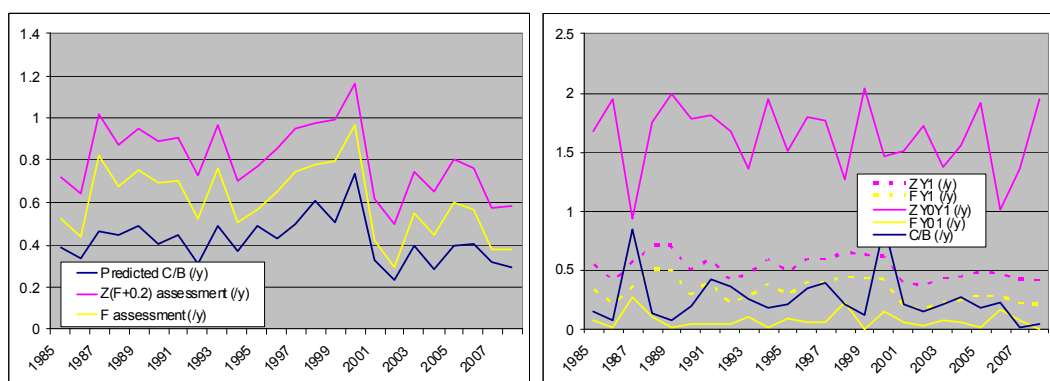


Figure 8.7: Driving data for juvenile and adult haddock. Stock assessment values for F and Z are presented as well as time series of catch/biomass for the adult stanza.

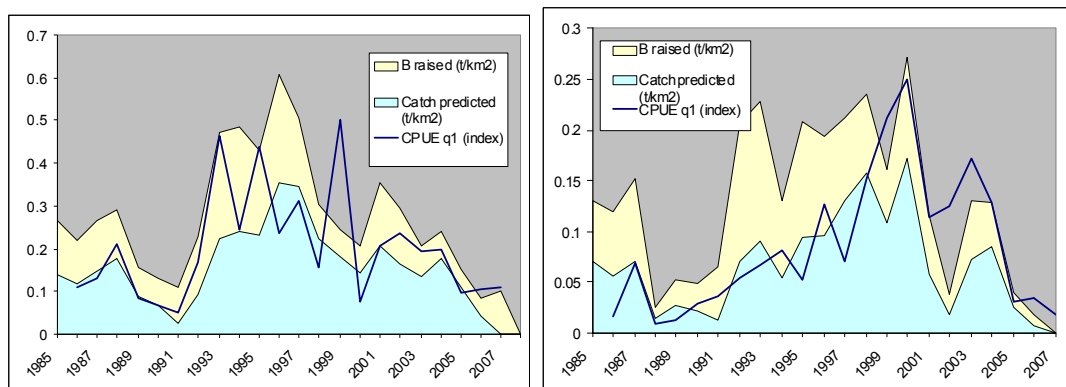


Figure 8.8: Fitting time series for mature and juvenile whiting. Biomass and catch are stock assessment output. The biomass is survey based stock assessment output (2007) rescaled to abundance estimates from a earlier full assessment (2004). The catch data is from the full assessment. CPUE from Q1 Sco.Vla.IBTS gives an index of relative abundance.

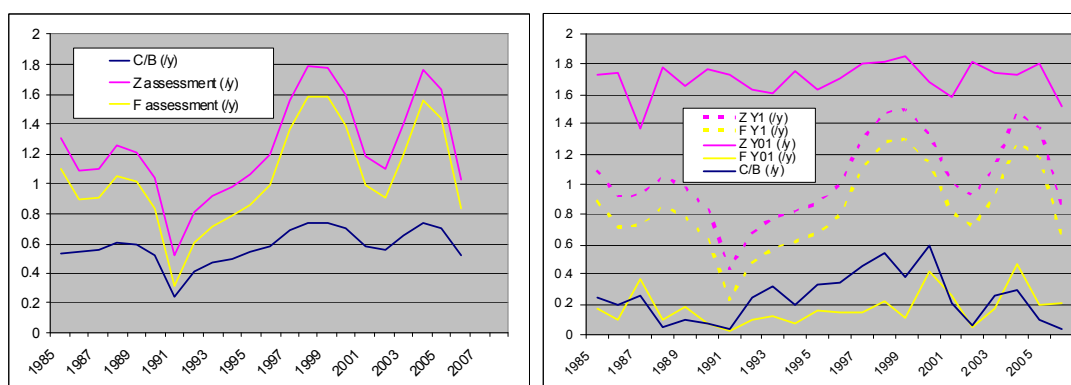


Figure 8.9: Driving data for juvenile and adult whiting. Stock assessment (WGNSSD, 2007) values for F and Z are presented as well as time series of catch/biomass for the adult stanza.Saithe/Pollock.

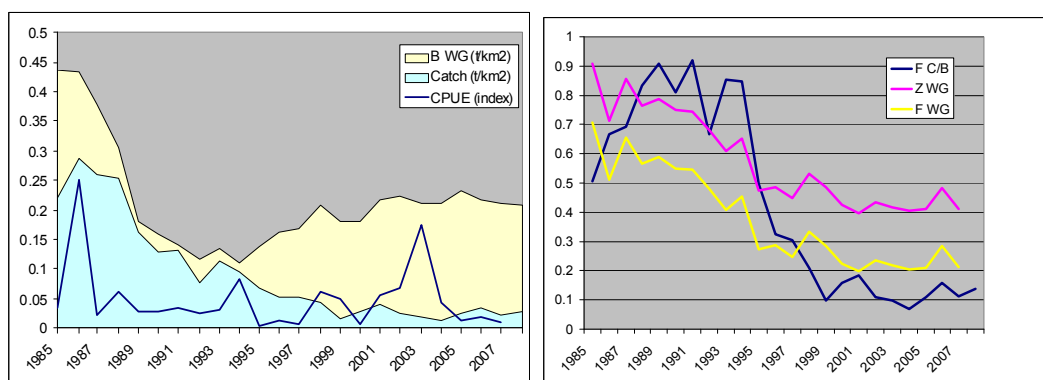


Figure 8.10: Fitting and driving data for the Saithe/Pollock group. Biomass data derived from the stock assessment, catch data from STATLANT and a CPUE trend from the IBTS are shown in the left hand frame. Mortality (F and Z) time-series from the stock assessment are shown as well as a time series of catch/biomass are shown in the second.

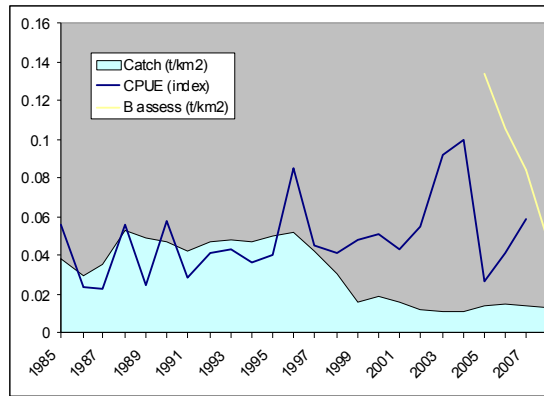


Figure 8.11: Fitting time-series of catch, CPUE and biomass from a stock assessment (2005 present) for the monkfish functional group

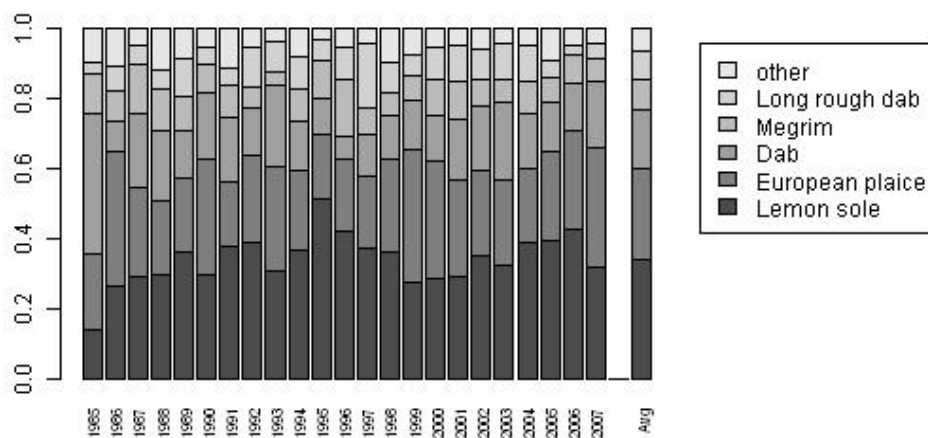


Figure 8.12: CPUE ratios of constituent species of the functional group Flatfish.

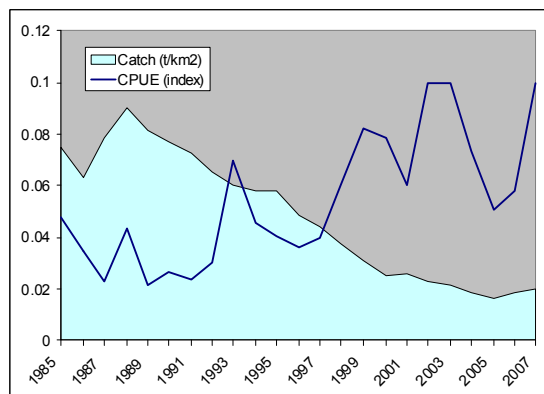


Figure 8.13 Catch data and CPUE index for the Flatfish functional group Rays and skates.

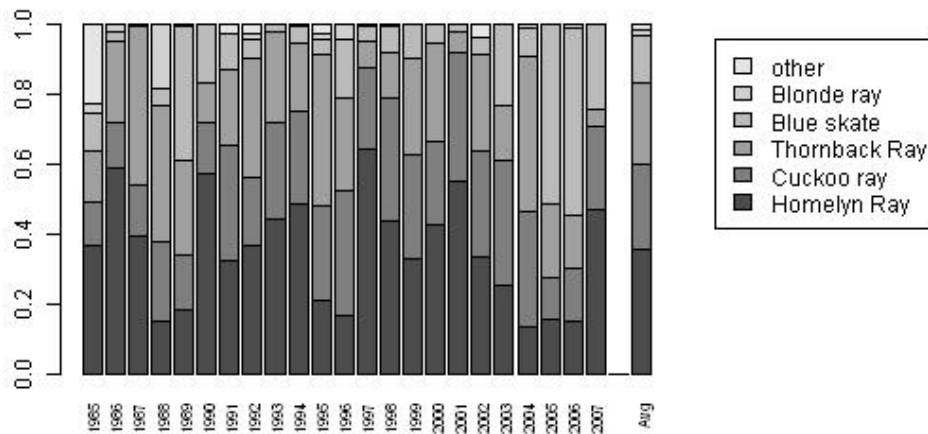


Figure 8.14: Composition of Rays and Skates Group based on proportion of CPUE (Q1 Sco.Vla.IBTS) for each constituent species.

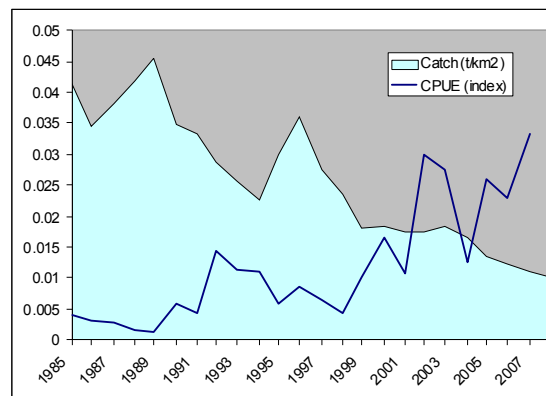


Figure 8.15: Catch data and CPUE index for the Rays and Skates functional group.

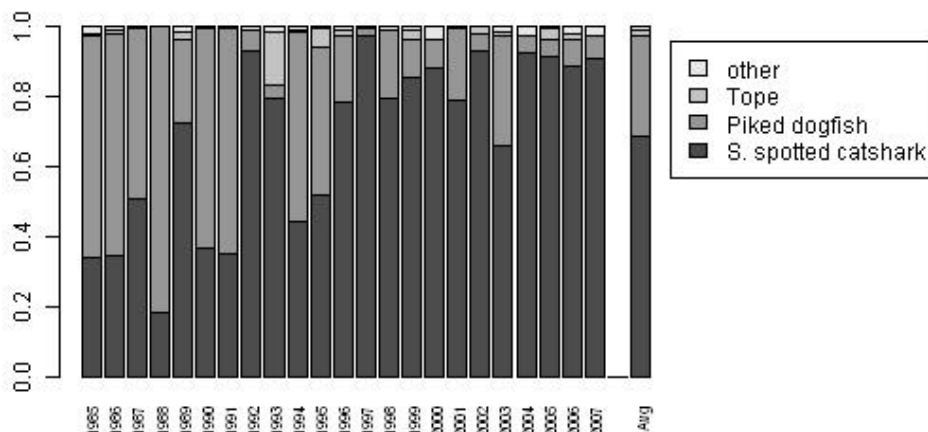


Figure 8.16: Proportions of CPUE from the Q1 Sco.Vla.IBTS for the Shark functional group.

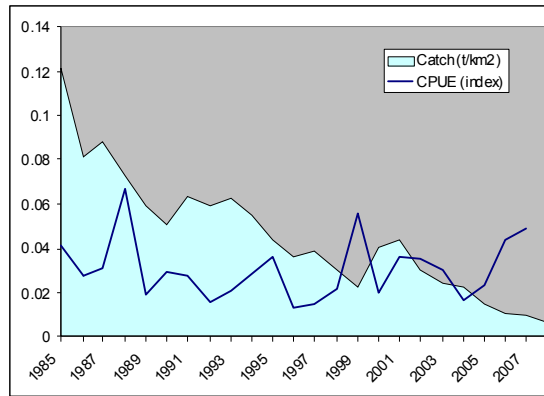


Figure 8.17: Catch data and CPUE index for the Sharks functional group.

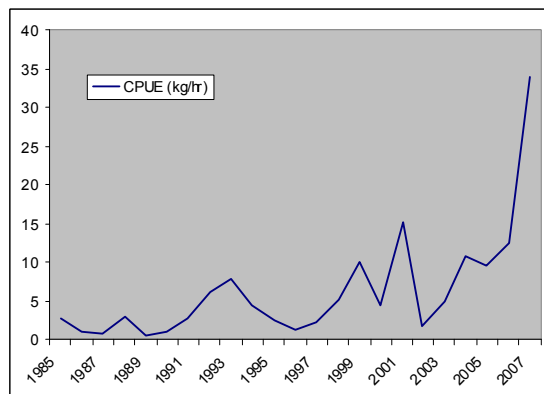


Figure 8.1: CPUE from Q1 Sco.Vla.IBTS trend for Poor cod.

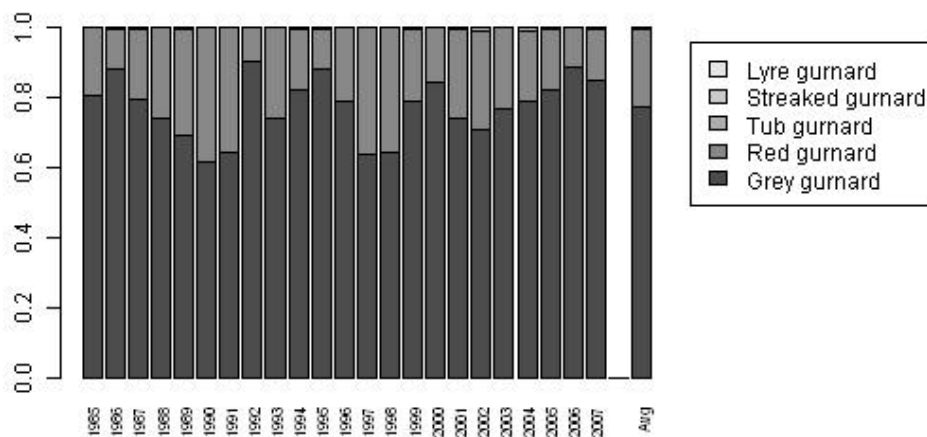


Figure 8.19: Ratios of CPUE (Q1 Sco.Vla.IBTS) for each of the species of gurnard found on the West Coast.

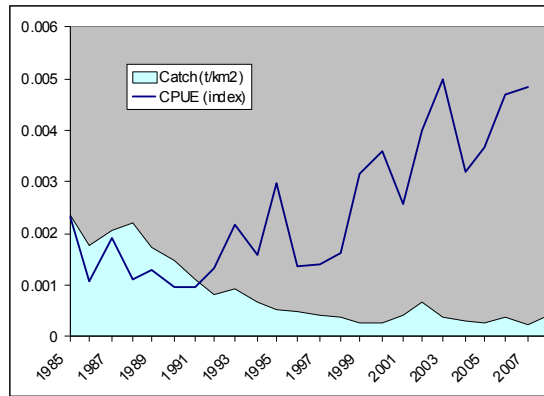


Figure 8.20: Catch data (STATLANT) and CPUE from Q1 Sco.Via.IBTS trend for the Gurnard functional group.

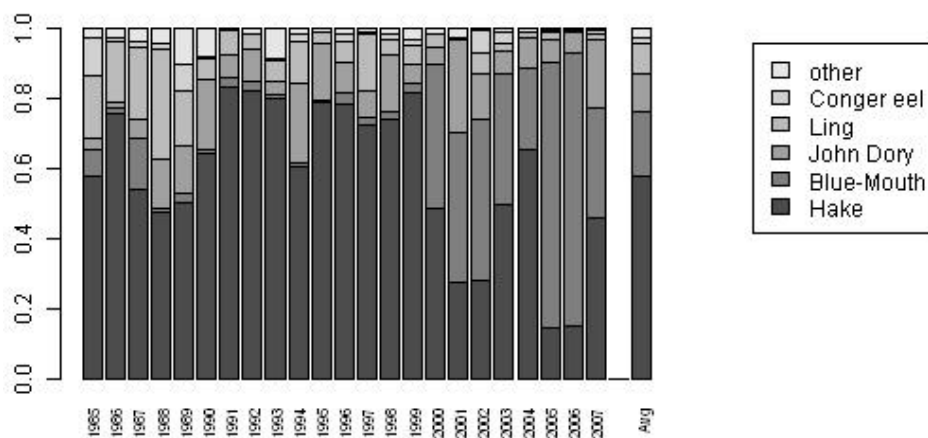


Figure 8.21: Proportions of CPUE from Q1 Sco.Via.IBTS for Other Demersal group.

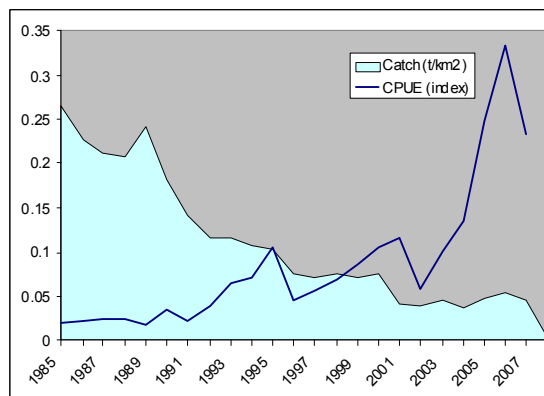


Figure 8.22: Catch data (STATLANT) and CPUE from Q1 Sco.Via.IBTS trend for the Other Demersal functional group.

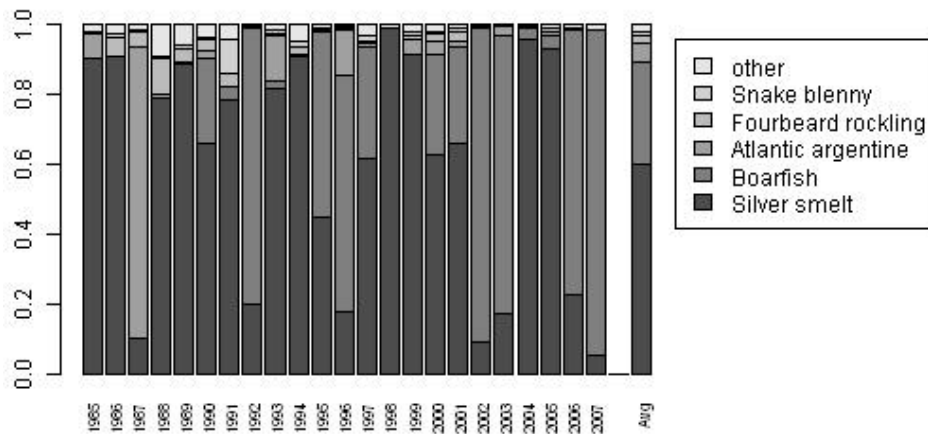


Figure 8.23: Proportions of CPUE from Q1 Sco.Vla.IBTS for Other Demersal group.

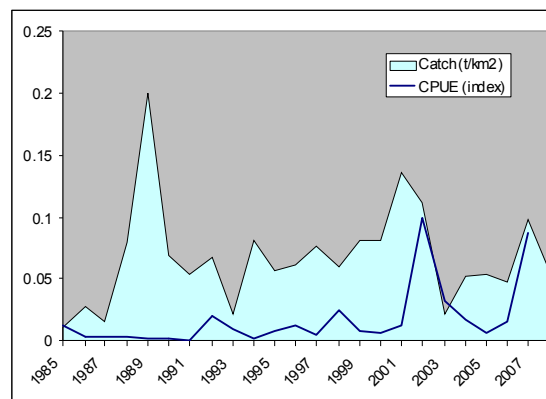


Figure 8.24: Catch and CPUE (Q1 Sco.Vla.IBTS) data for the group: Other benthopelagic fish.

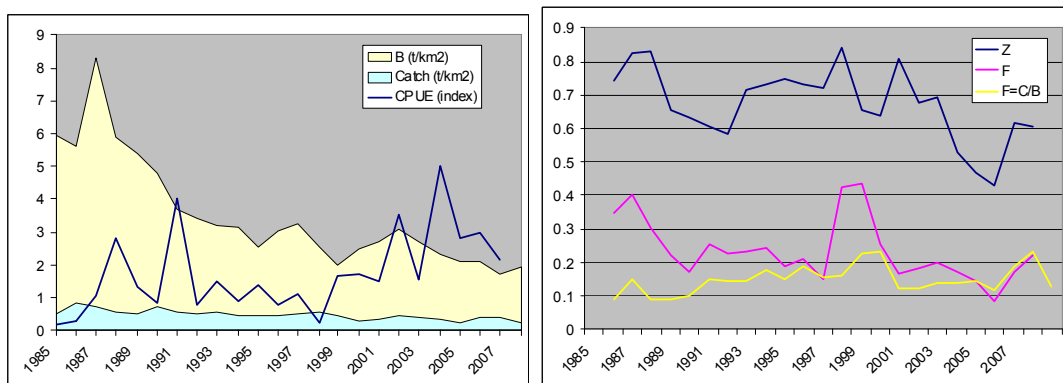


Figure 8.25: Stock assessment biomass and estimated catch are presented for the herring group. Catch/biomass, Z and F are time series available to drive the model. A choice of time series is available to drive the model. As with non-assessed species, the catch/biomass is calculated. Also shown are the VlaN assessment time-series of Z and an average over age-classes value for F.

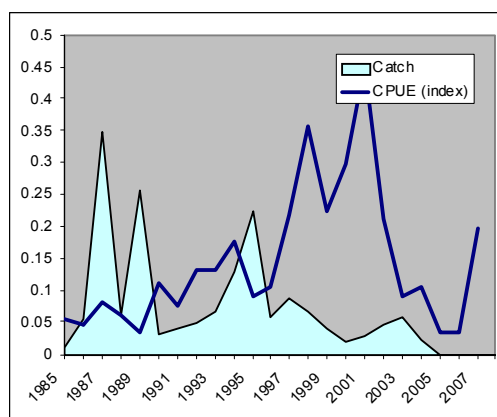


Figure 8.26: Reported catch and CPUE from the Q1 Sco.Vla.IBTS for Norway pout.

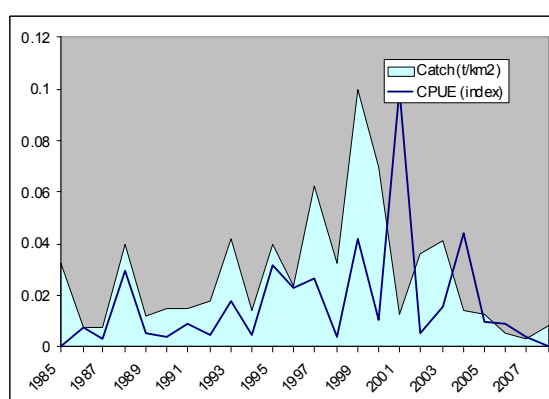


Figure 8.27 Catch data and Q1 Sco.Vla.IBTS CPUE series for sprat.

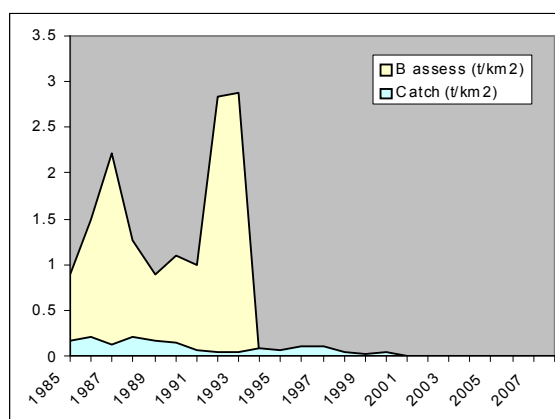


Figure 8.28: Biomass series up to 1996 from discontinued stock assessment and catch data from STATLANT for sandeels.

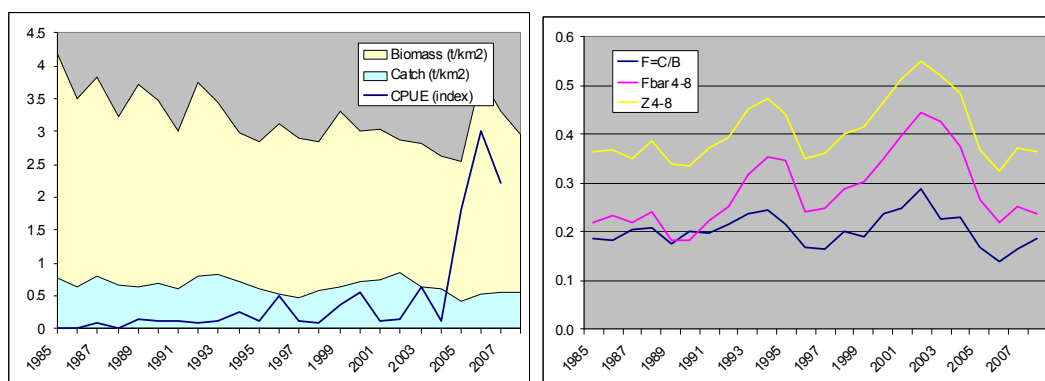


Figure 8.29: Stock assessment biomass, catch and CPUE for Mackerel are shown in the left hand frame. Catch/biomass, F and Z are in the right hand frame.

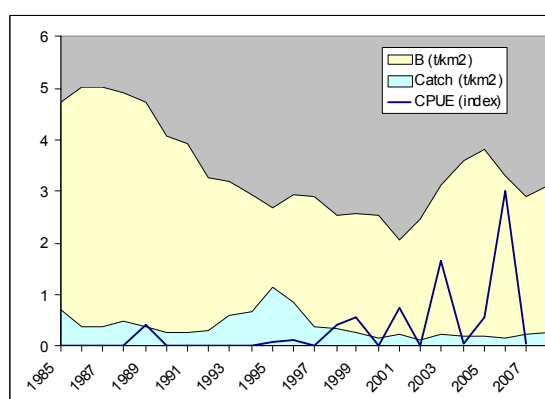


Figure 8.30: Biomass, catch and CPUE time series for Horse Mackerel.

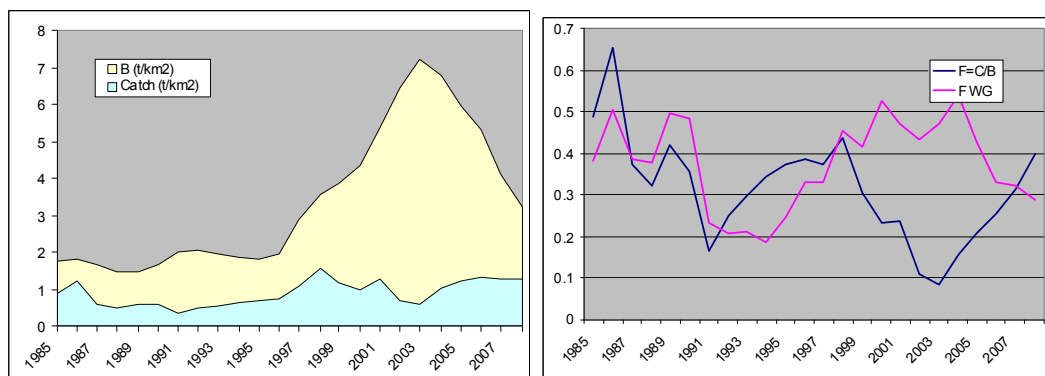


Figure 8.31: Biomass from the Western stock assessment for Blue Whiting scaled to Vla shelf along with catch from STATLANT is shown in the left-hand frame. F from the stock assessment and catch/biomass are in the right-hand frame.

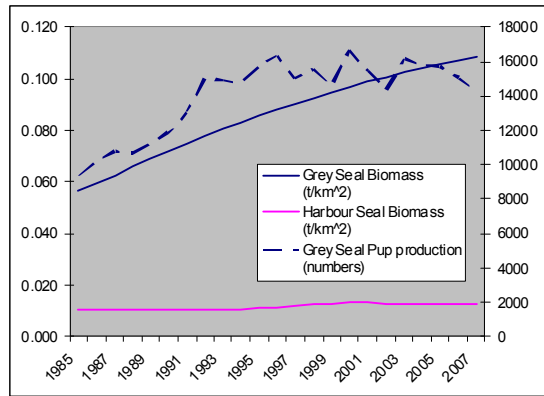


Figure 8.32: Grey seal and harbour seal trends in biomass.

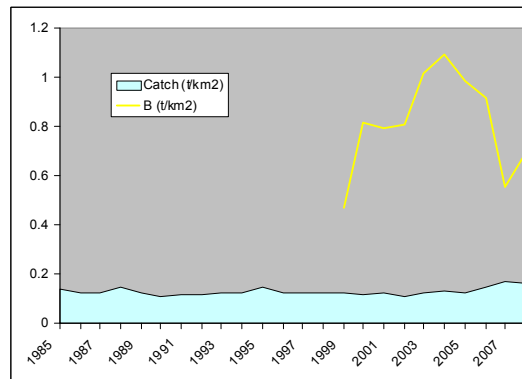


Figure 8.33: Assessed biomass summed over Functional Units and total catch for Nephrops.

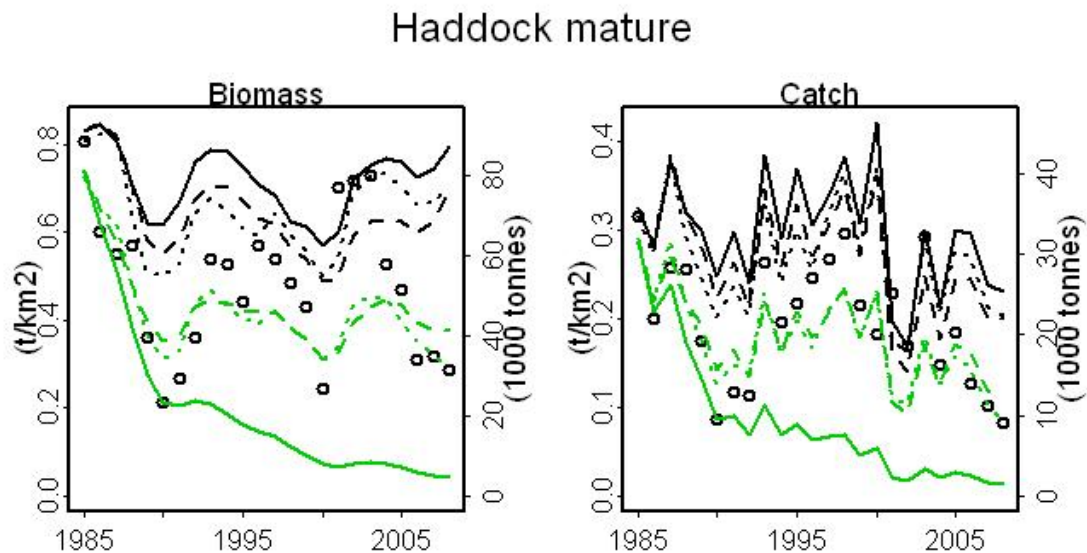


Figure 8.34: Biomass and catch data (black circles) and the model output (black and green lines) for mature haddock. The black lines represent output of the default model. The green lines represent the output of a model with biomass accumulation terms included. Solid lines: unfitted models; dashed lines: vulnerability fitted models; dotted lines: vulnerability and anomaly fitted models.

Haddock mature

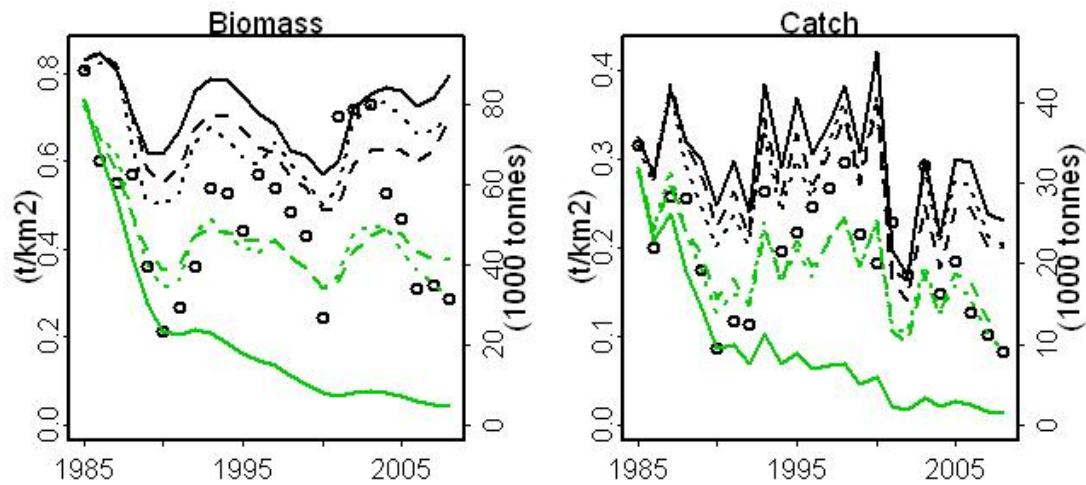


Figure 8.35: Data and model output for haddock. The green lines are as in Figure 8.34. The black lines show the model output with forced egg production. Solid lines: unfitted models; dashed lines: vulnerability fitted models; dotted lines: vulnerability and anomaly fitted models.

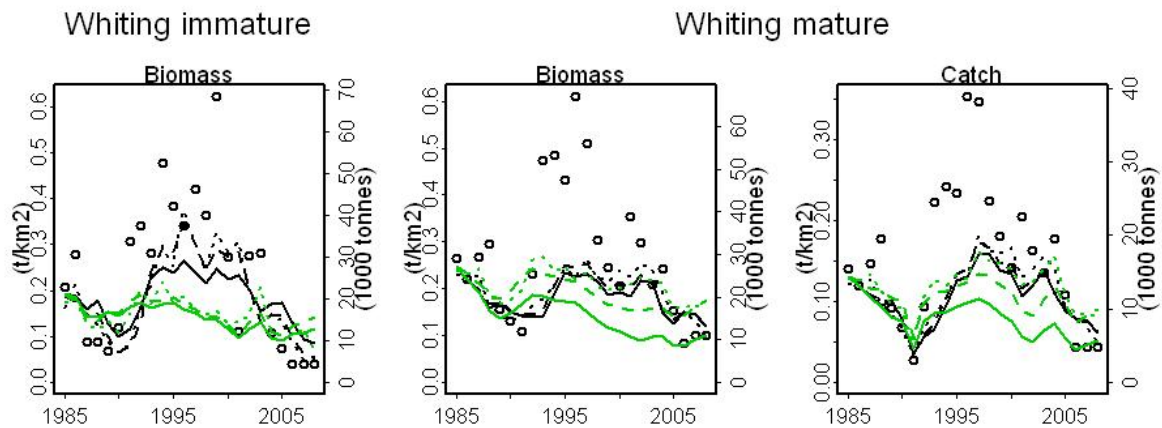


Figure 8.36: Data and model output for whiting. The green lines are from the model set up with negative biomass accumulation terms; the black lines show the output from the same model but with forced egg production. Solid lines: unfitted models; dashed lines: vulnerability fitted models; dotted lines: vulnerability and anomaly fitted models.

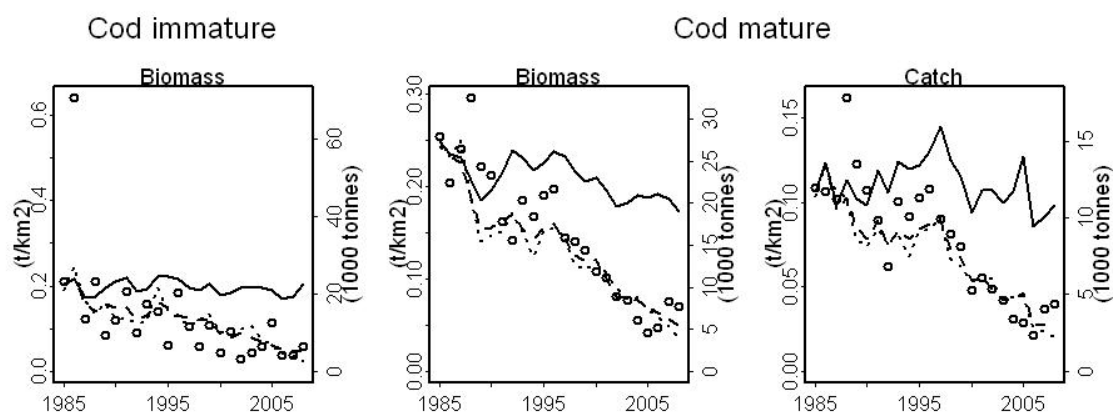


Figure 8.37: Biomass and catch data (black circles) and the model output (lines) for cod. The lines represent output of the model with biomass accumulation terms included. Solid lines: unfitted models; dashed lines: vulnerability fitted models; dotted lines: vulnerability and anomaly fitted models.

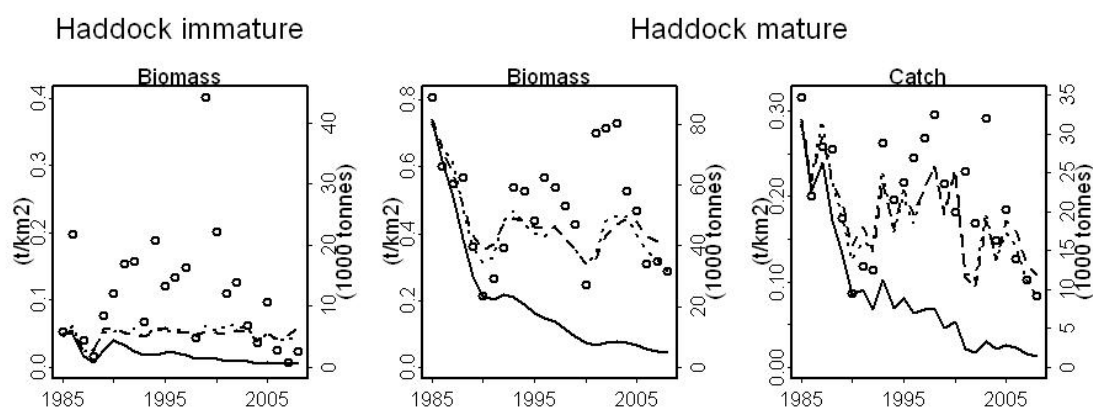


Figure 8.38: Biomass and catch data (black circles) and the model output (lines) for haddock. The lines represent output of the model with biomass accumulation terms included. Solid lines: unfitted models; dashed lines: vulnerability fitted models; dotted lines: vulnerability and anomaly fitted models.

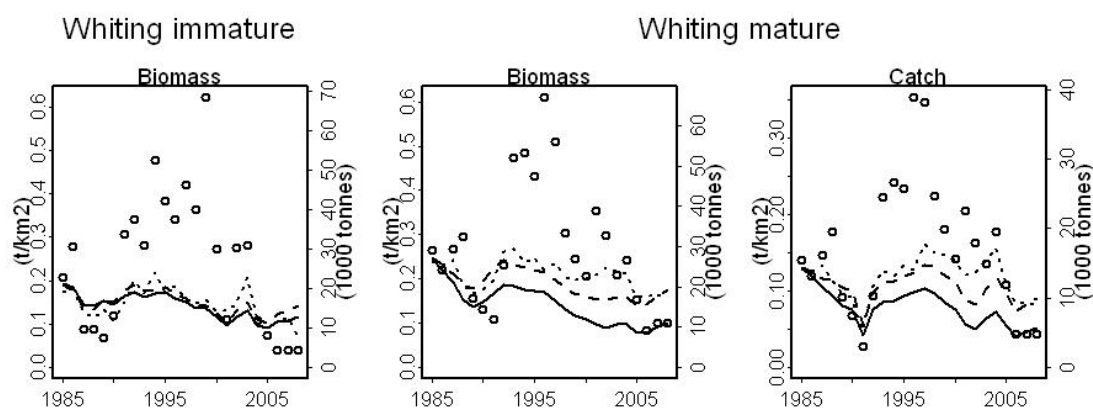


Figure 8.39: Biomass and catch data (black circles) and the model output (lines) for whiting. The lines represent output of the model with biomass accumulation terms included. Solid lines: unfitted models; dashed lines: vulnerability fitted models; dotted lines: vulnerability and anomaly fitted models.

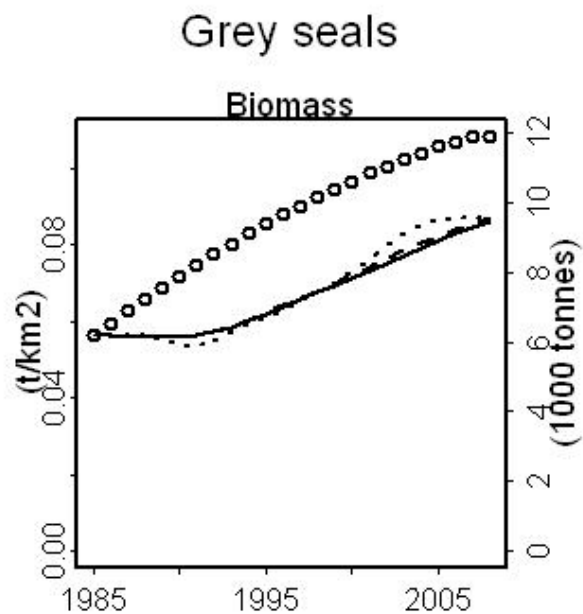


Figure 8.40: Grey seal biomass data (circles) and model output (lines). The solid line is the unfitted model output, the dashed line has fitted vulnerability parameters and the dotted line has both vulnerability and primary production fitted.

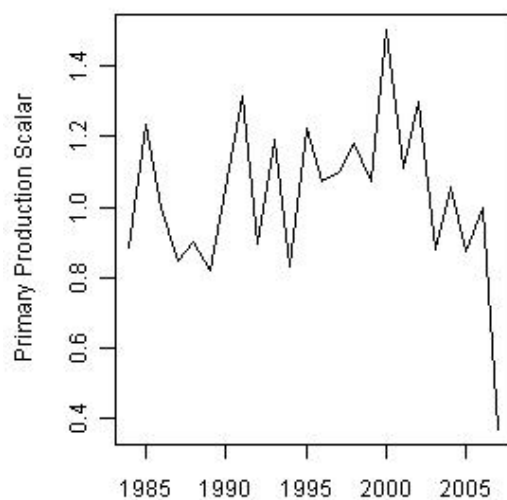


Figure 8.41: Fitted forcing-function for primary production.

Cod mature

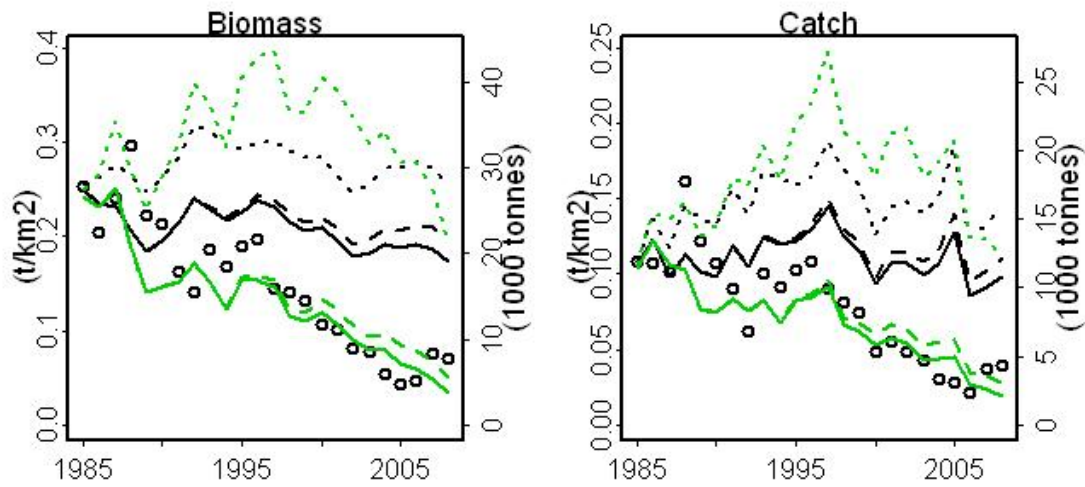


Figure 8.42: The result of applying a mortality to the grey seal population on cod. Dotted lines represent the result of high mortality on seals to remove them from the ecosystem, dashed lines represent the mortality to obtain a constant grey seal biomass and solid lines represent no mortality. Black represents the output of an unfitted model and green the fitted model.

Cod mature

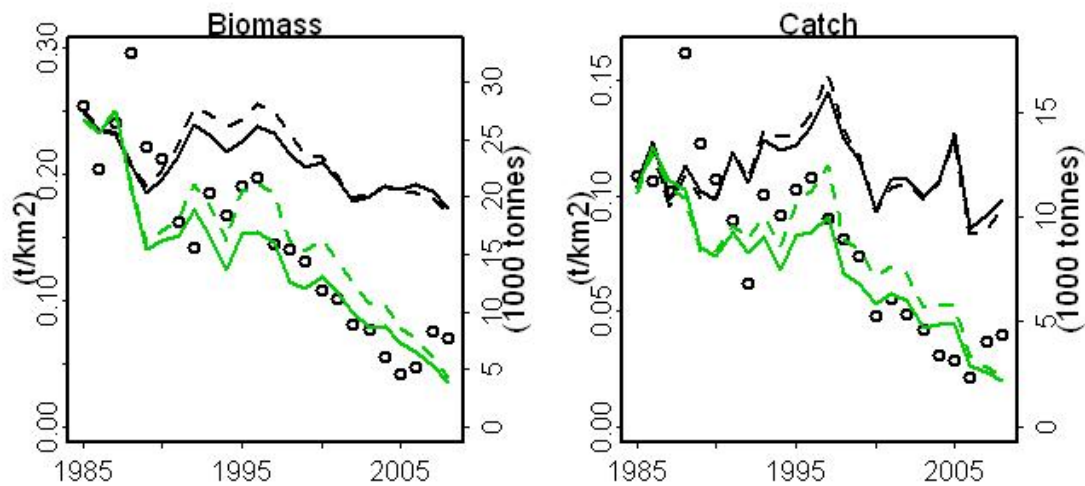


Figure 8.43: The results for cod of applying a clean Nephrops trawl fishery to all fish species. Dashed lines represent the clean Nephrops trawl. Black represents the output of an unfitted model and green the fitted model.

Haddock mature

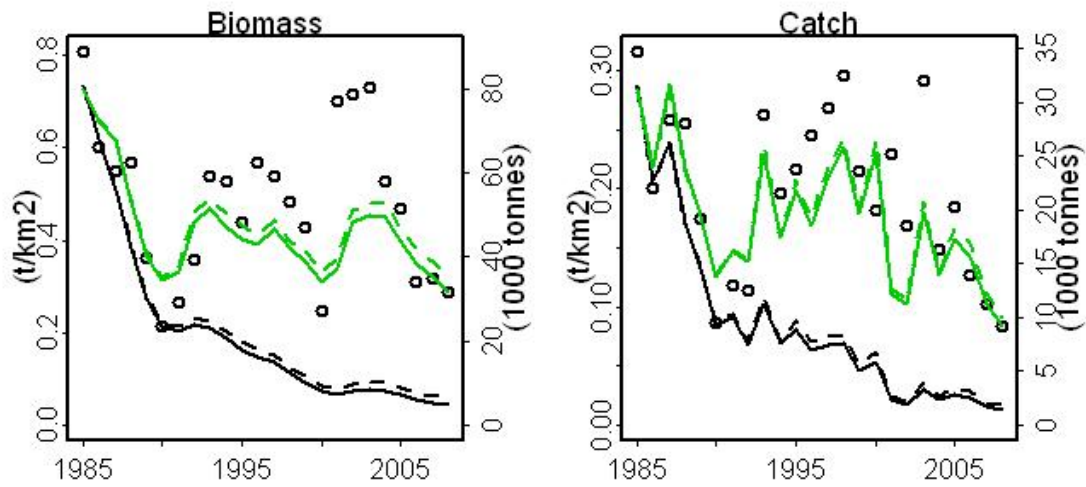


Figure 8.44: The results for haddock of applying a clean Nephrops trawl fishery to all fish species. Dashed lines represent the clean Nephrops trawl. Black represents the output of an unfitted model and green the fitted model.

Whiting mature

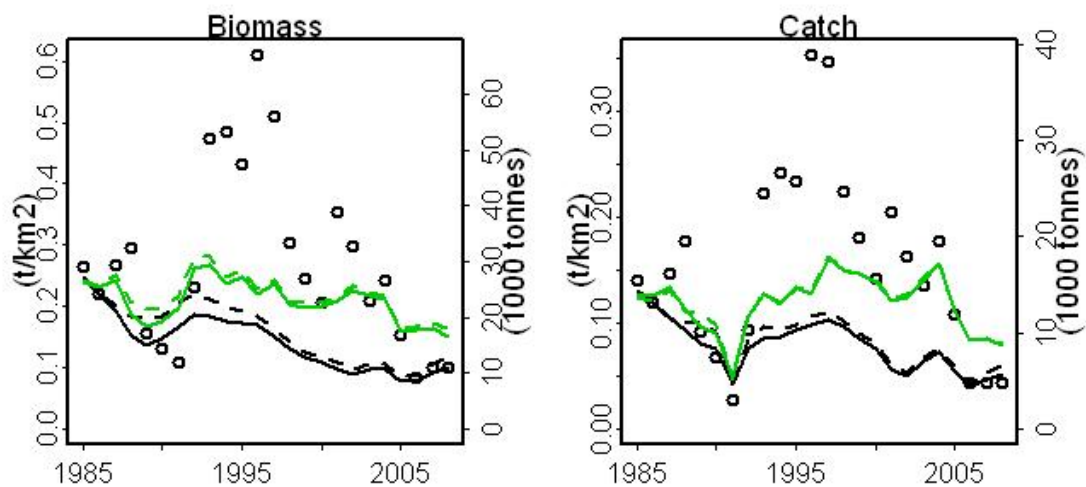


Figure 8.45: The results for whiting of applying a clean Nephrops trawl fishery to all fish species. Dashed lines represent the clean Nephrops trawl. Black represents the output of an unfitted model and green the fitted model.

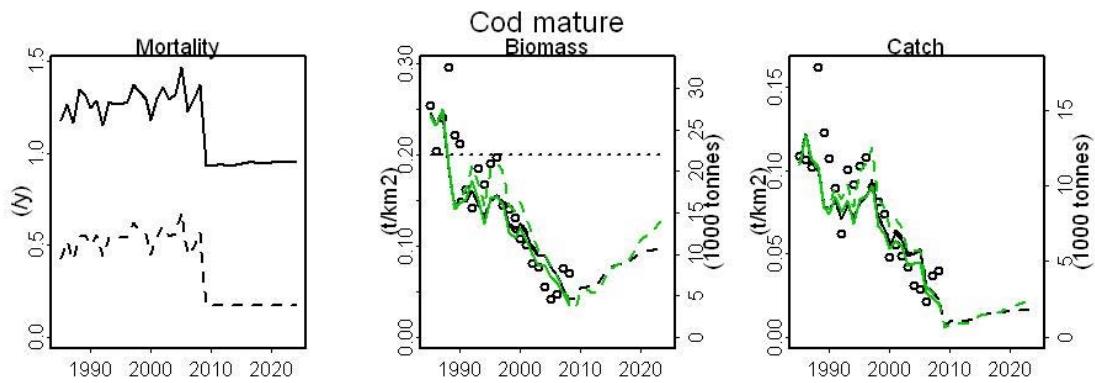


Figure 8.46: Forecasts for cod resulting from enforcing F_{MSY} on the three gadoid stocks. Black represents the output of a model initially in equilibrium (i.e. with Biomass Accumulation=0). Green shows the output for a model with biomass of the three gadoid species initially in decline. Total mortality (solid line) and fishing mortality (dashed line) are shown in the first frame. The dotted line in the biomass frame is B_{pa}

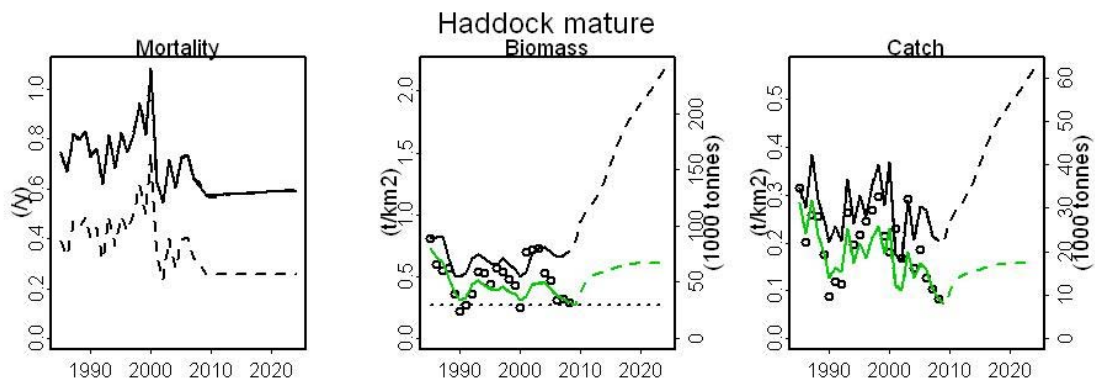


Figure 8.47: Forecasts for haddock resulting from enforcing F_{MSY} on the three gadoid stocks. Black represents the output of a model initially in equilibrium (i.e. with Biomass Accumulation=0). Green shows the output for a model with biomass of the three gadoid species initially in decline. Total mortality (solid line) and fishing mortality (dashed line) are shown in the first frame. The dotted line in the biomass frame is B_{pa}

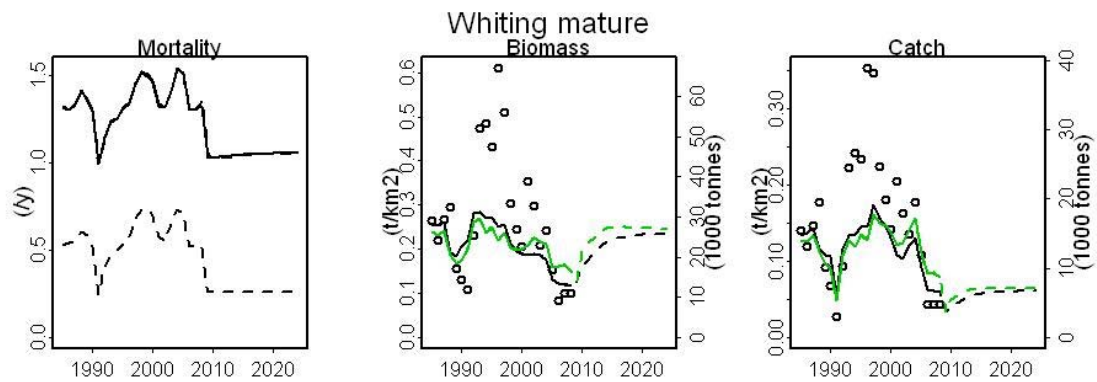


Figure 8.48: Forecasts for whiting resulting from enforcing F_{MSY} on the three gadoid stocks. Black represents the output of a model initially in equilibrium (i.e. with Biomass Accumulation=0). Green shows the output for a model with biomass of the three gadoid species initially in decline. Total mortality (solid line) and fishing mortality (dashed line) are shown in the first frame.

9 Appendix 1 – Ecosystem Model - Fish Functional Group Definitions

9.1 Fish Functional Groups

The table below includes all the fish functional groups in the model. All species that have been caught in the West Coast of Scotland Quarter 1 IBTS at depths less than 200m are allocated to model functional groups except for hatchet fish and three species of grenadier which are deep sea species and are rarely found on the shelf and hence are excluded from the model.

Species names from the ICES landings database STATLANT are matched to the IBTS names where possible. Many of the names within STATLANT are for species groups as opposed to individual species. Such a name is assigned to a model functional group if any of the species which it describes falls into that functional group. This is straightforward for a name such as “Flatfishes nei” which is assigned to model functional group Flatfish. For names such as “Marines fishes nei” a grouping is less obvious.

Species that are found in STATLANT but that are not found in the IBTS are generally deep sea species and are usually excluded. These species are listed after the table.

EWE Group	IBTS Name	Species	STATLANT Name
Anglerfish	Anglerfish	<i>Lophius piscatorius</i>	Angler(=Monk)
	Black bellied angler	<i>Lophius budegassa</i>	
Cod	Cod	<i>Gadus morhua</i>	Anglerfishes nei
			Monkfishes nei
Flatfish	Atlantic halibut	<i>Hypoglossal hippoglossus</i>	Atlantic cod
	Atlantic sole	<i>Peruse lascaris</i>	Atlantic halibut
	Brill	<i>Scophthalmus rhombus</i>	Sand sole
	Common European sole	<i>Solea vulgaris</i>	Brill
	Common topknot	<i>Zeugopterus punctatus</i>	Common sole
	Dab	<i>Limanda limanda</i>	Common dab
	European flounder	<i>Platichthys flesus</i>	European flounder
	European plaice	<i>Pleuronectes platessa</i>	European plaice
	Fourspot megrim	<i>Lepidorhombus boscii</i>	Four-spot megrim
	Grohmman's scaldfish	<i>Arnoglossus thori</i>	
	Imperial scaldfish	<i>Arnoglossus imperialis</i>	
	Lemon sole	<i>Microstomus kitt</i>	Lemon sole
	Long rough dab	<i>Hippoglossoides platessoides</i>	Amer. plaice(=Long rough dab)
	Megrim	<i>Lepidorhombus whiffiagonis</i>	Megrim
	Norwegian topknot	<i>Zeugopterus norvegicus</i>	
	Scaldfish	<i>Arnoglossus laterna</i>	
	Solenette	<i>Buglossidium luteum</i>	
	Thickback sole	<i>Microchirus variegatus</i>	
	Turbot	<i>Psetta maxima</i>	Turbot
	Witch flounder	<i>Glyptocephalus cynoglossus</i>	Witch flounder
			Solea spp
			Megrims nei
			Flatfishes nei
Gurnard	Grey gurnard	<i>Eutrigla gurnardus</i>	Grey gurnard
	Lyre gurnard	<i>Trigla lyra</i>	Piper gurnard
	Red gurnard	<i>Aspitrigla cuculus</i>	Red gurnard
	Streaked gurnard	<i>Trigloporus lastoviza</i>	
	Tub gurnard	<i>Trigla lucerna</i>	Tub gurnard
			Atlantic searobins
			Gurnards nei
			Gurnards, searobins nei
Haddock	Haddock	<i>Melanogrammus</i>	Haddock

Herring	Herring	<i>aeglefinus</i>	Atlantic herring
Horse mackerel	Horse mackerel	<i>Clupea harengus</i>	Atlantic horse mackerel
		<i>Trachurus trachurus</i>	Jack and horse mackerels nei
Other demersal	Blue-Mouth	<i>Helicolenus dactylopterus</i>	
	Brown trout	<i>Salmo trutta</i>	Sea trout
			Trouts nei
	Catfish	<i>Anarhichas lupus</i>	Salmonoids nei
			Atlantic wolffish
	Common eel	<i>Anguilla anguilla</i>	Wolffishes(=Catfishes) nei
	Conger eel	<i>Conger conger</i>	European eel
	Cusk	<i>Brosme brosme</i>	European conger
	Greater Forkbeard	<i>Phycis blennoides</i>	Tusk(=Cusk)
			Greater forkbeard
	Hake	<i>Merluccius merluccius</i>	Forkbeard
	John Dory	<i>Zeus faber</i>	European hake
			John dory
	Ling	<i>Molva molva</i>	Dories nei
			Ling
			Lings nei
			Gadiformes nei
	Lumpsucker	<i>Cyclopterus lumpus</i>	
	Norway haddock	<i>Sebastes viviparus</i>	Golden redfish
	Ocean perch	<i>Sebastes marinus</i>	Redfish
			Atlantic redfishes nei
	Red bandfish	<i>Cepola rubescens</i>	
	Red sea bream	<i>Pagellus bogaraveo</i>	Blackspot(=red) seabream
			Porgies, seabreams nei
Mackerel	Atlantic mackerel	<i>Scomber scombrus</i>	Atlantic mackerel
			Mackerels nei
Norway pout	Norway pout	<i>Trisopterus esmarkii</i>	Norway pout
Other pelagic	Alice shad	<i>Alosa alosa</i>	
	European anchovy	<i>Engraulis encrasicolus</i>	European anchovy
	Pearlside	<i>Maurolicus muelleri</i>	
	True sardine	<i>Sardina pilchardus</i>	European pilchard(=Sardine)
	Twaite shad	<i>Alosa fallax</i>	
			Capelin
			Clupeoids nei
			European smelt
			Hairtails, cutlassfishes nei
			Lanternfishes nei
			Largehead hairtail
			Pelagic fishes nei
			Pelagic percomorphs nei
			Silversides(=Sand smelts) nei
Poor cod	Poor Cod	<i>Trisopterus minutus</i>	
Pollock / Saithe	Pollock	<i>Pollachius pollachius</i>	Pollock
	Saithe	<i>Pollachius virens</i>	Saithe(=Pollock)
Sandeel	Corbin's sand eel	<i>Hyperoplus immaculatus</i>	
	Greater sand eel	<i>Hyperoplus lanceolatus</i>	
	Sand eel	<i>Ammodytes marinus</i>	
	Smoothed sandeel	<i>Gymnammodytes semisquamatus</i>	
			Sandeels(=Sandlances) nei
Other benthopelagic	Alaskan Stickleback	<i>Gasterosteus aculeatus</i>	
	Atlantic argentine	<i>Argentina silus</i>	Greater argentine
			Argentines
	Atlantic warbonnet	<i>Chirolophis ascanii</i>	
	Ballan wrasse	<i>Labrus bergylta</i>	
	Bib	<i>Trisopterus luscus</i>	Pouting(=Bib)
	Boarfish	<i>Capros aper</i>	Boarfish
			Boarfishes nei
	Bridled triggerfish	<i>Balistes carolinensis</i>	
	Bull rout	<i>Myoxocephalus scorpius</i>	
	Butterfish	<i>Pholis gunnellus</i>	
	Common dragonet	<i>Callionymus lyra</i>	
	Common goby	<i>Pomatoschistus microps</i>	
	Crystal goby	<i>Crystallogobius linearis</i>	
	Cuckoo wrasse	<i>Labrus mixtus</i>	Wrasses, hogfishes, etc. nei
	Echiodon drummondii	<i>Echiodon drummondii</i>	
	Fivebearded rockling	<i>Ciliata mustella</i>	
	Fourbeard rockling	<i>Enchelyopus cimbrius</i>	

	Freckled goby	<i>Pomatoschistus minutus</i>	Rocklings nei
	Fries' goby	<i>Lesueurigobius friesii</i>	
	Goldsinny wrasse	<i>Ctenolabrus rupestris</i>	
	Great Pipefish	<i>Syngnathus acus</i>	
	Hook-nose	<i>Agonus cataphractus</i>	
	Lesse forkbeard	<i>Raniceps raninus</i>	
	Lesser weever	<i>Trachinus vipera</i>	Weeverfishes nei
			Greater weever
	Longspine snipefish	<i>Macroramphosus scolopax</i>	
	Longspined sea scorpion	<i>Taurulus bubalis</i>	
	Moustache sculpin	<i>Triglops murrayi</i>	
	Northern rockling	<i>Ciliata septentrionalis</i>	
	Norway bullhead	<i>Taurulus lilljeborgi</i>	
	Ocean Pipefish	<i>Entelurus aequerius</i>	
	Red mullet	<i>Mullus surmuletus</i>	Red mullet
			Mullets nei
	Reticulated dragonet	<i>Callionymus reticulatus</i>	
	Silver smelt	<i>Argentina sphyraena</i>	Argentina
	Silvery pout	<i>Gadiculus argenteus</i>	
	Snake blenny	<i>Lumpenus</i>	
		<i>lumpretaeformis</i>	
	Spotted dragonet	<i>Callionymus maculatus</i>	
	Striped seasnail	<i>Liparis liparis</i>	
	Threebearded rockling	<i>Gaidropsarus vulgaris</i>	
	Three-bearded rocklings	<i>Gaidropsarus</i>	
	Tompot blenny	<i>Blennius gattorugine</i>	
	True gobies	<i>Gobiidae</i>	
	Viviporous blenny	<i>Zoarces viviparus</i>	
			Finfishes nei
			Groundfishes nei
			Marine fishes nei
			Scorpionfishes nei
			Scorpionfishes, rockfishes nei
Sharks	Blackmouth catshark	<i>Galeus melastomus</i>	
	Nursehound	<i>Scyliorhinus stellaris</i>	Nursehound
	Piked dogfish	<i>Squalus acanthias</i>	Picked dogfish
	Small-spotted catshark	<i>Scyliorhinus canicula</i>	Small-spotted catshark
	Smooth-hound	<i>Mustelus mustelus</i>	
	Starry smooth-hound	<i>Mustelus asterias</i>	
	Tope	<i>Galeorhinus galeus</i>	Tope shark
			Catsharks, etc. nei
			Catsharks, nursehounds nei
			Dogfish sharks nei
			Dogfishes and hounds nei
			Dogfishes nei
			Smooth-hounds nei
			Various sharks nei
			European sprat
Sprat	Sprat	<i>Sprattus sprattus</i>	
Rays and skates	Blonde ray	<i>Raja brachyura</i>	
	Blue skate	<i>Dipturus batis</i>	Blue skate
	Bottlenosed skate	<i>Rostroraja alba</i>	
	Cuckoo ray	<i>Leucoraja circularis</i>	Sandy ray
	Cuckoo ray	<i>Raja naevus</i>	Cuckoo ray
	Homelyn Ray	<i>Raja montagui</i>	Spotted ray
	Shagreen ray	<i>Leucoraja fullonica</i>	Shagreen ray
	Starry skate	<i>Amblyraja radiata</i>	Starry skate
	Thornback Ray	<i>Raja clavata</i>	Thornback ray
			Raja rays nei
			Rays and skates nei
			Rays, stingrays, mantas nei
Whiting	Blue whiting	<i>Micromesistius poutassou</i>	Blue whiting(=Poutassou)
Blue whiting	Whiting	<i>Merlangius merlangus</i>	Whiting

Species and species groups in STATLANT excluded from model:

Albacore, Alfonsino, Alfonsinos nei, Angelshark, Arctic skate, Atlantic pomfret, Atlantic salmon, Atlantic thornyhead, Axillary seabream, Baird's slickhead, Basking shark, Beaked redfish, Birdbeak dogfish, Black cardinal fish, Black dogfish, Black scabbardfish, Black scorpionfish, Black seabream, Blackbelly rosefish, Blackfin tuna, Blue ling, Blue shark, Bluntnose sixgill shark, Bogue, Carangids nei, Cardinalfishes, etc. nei, Cartilaginous fishes

nei, Common dentex, Common mora, Common stingray, Crest-tail catsharks nei, Demersal percomorphs nei, Dentex nei, Dusky grouper, Eelpout, European perch, European seabass, Gilthead seabream, Great lanternshark, Greenland halibut, Grenadiers nei, Groupers, seabasses nei, Gulper shark, Kitefin shark, Knifetooth dogfish, Lanternsharks nei, Large-eyed rabbitfish, Leafscale gulper shark, Longnose velvet dogfish, Longnosed skate, Moras nei, Northern bluefin tuna, Northern wolffish, Orange roughy, Porbeagle, Portuguese dogfish, Rabbit fish, Ratfishes nei, Red porgy, Red scorpionfish, Risso's smooth-head, Rough skate, Roughhead grenadier, Roundnose grenadier, Seabasses nei, Shortfin mako, Silver croaker, Silver scabbardfish, Slimeheads nei, Spectrunculus grandis, Splendid alfonso, Spotted wolffish, Straightnose rabbitfish, Sturgeon, Swordfish, Tuna-like fishes nei, Velvet belly, White hake, Winter flounder, Wreckfish

9.2 Invertebrate Functional Group Definitions

EwE Code	STATLANT Name	EwE Code	STATLANT Name
CEP	Broadtail shortfin squid	CRU	Pandalus shrimps nei
CEP	Cephalopods nei	CRU	Penaeus shrimps nei
CEP	Common cuttlefish	CRU	Portunus swimcrabs nei
CEP	Common octopus	CRU	Red crab
CEP	Common squids nei	CRU	Spinous spider crab
CEP	Cuttlefish,bobtail squids nei	CRU	Velvet swimcrab
CEP	European flying squid	EPF	Aquatic invertebrates nei
CEP	Inshore squids nei	EPF	Blue mussel
CEP	Northern shortfin squid	EPF	Clams, etc. nei
CEP	Octopuses nei	EPF	Common edible cockle
CEP	Octopuses, etc. nei	EPF	Common periwinkle
CEP	Squids nei	EPF	Cupped oysters nei
CEP	Various squids nei	EPF	European flat oyster
CRU	Blue crab	EPF	Flat oysters nei
CRU	Common prawn	EPF	Great Atlantic scallop
CRU	Common shrimp	EPF	Grooved carpet shell
CRU	Common spiny lobster	EPF	Marine molluscs nei
CRU	Craylets, squat lobsters	EPF	Periwinkles nei
CRU	Deep-sea red crab	EPF	Pullet carpet shell
CRU	Edible crab	EPF	Queen scallop
CRU	European lobster	EPF	Razor clams nei
CRU	Green crab	EPF	Razor clams, knife clams nei
CRU	King crabs	EPF	Sand gaper
CRU	Lobsters nei	EPF	Scallops nei
CRU	Marine crabs nei	EPF	Sea mussels nei
CRU	Marine crustaceans nei	EPF	Sea urchins, etc. nei
CRU	Natantian decapods nei	EPF	Surf clams nei
CRU	Northern prawn	EPF	Whelk
CRU	Palaemonid shrimps nei	NEP	Norway lobster
CRU	Palinurid spiny lobsters nei		

9.3 Summary of Ecopath Model Parameters

Ecopath Model: Balanced

	Group name	Trophic level	Biomass	Z	P/B	Q/B	EE	P/Q
			t km ⁻²	year ⁻¹	year ⁻¹	year ⁻¹		
1	Grey seals	4.49	0.06		0.11	11.39	0.00	0.01
2	Harbour seals	4.61	0.01		0.10	10.12	0.00	0.01
3	Cetaceans	4.35	0.01		0.02	14.00	0.10	0.00
4	Seabirds	4.20	0.03		0.40	83.05	0.29	0.00
	Cod							
5	Cod mature	3.96	0.25	1.17		3.50	0.50	0.33
6	Cod immature	3.15	0.21	2.21		9.11	0.99	0.24
	Haddock							
7	Haddock mature	3.70	0.84	0.72		4.96	0.67	0.15
8	Haddock immature	2.95	0.05	1.67		17.58	0.99	0.09
	Whiting							
9	Whiting mature	4.17	0.27	1.30		4.50	0.53	0.29
10	Whiting immature	3.05	0.21	1.73		9.23	1.00	0.19
11	Pollock	3.94	0.44		0.94	4.69	0.86	0.20
12	Gurnards	3.67	0.13		1.23	4.12	0.65	0.30
13	Monkfish	4.37	0.22		0.48	1.71	0.95	0.28
14	Flatfish	3.51	1.52		1.13	3.77	0.95	0.30
15	Rays	3.88	0.68		0.24	2.24	0.59	0.11
16	Sharks	4.07	0.79		0.34	3.41	0.88	0.10
17	Large demersals	4.33	1.09		0.49	2.44	0.95	0.20
18	Other small fish	3.28	0.78		1.58	5.27	0.95	0.30
19	Mackerel	3.34	4.19		0.77	4.40	0.88	0.17
20	Horse Mackerel	3.17	4.73		0.74	3.70	0.64	0.20
21	Blue Whiting	3.67	1.78		1.50	6.00	0.96	0.25
22	Other pelagics	3.68	2.00		1.80	6.00	0.95	0.30
23	Herring	3.16	5.95		1.50	10.10	0.84	0.15
24	Norway pout	3.30	0.84		1.68	5.60	0.95	0.30
25	Poor cod	3.60	0.07		1.17	3.90	0.95	0.30
26	Sandeel	3.19	1.31		1.83	6.09	0.95	0.30
27	Sprat	3.19	1.37		2.10	8.40	0.95	0.25
28	Nephrops	3.25	0.80		0.73	4.88	0.95	0.15
29	Crustaceans	2.74	9.63		0.87	5.81	0.95	0.15
30	Cephalopod	3.25	1.09		1.98	15.00	0.95	0.13
31	Large zooplankton	2.16	15.10		10.00	35.00	0.95	0.29
32	Small zooplankton	2.03	14.06		18.00	72.00	0.95	0.25
33	Infafauna	2.04	11.86		20.00	80.00	0.95	0.25
34	Epifauna	2.57	15.00		20.00	80.00	0.88	0.25
35	Phytoplankton	1.00	31.87		70.00	0.00	0.95	
36	Detritus	1.00	100.00				0.94	

9.4 Ecopath Model Diet Composition: Balanced

Prey \ predator	Grey seals	Harbour seals	Cetaceans	Seabirds	Cod mature	Cod immature	Haddock mature	Haddock immature	Whiting mature	Whiting immature	Pollock	Gurnards
Grey seals	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Harbour seals	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cetaceans	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Seabirds	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cod mature	0.055	0.038	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cod immature	0.050	0.000	0.000	0.030	0.010	0.010	0.000	0.000	0.009	0.005	0.030	0.013
Haddock mature	0.030	0.115	0.000	0.000	0.036	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Haddock immature	0.000	0.000	0.000	0.014	0.000	0.000	0.000	0.010	0.014	0.005	0.001	0.003
Whiting mature	0.027	0.229	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Whiting immature	0.000	0.000	0.000	0.004	0.000	0.000	0.000	0.000	0.047	0.000	0.000	0.074
Pollock	0.059	0.000	0.001	0.007	0.000	0.000	0.000	0.000	0.007	0.000	0.000	0.000
Gurnards	0.000	0.000	0.000	0.000	0.002	0.000	0.003	0.000	0.000	0.000	0.000	0.000
Monkfish	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Flatfish	0.096	0.005	0.000	0.035	0.039	0.000	0.000	0.000	0.001	0.000	0.029	0.001
Rays	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Sharks	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Other demersals	0.117	0.097	0.022	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Benthopelagic fish	0.010	0.000	0.007	0.030	0.000	0.000	0.039	0.000	0.000	0.000	0.000	0.029
Mackerel	0.000	0.018	0.001	0.030	0.006	0.000	0.000	0.000	0.123	0.000	0.000	0.000
Horse Mackerel	0.000	0.346	0.000	0.013	0.000	0.000	0.000	0.000	0.013	0.000	0.060	0.000
Blue Whiting	0.010	0.000	0.000	0.032	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Other pelagics	0.000	0.000	0.637	0.124	0.107	0.000	0.018	0.000	0.120	0.000	0.012	0.015
Herring	0.054	0.077	0.001	0.352	0.160	0.000	0.001	0.000	0.340	0.000	0.208	0.030
Norway pout	0.038	0.054	0.000	0.020	0.015	0.000	0.009	0.000	0.020	0.000	0.200	0.004
Poor cod	0.010	0.000	0.050	0.001	0.000	0.000	0.001	0.000	0.001	0.000	0.000	0.001
Sandeel	0.436	0.012	0.076	0.050	0.043	0.040	0.030	0.000	0.050	0.020	0.030	0.082
Sprat	0.010	0.000	0.005	0.134	0.001	0.000	0.080	0.000	0.130	0.000	0.090	0.006
Nephrops	0.000	0.000	0.000	0.000	0.014	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Crustaceans	0.000	0.000	0.002	0.035	0.407	0.050	0.370	0.050	0.035	0.050	0.115	0.372
Cephalopod	0.000	0.010	0.005	0.000	0.018	0.000	0.000	0.000	0.000	0.000	0.005	0.000
Large zooplankton	0.000	0.000	0.119	0.020	0.000	0.400	0.000	0.500	0.020	0.450	0.220	0.217
Small zooplankton	0.000	0.000	0.070	0.000	0.000	0.250	0.000	0.150	0.000	0.200	0.000	0.000
Infauna	0.000	0.000	0.000	0.000	0.074	0.150	0.116	0.050	0.000	0.100	0.000	0.054
Epifauna	0.000	0.000	0.000	0.070	0.067	0.050	0.332	0.040	0.070	0.045	0.000	0.098
Phytoplankton	0.000	0.000	0.000	0.000	0.000	0.050	0.000	0.200	0.000	0.125	0.000	0.000
Detritus	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Import	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Sum	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
(1 - Sum)	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000

Prey \ predator	Monkfish	Flatfish	Rays	Sharks	Other demersals	Benthopelagic fish	Mackerel	Horse Mackerel	Blue Whiting	Other pelagics	Herring
Grey seals	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Harbour seals	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cetaceans	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Seabirds	0.000	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cod mature	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cod immature	0.156	0.002	0.000	0.022	0.000	0.000	0.005	0.000	0.000	0.000	0.000
Haddock mature	0.059	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Haddock immature	0.000	0.001	0.000	0.001	0.001	0.000	0.000	0.000	0.000	0.000	0.000
Whiting mature	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Whiting immature	0.052	0.001	0.000	0.020	0.000	0.000	0.007	0.000	0.000	0.000	0.000
Pollock	0.143	0.001	0.000	0.001	0.001	0.000	0.000	0.000	0.000	0.000	0.000
Gurnards	0.002	0.004	0.005	0.001	0.000	0.000	0.000	0.000	0.000	0.002	0.000
Monkfish	0.010	0.010	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Flatfish	0.063	0.004	0.041	0.001	0.037	0.000	0.000	0.000	0.000	0.072	0.000
Rays	0.001	0.000	0.000	0.001	0.020	0.000	0.000	0.000	0.000	0.000	0.000
Sharks	0.000	0.000	0.000	0.042	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Other demersals	0.000	0.000	0.000	0.000	0.056	0.000	0.000	0.000	0.000	0.000	0.000
Benthopelagic fish	0.098	0.010	0.000	0.024	0.100	0.050	0.006	0.000	0.000	0.000	0.000
Mackerel	0.010	0.005	0.000	0.017	0.080	0.000	0.000	0.000	0.070	0.000	0.000
Horse Mackerel	0.000	0.000	0.000	0.010	0.100	0.000	0.020	0.000	0.000	0.030	0.000
Blue Whiting	0.000	0.000	0.000	0.000	0.171	0.000	0.000	0.000	0.020	0.000	0.000
Other pelagics	0.023	0.038	0.010	0.000	0.081	0.000	0.010	0.000	0.150	0.034	0.000
Herring	0.108	0.000	0.130	0.322	0.164	0.000	0.100	0.016	0.050	0.032	0.000
Norway pout	0.122	0.015	0.010	0.002	0.000	0.000	0.010	0.016	0.000	0.005	0.000
Poor cod	0.001	0.000	0.000	0.001	0.001	0.000	0.001	0.000	0.000	0.000	0.000
Sandeel	0.104	0.020	0.050	0.021	0.001	0.000	0.010	0.000	0.010	0.027	0.000
Sprat	0.000	0.000	0.040	0.098	0.000	0.000	0.010	0.000	0.100	0.007	0.000
Nephrops	0.000	0.021	0.130	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.000
Crustaceans	0.001	0.107	0.300	0.111	0.038	0.004	0.010	0.010	0.010	0.010	0.010
Cephalopod	0.045	0.007	0.000	0.153	0.033	0.000	0.000	0.000	0.000	0.122	0.000
Large zooplankton	0.000	0.100	0.000	0.041	0.004	0.648	0.705	0.732	0.590	0.119	0.983
Small zooplankton	0.000	0.102	0.000	0.000	0.000	0.000	0.100	0.215	0.000	0.034	0.000
Infauna	0.000	0.204	0.040	0.000	0.000	0.110	0.005	0.009	0.000	0.136	0.000
Epifauna	0.000	0.347	0.244	0.111	0.112	0.188	0.000	0.000	0.000	0.368	0.000
Phytoplankton	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.007
Detritus	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Import	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Sum	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
(1 - Sum)	0.000	0.000	0.000	0.000	0.000	0.000			0.000	0.000	0.000

Prey \ predator	Norway pout	Poor cod	Sandeel	Sprat	Nephrops	Crustaceans	Cephalopod	Large zooplankton	Small zooplankton	Infauna	Epifauna
Grey seals	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Harbour seals	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cetaceans	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Seabirds	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cod mature	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cod immature	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Haddock mature	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Haddock immature	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Whiting mature	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Whiting immature	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Pollock	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Gurnards	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000
Monkfish	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Flatfish	0.000	0.001	0.000	0.000	0.000	0.000	0.015	0.000	0.000	0.000	0.000
Rays	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Sharks	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Other demersals	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Benthopelagic fish	0.002	0.111	0.000	0.000	0.000	0.001	0.005	0.000	0.000	0.000	0.000
Mackerel	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Horse Mackerel	0.000	0.000	0.000	0.000	0.000	0.000	0.019	0.000	0.000	0.000	0.000
Blue Whiting	0.000	0.000	0.000	0.000	0.000	0.000	0.005	0.000	0.000	0.000	0.000
Other pelagics	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000
Herring	0.000	0.000	0.000	0.000	0.000	0.000	0.031	0.000	0.000	0.000	0.000
Norway pout	0.000	0.000	0.000	0.000	0.000	0.000	0.004	0.000	0.000	0.000	0.000
Poor cod	0.001	0.001	0.000	0.000	0.000	0.000	0.001	0.000	0.000	0.000	0.000
Sandeel	0.000	0.002	0.000	0.000	0.000	0.000	0.019	0.000	0.000	0.000	0.000
Sprat	0.000	0.001	0.000	0.000	0.000	0.000	0.004	0.000	0.000	0.000	0.000
Nephrops	0.000	0.031	0.000	0.000	0.000	0.001	0.002	0.000	0.000	0.000	0.000
Crustaceans	0.193	0.300	0.050	0.050	0.000	0.010	0.002	0.000	0.000	0.000	0.000
Cephalopod	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Large zooplankton	0.752	0.097	0.948	0.950	0.499	0.084	0.592	0.012	0.000	0.000	0.000
Small zooplankton	0.000	0.004	0.001	0.000	0.161	0.105	0.273	0.140	0.030	0.000	0.098
Infauna	0.000	0.161	0.001	0.000	0.051	0.133	0.020	0.000	0.000	0.036	0.149
Epifauna	0.052	0.291	0.000	0.000	0.289	0.240	0.004	0.000	0.000	0.000	0.200
Phytoplankton	0.000	0.000	0.000	0.000	0.000	0.100	0.000	0.710	0.800	0.499	0.378
Detritus	0.000	0.000	0.000	0.000	0.000	0.326	0.000	0.138	0.170	0.465	0.175
Import	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Sum	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
(1 - Sum)		0.000	0.000	0.000	0.000	0.000		0.000	0.000	0.000	0.000