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PROTECT

Marine Protected Areas as a Tool for Ecosystem Conservation and Fisheries Management

PRIORITY 8, Scientific Support to Policy (SSP)

Area 1.3 Modernisation and sustainability of fisheries incl. aquaculture-based production systems

24 Month Periodic Activity Report

Section 2 – Modelling MPA effects

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PROTECT 24-month Periodic Activity Report

Section 2 – WP5 Modelling to evaluate MPA effects

List of contents:

G. Holland and J. Sharples (2006). Variation in the abundance of sandeels Ammodyes marinus off southeast Scotland: an evaluation of area-closure fisheries management and stock abundance assessment methods. ICES J. Mar. Sci., Ices J Mar Sci 63, 1530-1550.

Annex 7. Frederiksen, M. & Wanless, S. 2006. Draft report, PROTECT Work Package 5, Case Study 2: Assessment of the effects of the Firth of Forth sandeel fishery closure on breeding Seabirds.

Annex 8: Armstrong, C.W. A note on the ecological–economic modelling of marine reserves in fisheries. *Ecological Economics* (in Press).

Annex 9: Prem Wattage: Draft Environmental valuation questionnaires: (A) Cold Water Corals, (B) Seabirds, fish and MPAs.

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1. PROTECT modelling strategy

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1.1 Tools and concepts

WP5 seeks to develop a suite of modelling tools for predicting and assessing the expected performance of planned and implemented MPAs in the context of EU Fisheries and environmental priorities. WP5 is the largest work package in PROTECT, consuming more than 1/3 of the total number of man-months available in the project.

During the first reporting period, a range of WP and CS sessions and work has led to development of modelling strategies for each case study, as outlined below, and modelling work is in good progress. In Table 1.1.1, the modelling approaches included in the project Technical Annex are listed with definitions as well as their linkages to other ongoing research projects (in *italics*):

Table 1.1.1: Overview of modelling approaches in the PROTECT Technical Annex and their linkages to other ongoing research activities

Tool	**Definitions and linkages to ongoing projects**
Ecosystem indicators and community metrics	Absolute and relative abundances of (i) target species to be protected and key species in the ecosystem and (ii) size categories, trophic guilds or life-history types that determine the structural and functional biodiversities of the systems *Specific and generic results obtained in projects RESPONSE, EFEP, and BIOMEX will be reviewed in WP 2, and tested and applied in WP 5*
Stock specific spatial models	A tool able to model the temporal and spatial distribution of the resource and the fishery, developed to use flexible spatial and temporal scales depending on data availability, will provide quantitative information on the effects of shape and size of the MPA and will be a cornerstone of the next level of model integration *Input from ongoing or finalised EU projects targeting (i) individual fish behaviour with data storage tags (CODYSSEY and LIFECO), (ii) the construction of coupled IBM and hydro/ecosystem models on fish early life stage and zooplankton transport and survival (STORE and LIFECO), and (iii) the distribution of fish in relation to environmental conditions (STORE and HERSUR)*
Multi species and multi fleet models	i) A traditional age-structured model extended to account for species and technical interactions and seasonal variations (*results from the FP6 project BECAUSE)*. Prediction of recruitment and other uncertain parameters will be handled by stochastic simulation techniques and sensitivity analyses. ii) Implement socio-economic model components to simulate the reaction of fishers to MPAs and other management measures (*FP6 EFIMAS*) iii) Make models and management criteria operational with respect to simulation performance, user-friendly and documented software and application of model/software to case studies.
Bio- and socio-economic models	i) Use the most appropriate modelling frame for each case study, drawing from the range of options and data available and reviewed during earlier WPs. Develop existing bio-economic models to further explore MPA performance and effectiveness (*FP6 EFIMAS*). ii) Environmental valuation techniques will be applied to resources and environmental attributes that do not command a market price

Modelling strategies and work have been addressed in great detail during the kick-off meeting, followed by local meetings and active email correspondence. Based on these, the WP5 group has developed the following definitions for each modelling tool (Table 1.1.2):

Table 1.1.2 : Proposed modelling tools and definitions on modelling tools to be used in WP5

Tool	**Definition**	**Comment**
Ecosystem indicators and community metrics	Indicators[1] and community metrics describing biodiversity to assess and predict consequences of introducing MPAs in relation to ecosystem management objectives	A priori useful only for existing MPA?
Stock-specific spatial models	Single species models that focus on population dynamic processes, possibly in relation with environment	-should account for available habitat, distribution range, dispersion of the species involved, meso and large scale fish behaviour, e.g. homing, migration and dispersion relative to the location of spawning and nursery areas -should account for the temporal and spatial distribution of the resource and the fishery, developed to use flexible spatial and temporal scales depending on data availability, will provide quantitative information on the effects of MPA shape and size[2]
Multi-species multi-fleet spatial models	Age- or length- structured model extended to account for species and technical interactions and seasonal variations[3]	Same as above
Bio- and socioeconomic models		-study economic trade-offs between conservation effects and cost implications of MPA implementation -effects of alternative MPA management measures and designs, and associated potential benefits and/or drawbacks -impacts from activities other than commercial fishing incorporated where appropriate and data permits -environmental valuation techniques for unpriced effects

[1] *Definition of an indicator: a metric that indicates MPA effects and can be used to test these effects*

[2] *Refers to ISIS-Fish (but no multifleet aspects)*

[3] *See e.g. the BECAUSE project and 4M (with species interactions) model and ISIS-Fish (without species interactions)*

Synthesis of modelling approaches

A synthesis of modelling approaches for MPA evaluation was provided for the review under WP2 (see WP2 report). The synthesis presents a state of the art of modelling approaches. It discusses the two kinds of approaches have been envisaged to assess ecological and fisheries-related impacts of MPAs: mathematical models depicting the dynamics of populations, communities or ecosystems, that are generally used for policy screening analyses; and empirical approaches based on statistical modelling of field data that are used to test effects and provide diagnostics about the ecosystem and resources. Statistical models lead to defining empirical indicators and sampling designs for long-term programmes of experimental monitoring. Mathematical models enable to explore issues related to MPA design and its consequences on the dynamics of populations and fisheries; they provide reference points against which system dynamics can be gauged.

Modelling discussions at the first thematic workshop.

An overview of the previous synthesis was presented during the plenary meeting. In a second step, the WP5 coordinator has worked with each case study to assist structuring and designing project modelling actions adapted to the questions raised, and to the data available in each case (see further details in the case specific paragraphs below).

Modelling strategy for PROTECT

In the light of previous discussions, it is necessary to explicitly define each tool, and to rephrase the typology of tools presented in the technical annex, which bears some ambiguities. Hence, the term "bio-economic modelling" encompasses a wide range of tools and may not have the same meaning for different partners. Ecosystem indicators correspond to ecological performance, but the term indicator may also be ambiguous. Multi-fleet models may be spatial, and they may also incorporate economic features.

Additional terms are proposed below; they are justified by the variety of standpoints for assessing MPA performance.

1.2 Standpoints and scales for assessing MPA performance

MPA performance may be envisaged from disciplinary **standpoints,** such as:

- Ecological: e.g. what are the effects of the MPA on the ecosystem and resources?
- Economical: e.g. what are the effects on the dependent economic activities, among which fishing is a primary activity, but not the only one? Fishing includes commercial both fishing and recreative fishing.
- Sociological: e.g. what are the perceptions and attitudes of users with respect to the MPA? What is the overall socio-economic value of the MPA? Are there any conflicts?

It may also be assessed at several **spatial and temporal scales**:

Spatial scales:

- At the scale of the MPA: are there differences in ecological status within and outside the MPA?
- At the scale of the ecosystem or fishery: what are the (real or anticipated) consequences of the existence of the MPA versus a scenario without MPA, on the ecosystem and/or fishery?

T*emporal scales:*

- Diagnostic tools: current status of resources and fisheries
- Predictive tools: what would be the consequences of changes in MPA design or of the current design under a range of evolutions of ecological and economic conditions?

These standpoints relate to management objectives and therefore to a range of success criteria. They largely determine the kinds of tools to be used for evaluating the performance of MPA and hence the data requirements for implementing these tools.

1.3 Definitions and revised typology of approaches

Based on parag. 1.2, approaches may be considered from the methodological standpoint and from the disciplinary standpoint. The methodological standpoint distinguishes empirical approaches based on the analysis and statistical modelling of data, from dynamic modelling approaches describing the dynamics of the system (the fishery) among which many models have not been calibrated from field data (see WP2 review). Approaches provide assessment at distinct scales and are complementary (Table 1.3.1).

Table 1.3.1. Definitions of methodologies and complementary aspects with regard to the assessment of MPA effects and subsequent provision of indicators.

Empirical approaches	**Dynamic modelling**
Local assessment Snapshot information	Assessment at system (i.e. fisheries or ecosystem) scale and throughout a period of time Integrate knowledge about system components Use of local knowledge for assessment at the system scale
Statistical testing of effects Field validation Actual estimates of variance components and sizes of MPA effect	Quantitative assessment of system dynamics Exploration of scenarios (e.g. MPA design) and hypotheses Possible projections in the future Generate hypotheses to be tested from field experiments
Local diagnostic Direct link with monitoring	Overall diagnostic on system
Empirical indicators	Model-based indicators (outputs of models)
Control sites but in general no system-wide reference point	Provision of theoretical reference points
In principle, only applies to existing MPA, but may be used before MPA creation to compare potential sites for MPA designation.	Apply to existing MPA or MPA projects

It is also necessary to **define indicators and metrics**. An *indicator* is a function of observations or the output of a model, which value indicates the present state and/or dynamics of the system of interest (FAO 1999[1]). It may be linked to a management question or a research hypothesis, and should meet desirable performance criteria in this respect. In order to stress the importance of validating indicators through success criteria, we define a metric as a biological response at a given scale, while an indicator is a metric displaying desirable performance in terms of MPA assessment. Indicators linked to research questions about MPA effects (termed "assessment indicators") are used for testing hypotheses. The performance of these indicators mainly lies in their statistical properties and their sensitivity to the question addressed. Indicators aimed at managers (termed "management indicators" below) should be simple and understandable by non-experts. They are generally built from assessment indicators, and communication issues provide additional criteria for assessing their performance. In the project, we will mainly use assessment indicators, but may be interested at deriving management indicators for stakeholders.

The two approaches require different types of data:

- Empirical approaches need to test for differences over time and space in ecological or economic metrics; they thus require data collected according at a series of dates, preferably before and after

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[1] FAO, 1999. Indicators for sustainable development of marine capture fisheries. Food and Agriculture Organisation, Rome.

the establishment of the MPA. They also require data collected in the MPA and in several other sites outside the MPA (referred to as control sites in assessment methodology).

- Dynamic modelling need inputs to depict and parameterize the dynamics of the fishery or populations of interest. Knowledge about the population dynamics and the components of fishing effort is needed. In order to investigate MPA, spatially-explicit models are needed. Models may be calibrated with respect to observed catch and abundance data if available.

Based on the previous consideration, the following typology of tools, obtained by crossing and methodology, is proposed in relation with the terms used in the PROTECT Technical Annex (Table 1.3.2):

Table 1.3.2. Revised typology of tools for MPA evaluation.

		Focus of assessment		
		Ecological performance	Fisheries/Economic performance	Sociologic issues
Methodology	Empirical approaches	- Ecosystem indicators and community metrics -Population-level (empirical) indicators	-Economic indicators -Environmental valuation techniques	-Environmental valuation techniques? (*Willingness To Pay surveys*)
	Dynamic modelling	-Ecosystem models (*ex: ECOSIM/ECOSPACE*) or multispecies models (*ex: 4M*) -Spatially-explicit population models (*ex: ISIS-Fish*)	-Bio-economic dynamic modelling (*ex: TEMAS, ISIS-Fish, BEMCOM*)	

Note: Indicators may also be computed from the results of dynamic models (denoted model-based indicators). When it might be ambiguous, "empirical" indicator denotes an indicator obtained from the statistical analysis of field data.

1.4. Case study modelling strategies

1.4.1 Modelling strategy for Baltic Case Study (CS1)

The main objectives of the case study being addressed through modelling are:

- Evaluate the potential of MPAs to protect a top-down controlled ecosystem from the effects of fishing.
- Develop a suite of implementation, monitoring and assessment tools to manage the fisheries impacts on cod and clupeids stocks.
- Assess the socio-economic impacts of potential MPAs on cod, herring and sprat fisheries.

More specifically, the activities undertaken seek to address the following *objectives*:

1. Evaluate changes in ecosystem indicators and community metrics describing the structure and functioning of upper trophic ecosystem levels;
2. Evaluate variability in transport of larvae and juveniles and distribution of potential habitats (nursery, feeding and spawning) in relation to hydrodynamic forcing to determine exchange rates between protected and unprotected areas;
3. Evaluate changes in stock structures and abundances in different regions before and after the implementation of MPAs, accounting for 2.
4. Evaluate temporal and spatial dynamics of fishing fleets, and economic consequences of different MPA designs.

Tools and data used for this purpose are listed in Table 1.4.1.

Table 1.4.1

Objective	Tools and data
1	• Empirical analysis of growth rates and condition indices for herring, of cod and sprat abundance and distribution • Multispecies modelling within the ICES Baltic multispecies study group, (spatially explicit, i.e. according to sub-division, linked to BECAUSE
2	• Empirical analysis of existing time-series of cod abundance, habitat and hydrodynamic models
3	• Empirical analysis of changes in the abundance of exploited stocks • Spatially-explicit fisheries dynamic model: ex: ISIS-Fish
4	• Empirical analysis of effort data from log-books and VMS (temporal and spatial dynamics of fleets) • Dynamic bio-economic model: TEMAS, BEMCOM (consequences on fleet capacity), ISIS-Fish (consequences of MPA designs) •

Note: Empirical analyses that aim at evaluating temporal changes in stock structures, abundances and community metrics need to account for changes in fishing effort and mortality.

The case study partners have defined a set of specific questions to be addressed by modelling (see Table 1.4.2). Most of these questions pertain to the evaluation of existing (questions 1 and 2) and anticipated (questions 3 to 6) MPA designs, which require spatially-explicit dynamic models of mixed fisheries with in addition a description of spawning location and timing. Question 7 may be handled through a bio-economic dynamic model, while question 8 may be explored using a dynamic model with species interactions or by analysing ecosystem and community empirical metrics (for existing MPA only).

Table 1.4.2: Specific question to be addressed in the project and tools that are needed

No.	Baltic Case Study research question and evaluation tools
1	Evaluation of area and seasonal closures enforced in 1995-2003, 2004 and potential alternatives to ensure undisturbed spawning in 2005 and subsequent years
2	Evaluation of closures enforced in 2005 to reduce fishing mortality
3	Evaluation of alternative closures to reduce F, e.g. original EU commission proposal which consider not only spawning areas as aggregation areas
4	Do we need to close nursery areas to reduce discards and utilise growth potential better? Should closures be gear or fleet specific?
5	Do other fisheries than those targeting cod need to be banned in specific seasons/areas?
6	Effect of closures in combination with alternative management measures, e.g. effort regulation?
7	Socio-economic impact of above closures for the fishery
8	Evaluation of ecological impact of above closures

1.4.2 Modelling strategy for North Sea Sandeel Case Study (CS2)

The objectives of the North Sea case study being addressed through modelling are:

- To evaluate the potential of MPAs to protect a "wasp-waist" ecosystem (North Sea sandeel areas) from the effects of fishing
- To outline and develop a suite of implementation, monitoring, assessment and management tools for MPAs in this type of ecosystem aiming to assist managers in assessing (i) the fisheries impact on the sandeel stock and the ecosystem as sandeels represent a key link between lower trophic levels and top predators, (ii) the impact of introducing MPAs with varying level of protection, and (iii) the impact of MPAs on the sandeel fishery and related socio-economic effects.

Approaches and tools used for this purpose are listed in Table 1.4.3.

Table 1.4.3.

Approach	Tools used	Objective
Empirical approaches	-Statistical analysis of seabird and sandeel abundance series -Statistical analysis of effort data	-Evaluate changes in sea bird and sandeel populations -Evaluate changes in effort distribution
Dynamic modelling	-Larval dispersion – bank specific population dynamic – bio-economic coupled model	-Evaluate potential spatial closures -Evaluate changes in effort

	components: SLAM – SPAM – BEMCOM - Species modelled are sandeel + fishers, birds - Bio-physical model for East cost of Scotland	distribution - Evaluate closure at Firth of Forth

1.4.3 Deep-water corals (CS3)

The objectives of this case study being addressed through modelling are:

- To evaluate the potential of MPAs to protect deep water coral ecosystems from the effects of fishing
- To outline and develop a suite of implementation, monitoring, assessment and management tools for MPAs for deep water coral ecosystems. These methods are intended to assist managers in assessing (i) the fisheries impact on coral and coral communities, (ii) the impact of introducing MPAs with varying level of protection on the ecosystem, and (iii) the impact of MPAs on the fisheries operating in deep water coral areas and related socio-economic effects.

The further objectives for the bio-economic modelling of deep-water coral (DWC) reef management are:

1. Design a bio-economic model with renewable and non-renewable interaction, where the non-renewable resource (DWC) enters into the growth function of the renewable resource (commercial fish species). Non-use values of the DWC could also be included in the model description.
2. Explore management involving gear restrictions/marine reserves/transferable habitat quotas, applied to the model in 1.
3. Design an applied model using a specific fishery in the proximity of DWC; test and simulate management options.

Approaches and tools used for this purpose are listed in Table 1.4.4.

Table 1.4.4.

Approach	Tools used	Objective
Empirical approaches	-Statistical analysis of abundance series -Mapping of coral reefs -Statistical analysis of effort data (logbooks and VMS where available)	-Evaluate changes in abundances -Evaluate changes in effort distribution -Provide baseline information before coral closure in 2006
Dynamic modelling	-Bio-economic dynamic modelling, *ISIS-Fish : orange roughy (and possibly grenadier, black scabbard) *Heuristic model with renewable (fish) and non-renewable (coral) resources accounting for non-use value of coral	-Evaluate consequences of fisheries-related closures -Evaluate changes in effort reallocation to other areas or metiers

2. Description of modelling tools used in the project

In order to facilitate the readability of modelling tools in the project, in particular to promote synergies across approaches, a template was elaborated to describe each modelling approach used in the project. The synthesis of these descriptions provides the information needed to utilize each tool.

2.1 ISIS-Fish

ISIS-Fish is a spatially and seasonally-explicit fisheries simulation model aimed at policy evaluation, in particular MPA (Mahévas and Pelletier 2004, Pelletier and Mahévas 2005). From version 2.0, ISIS-Fish also includes an economic description of the fishery, based on a range of exploitation costs. Economic conditions may impact fishing effort allocation following any pre-specified equation (Pelletier et al. (in prep.).

2.1.1 Main issues addressed – Non technical description

ISIS-Fish is a deterministic simulation model aimed at reproducing the dynamics of a mixed (multispecies multifleet) fishery to evaluate the consequences on population and exploitation of a variety of management scenarios, particularly MPA. The model focuses on mixed fisheries issues (several fishing activities, several vessel types, e.g. technical interactions), and is aimed at being generic.

2.1.2. Model scales (space and time)

- Spatially-explicit with spatial scale adaptable to data resolution
- Zones independently defined for population, exploitation and management
- Monthly time-step

2.1.3. Main model features

Populations

- Seasonal migrations
- Spawning areas and nurseries with possible transfers to be specified
- Relationship between spawners and egg production

Exploitation

- Structured in metiers and strategies
- Exploitation costs and revenues (calculation of economic indicators and impact on effort reallocation)
- Fisher's response to management and economic conditions (fisher reallocate effort to other metiers or strategies)

2.1.4. Main assumptions (of present version 3.0)

- No interspecific relationships between populations, but it is possible to make natural mortality of a population class dependent on other species' or other classes' abundances
- Instantaneous migrations of populations
- Each population stage is uniformly distributed over a zone at a given month
- The effort of a giving fishing activity is uniformly distributed over a zone at a given month;

- Instantaneous relationship between fishing mortality and fishing effort is proportional to the intersection between population area and effort area during that month;
- No investment dynamics for fleets, only short-term reallocation issues

2.1.5. Data requirements

Populations

- If spatial model for population: spatial and seasonal description of population dynamics
- Biological parameters, spawner-egg relationship, catchability coefficients

Exploitation

- If spatial model for exploitation : spatial and seasonal description of exploitation structure
- Selectivity function for each fishing gear
- Estimates for parameters that are needed to compute fishing mortalities across fishing activities : target factors, standardization factor for fishing gears

Management

- Definition of management scenarios: For ex., a TAC is described by a population, the TAC level, the simulation years during which it applies. A closure is described by the area closed, the metiers or gears targeted (if relevant), the months closed, and the years closed.

NB: A guide is currently being elaborated for proposing methods to estimate exploitation-related parameters from conventional log-books and interview data.

2.1.6. Software and computer requirements

- Software: free software no requirements, runs from Java version 1.5 and above. Everything is freely downloadable.
- The user’s manual is available at http://www.ifremer.fr/isis-fish
- Computer: runs OK on any recent computer. Slower when the amount of fishing activities increases.

2.1.7. Outputs

- Simulation designs can be run to test a range of model assumptions and run sensitivity analysis on model parameters.
- Results visualized in the software through an interface
- Results exported under text format
- Available R scripts for building graphics from results

2.1.8. Target audience/ users

- Scientists
- MPA managers with the help of scientists
- Fishers and industry with the help of scientists

2.1.9. References

1. Drouineau, H., S. Mahévas, D. Pelletier, B. Beliaeff. 2006. Assessing the impact of marine protected areas on the hake-nephrops fishery of the Bay of Biscay using ISIS-Fish. *Aquatic Living Resources* **19**, 15-29. (Application)
2. Mahévas, S., & D. Pelletier, 2004. ISIS-Fish, a generic and spatially-explicit simulation tool for evaluating the impact of management measures on fisheries dynamics. *Ecological Modelling* 171, 65-84. (Model description version 1.0)
3. Pelletier D. & S. Mahévas. 2005. Spatially-explicit fisheries simulation models for policy evaluation. *Fish and Fisheries* 6, 307-349. (Model description version 1.5)
4. Pelletier D., S. Mahévas, H. Drouineau, Y. Vermard, O. Thébaud, O. Guyader, B. Poussin. Assessing the bioeconomic sustainability of complex fisheries under a wide range of policy options using ISIS-Fish. *Ecological Modelling (in prep.)* (Model description version 2.0 and 3.0)
5. http://www.ifremer.fr/isis-fish

2.2 Sandeel Larval Advection Model (SLAM)

2.2.1. Main issues addressed – Non technical description

SLAM is aimed at identifying the consequences on sandeel recruitment dynamics of a stochastic hydrographical setup, particularly given a dynamic mosaic of fishing banks related to MPAs.

2.2.2. Model scales (space and time)

- 8x8 km^2 x 5-8m depth layers in a North Sea grid with residual flow fields and temperature
- Particle tracking per hour (time step)
- Sandeel fishing banks correspond to sandeel optimal habitat defined as shape files with a 1km resolution.
- Daily time-steps for updates of population dynamics

2.2.3. Main model features

Populations
- Forward/backward time advection of sandeel larvae, using an individual-based model
- Explicit account of spatial and temporal heterogeneity turbulent dispersal
- Growth and survival of larvae until reaching suitable habitats for recruitment
- Equal seeding of drifting larvae for each sandeel spawning area

Habitats
- Hydrological characteristics taken as output from HAMSOM model runs for 35 years (Schrum and Backhaus 1999; Schrum et al. 2006)
- Spawning areas defined as historic fishing grounds with suitable sand substrate
- Predator distributions taken from IBTS and other databases

Exploitation
- Fishing mortality on 0-group

2.2.4. Main assumptions

- No effect of tidal flow with present biological resolution
- No active oriented horizontal migration of larvae
- Instantaneous settling or death at specified larval size
- No redistribution of settled sandeels
- Reproduction and growth determined by temperature

2.2.5. Data requirements

Populations

- Initial hatch time distribution determined by a temperature model (may be modified with new data on spawning time by area)
- Predator fields for larvae
- Predation risk at different activity levels
- Food availability for sandeel larvae (standard assumptions, may be changed to input by NPZ model)
- Feeding activity as a function of life stage, temperature and food availability for sandeel larvae

Physical environment

- Bathymetry of the North Sea
- Stochastic variations around average flow field of the North Sea
- Stochastic variations around average temperature fields
- Mapped sandeel optimal habitats

Exploitation

- Sandeel 0-group are fished when expected availability is above a set limit based on yields in kg x price per kg vs. costs (investment + running)
- Sandeel 0-group are only available at sandeel fishing banks
- Sandeel are only available in certain life stages and during light hours
- Marginal fishery where alternative targets for the fleet are chosen when available; primarily Norway pout, blue whiting, and sprat with herring by-catches

Management

- Definition of management scenarios: Most important transportation routes are analysed to secure sandeel recruitment on all available sandeel habitats.

2.2.6. Software and computer requirements

- Software: software development primarily in Python; no particular requirements, runs from PC. Everything is freely downloadable
- A user’s manual will be developed.
- Computer: runs OK on any recent computer. Database requirements for hydrology is 200GB hard disc. Individual-based core model requires a Fortran 90 compiler

2.2.7. Outputs

- A range of model assumptions can be tested and sensitivity analyses can be run on model parameters.
- Results visualized in the software through various interfaces under construction
- Results exported under text format.

2.2.8. References

1. Christensen, A., U. Hochbaum, I. Alekseeva, H. Jensen, H. Mosegaard, U. Høgsbro Thygesen, M. St. John, & C. Schrum. Sandeel larval transport patterns in North Sea from an individual-based hydrodynamic egg and larval model (*In prep).*
2. Schrum C, Alekseeva I, & M.A. St John. 2006. Development of a coupled physical-biological ecosystem model ECOSMO part i: Model description and validation for the North Sea. *Journal of Marine Systems.* Doi:10.1016/j.jmarsys.2006.01.005
3. Schrum C, & J.O. Backhaus. 1999. Sensitivity of atmosphere-ocean heat exchange and heat content in the North Sea and the Baltic Sea. *Tellus Series A: Dynamic Meteorology and Oceanography,* 51A (4), 526-549.

2.3 The Sandeel Population Analysis Model (SPAM)

Model is presently under construction

2.3.1. Main issues addressed

- Population dynamics component to interact with an economic model of fishery.
- Spatially and temporally resolved fisheries issues (dedicated fishing activities with population distribution knowledge)

2.3.2. Model scales (space and time)

- Sandeel fishing banks = sandeel optimal habitat defined as shape files with a minimum resolution.
- Sub-daily time-steps

2.3.3. Main model features

Population related
- Relative recruitment level per bank (input from SLAM)
- Recruitment scaled by spawning stock biomass (SSB) of year -1 for each bank (self-generated)
- Population dynamics by area (initiated at year (1-x) from observations of landings and age structure, with x years of burn in period)
- Relationship between spawners and egg production (statistical submodel; Boulcott et al., 2006)
- Spawning biomasses by area (self-generated by (a) x (b) x (d) x (f) with different scenarios of predator distribution and fishing effort)
- Sandeel size and age distribution by area (generated from ECOSMO production fields and sandeel growth responses x (c))

Habitats
- Sandeel foraging areas defined as historic fishing areas near suitable sand substrate
- Sandeel potential settling and spawning banks defined as historic fishing banks with suitable sand substrate
- Seabird feeding areas based on radius around known colonies with species specific flight distances
- Fish predator distributions taken from IBTS and other databases

Exploitation

- Effort allocation according to time series of fishing pattern (statistical model of bank specific effort distribution in relation to expected (year-1), and realised catch rates)
- Fishing mortality by area determined by catch-effort, given bank specific population dynamics and sandeel behavioural pattern = sandeel availability
- Exploitation structured by strategies, vessel size, distance from harbour,

2.3.4. Main assumptions

- Sandeel behaviour optimised by life history maximisation of fitness in an evolutionary time scale
- No redistribution of settled sandeels
- Reproduction determined by size, age, area and temperature
- Activity period determined by temperature, light, feeding opportunity and satiation function

2.3.5. Data requirements

Populations

- Initial size and age structured abundance estimates of sandeel per area (initialised from bank specific catches and estimated partial Fs – burn period of x years free run)
- Annual bank specific relative recruitment rates from SLAM
- Average predator fields for settled sandeels (from IBTS and other cruises)
- Predation risk at different activity levels (input from multi species studies at Dogger by CEFAS and at Firth of Forth by FRS)
- Food availability for settled sandeels (e.g. NPZ model output by ECOSMO Hamburg Univ.)
- Feeding activity as a function of max consumption rate by life stage, sandeel density, food availability, temperature, and predation risk

Physical environment

- Bathymetry of the North Sea
- Stochastic and tidal variations around average flow field of the North Sea
- Stochastic variations around average temperature fields
- Light and water transparency variation
- Mapped sandeel optimal sand habitats (revised maps produced 2006 by Henrik Jensen DIFRES)
- Mapped restrictions to fishing activities

Exploitation

- Sandeels are fished when expected availability is above a set limit based on yields in kg x price per kg vs. costs (investment + running)
- Sandeels are only available to the fishery at or close to sandeel fishing banks
- Sandeels are only available according to behavioural model.
- Alternative targets for the fleet are primarily Norway pout, blue whiting, and sprat with herring by-catches
- Trips are limited by degrading value of landings with storage time x temperature
- Effort is restricted by management and risk of by catch

Management

- Definition of management scenarios: A mosaic of closed fishing banks are parameterized by the sum of estimated sandeel biomasses per bank. The pattern of closure is varied until an optimal solution to a given set of objectives is achieved.

- Objectives stated in the GOIS table concerning primary goals of sandeel population sustainability, secondary and yield will be investigated

2.3.6. Software and computer requirements

- Software: A demonstration β-version has been developed in Python; no requirements, runs from PC. Everything is freely downloadable
- A user's manual will be developed
- Computer: runs OK on any recent computer

2.3.7. Outputs

- A range of model assumptions can be tested and sensitivity analyses can be run on model parameters
- Results visualized in the software through various interfaces under construction
- Interface to fishery economic part is being developed

2.3.8. References

1. Boulcott, Philip, Peter J. Wright, Fiona M. Gibb, Henrik Jensen and Iain M. Gibb. 2006. Regional variation in the maturation of sandeels in the North Sea (ICES J. Mar. Sci. Advance Access published 18 Dec 2006 18 pp)

2.4 Statistical models for Assessment of the effects of the Firth of Forth sandeel fishery closure on the ECOsystem - AFFECO

2.4.1. Main issues addressed - Non-technical description

Evaluation of changes in seabird foraging ecology and demographic performance linked to the presence/absence of a sandeel fishery:

- Is seabird breeding success linked to regional sandeel availability, as predicted by the SPAM model?
- Did the activity of the sandeel fishery off the Firth of Forth in the 1990s affect breeding seabirds?

These analyses are based on the statistical modelling of field data. Using detailed data from the long-term intensive study on the Isle of May, as well as less intensive data from several other seabird colonies in the region, we evaluate whether the opening of the fishery in 1990 and the subsequent closure from 2000 affected breeding seabirds. We also test whether seabird breeding success is correlated with sandeel availability, as predicted by the SPAM model; this will serve as a validation of SPAM output in one region. All these analyses take into account variation in other environmental variables, such as sea surface temperature, and thus allow robust conclusions about important ecological relationships.

2.4.2. Available data

- Detailed annual data on seabird breeding success and other demographic and foraging parameters (diet composition) from the Isle of May, 1986-2005.

- Extensive annual data on breeding success of black-legged kittiwakes in several colonies inside and outside the Firth of Forth sandeel closure, 1986-2005.
- Predicted sandeel availability in the Firth of Forth from the SPAM model.
- Other environmental data (sea surface temperature etc).

Sampling design: data are available before, during and after fishery was active, in some cases inside and outside closure zone. Number of years and locations (colonies) variable (up to 20 years and >25 locations).

2.4.3. Methods to be considered for the analysis

- Generalized Linear Models
- ANOVA with Before-After Control-Impact (BACI) design.
- Multivariate ANOVA and regression.
- Non-parametric multivariate methods
- Possibly others

2.4.4. Main assumptions of approach

- Fishery/closure effect is assessed through a significant interaction between location and year effect in BACI design.
- Seabird population size changes too slowly to be an appropriate response variable. Demographic and foraging parameters are suitable for investigating effects of the closure.
- Closure effect can be demonstrated statistically through a significant interaction between location and period effects. 'Before-after-before' temporal design helps interpretation. Other environmental covariates may be included.

2.4.5. Software and computer requirements

- Standard software used throughout (e.g. SAS, CANOCO).
- No computer performance issues.

2.4.6. Target audience/ users

- Fisheries and marine scientists.
- Fishery managers.
- Relevant NGOs.

2.5 Statistical models for describing fleet movements in the Baltic Sea.

2.5.1. Main issues addressed - Non technical description

- Identification of MPA-induced fleet movements in the Baltic Sea cod, sprat and herring fishery
- Identification of MPA induced compensation actions of fishers'
- Identification of most important factors affecting fishers' location choice
- Identification of socio-economic implications of MPAs enforced in the Baltic Sea

2.5.2. Methods to be considered for the analysis

- Probabilistic neural networks (PNN) (Specht, D. F., 1990. Probabilistic Neural Networks, Neural Networks, Vol. 3, pp. 109-118), a non-parametric classification (discrete choice) approach
- Random utility model, RUM (Holland and Sutinen 1999)

2.5.3. Model scales (space and time)

- Space: ICES-square + fishing ground and landing port coordinates in metric scale (from SDs 22-28 i.e. the whole area of the Baltic Sea cod fishery)
- Time: Julian day (i.e. days 1-365)
- Years 1996 – 2005
- Analysis at tow scale

2.5.4. Main model features

- vessels type (e.g. stern trawler, side trawler)
- vessel characteristics (length, kW, GRT, crew size, building year)
- gear (e.g. trawler, gill-netter, long-liner)
- homeport (name + coordinate in metric scale)
- landing port (name + coordinate in metric scale)
- distance from the landing port to fishing ground
- effort in hours
- catch by species
- Julian day
- ICES-square of a fishing ground
- coordinate of a fishing ground (if available) to calculate steaming distance in between landing port and fishing ground

2.5.5. Main assumptions

- MPA-induced redistribution of fishing effort can be determined by the expected returns to individual fishers from the alternative fisheries and locations

2.5.6. Available data

- Swedish, Polish and Danish economic and catch effort log book data from years 1996 – 2005 consisting of over 10^6 rows of input data (tow-by-tow based analyses)

2.5.7. Outputs

- PNN: nonparametric Bayesian probability density of fishers' location choice
- RUM: parametric probability density of fishers' location choice

2.5.8. Software and computer requirements

PNN:

- Software options: 1) Statistical neural networks (www.statsoft.com), 2) Neural tools Professional + Risk Industrial 4.5. Optimizer (www.palisade.com), 3) MatLab neural network Toolbox (www.mathworks.com) etc.

- Computer requirements: Depends highly on the given amount of input data, these analyses are done using AMD Athlon™ 64 X2 Dual Core Processor 4400+2.20 GHz, 2.00 GB of RAM, Physical Address Extension

RUM:

- Software options: 1) Statistica (www.statsoft.com), 2) SAS (www.sas.com), 3) MatLab (www.mathworks.com) etc.
- Computer requirements: Depends highly on the given amount of input data, these analyses are done using AMD Athlon™ 64 X2 Dual Core Processor 4400+2.20 GHz, 2.00 GB of RAM, Physical Address Extension.

2.5.9. Target audience/ users

- Scientists
- MPA managers with the help of scientists
- Fishers and industry

2.5.10. References

1. Holland, D.S., Sutinen, J.G., 1999. An empirical model of fleet dynamics in New England trawl fisheries. *Canadian Journal of Fisheries and Aquatic Sciences* **56**, 253–264.
2. Specht, D. F., 1990. Probabilistic Neural Networks. *Neural Networks* **3**, 109-118.

2.6 Statistical modelling and simulation of fishing effort redistribution in relation to coldwater coral closures

2.6.1. Main issues addressed - Non-technical description

- Closing areas of important habitat, such as deep water corals, can have a significant effect on the harvest of certain fish species, especially if there is an ecological link. Assuming different scenarios, what would be the biological effect of deep water closures on a commercial species?
- How would such closures affect harvest displacement and harvest levels in the short and long term?

The analysis is based on the statistical modelling of Catch Effort Return (CER) data from New Zealand and Vessels Monitoring System (VMS) data from Ireland. The CER data are very detailed and cover quantity and location (lat/long) of all orange roughy catch within the south of New Zealand between 1989 and 2006. The VMS data are less detailed and contain movement of 4 Irish vessels targeting orange roughy but without information on catch. Using these data in combination with known deep water coral distribution and biological data on orange roughy, we evaluate different scenarios of interaction between corals as a habitat and orange roughy in response to closures. We also evaluate the impact such closures would have on effort distribution in the short and long term. This will provide some indication to policy makers about the impact of spatially protected habitat areas.

2.6.2. Available data

- CER data of orange roughy: Ministry of Fisheries, New Zealand

- Fishing start/end date
- Start/end latitude/longitude (bottom trawling)
- Fishing duration, vessel headline height, number of tows, wing spread, etc.
- Vessel characteristics
- Total quantity harvested (kg)
- Time period: 1 Oct 1989 - 31 Aug 2006
- Number of observations: 70,919
- General location: South New Zealand (Quota Management Area ORH 3B)
- VMS data: Ireland
- Movement of 4 Irish vessels targeting orange roughy every 2 hours
- Naval boarding reports (vessel type, length, engine power, etc.)
- Dates of vessel boarded, details of fines, etc.
- General data on orange roughy prices, weather, deep water coral distribution, etc.

2.6.3. Methods to be considered for the analysis

- Random Utility Model (RUM): nested logit model and/or random parameters logit model : evaluation of parametric probability density of fishers' location choice
- ISIS-Fish: use of above probabilities as input and simulate the consequences of effort reallocation on fisheries dynamics.

2.6.4. Main assumptions of approach

- There is some ecological link between orange roughy and deep water corals, i.e. the existence of deep water corals benefits to orange roughy population dynamics.
- Deep water coral closures will affect harvest level of orange roughy through effort reallocation.

2.6.5. Software and computer requirements

- Standard software (SAS, STATA) and ISIS-Fish (§2.1)
- No computer performance issues.

2.6.6. Target audience/ users

- Fisheries and marine scientists.
- Fishery managers.
- Relevant NGOs.

2.6.7. References

1. Annala, J.H., Sullivan, K.J., Mace, P.M., Smith, N.W.McL., Griffiths, P.R., Livingston, M.E., Harley, S.J., Key, J.M. & A.M. Connell. 2005. Report from the Fishery Assessment Plenary, May 2005: stock assessments and yield estimates. Ministry of Fisheries.
2. Kahui (née Schneider), V. 2006. A bioeconomic analysis of marine reserves for paua (abalone) management at Stewart Island, New Zealand. PhD Thesis, University of Otago, Dunedin, New Zealand.
3. Smith, M.D. & J.E. Wilen. 2003. Economic impacts of marine reserves: the importance of spatial behavior. *Journal of Environmental Economics and Management* **46** (2), 183-206.

2.7 Environmental valuation techniques

2.7.1. Main issues addressed

Because MPA have economic implications for the human activities in the area, decision makers must take into account the trade-off between the expected conservation effects and the cost implications of their implementation. In general, environmental valuation modelling techniques are being applied to resources and environmental attributes that do not command a market price, addressing the challenge of imputing monetary values there for. This applies to situations where,

The absence of market prices means that some other way of imputing monetary values has to be found, which in practice generally involves establishing people's preferences (reflected in their willingness to pay, WTP) for specified benefits derived from marine environmental assets. An extensive literature on environmental valuation now exists that best illustrates the valuation problem in the context of MPAs.

2.7.2. Model scales (space and time)

Depend on the sample size selection and it can expand from a small area to a wider area such as a country.

2.7.3. Main model features

Contingent valuation method (CVM) is widely used for eliciting respondent's preferences for unpriced benefits associated with marine environmental quality, especially the non-use values. Main model feature of the CVM study is that the estimation of willingness to pay function. The results provide a monetised measure of environment value. For further background on the approach and case specific aspects, please see **section 6.3**.

2.7.4. Main assumptions

There are no specific assumptions on either CE or CVM modelling exercise; however, both models are based on the general technique of multiple regression analysis. The general assumptions underlying regression analysis will therefore apply in modelling preferences.

2.7.5. Data requirements

In the North Sea sandeel case study, valuation of the kittiwake will be undertaken using CE and CVM, where the general public in Scotland will be interviewed to find out preferences for different characteristics of kittiwake. Simple random sample will be drawn from a list of household purchased from the Royal Mail, UK.

In the deep water coral reef case study, valuation of coral reefs protection benefits in Irish seas will rely on the CE approach based on the knowledge of its continued existence and availability to future generations. General public of Ireland will be interviewed to find out their preferences for different characteristics of the coral reefs. Sample will be drawn from the geographical customer databases available at the "An Post", Ireland.

2.7.6. Software and computer requirements

SAS software will be used in experimental design and analysis of the CE and CVM results.

2.7.7. Outputs

Stakeholder preference values for coral reef protection and birds can be used as an estimate of the total value of the resource. In cases where it is clearly possible to attach a market price to outputs, as in the case of commercially-traded fishery products, policy decisions can be made based on the criterion of efficiency. However, the marine resource from which production derives may itself be unpriced by virtue of its public-good characteristics; it is leading to inefficiency and market failure. Two environmental valuation techniques chosen in this analysis generally establishes people's preferences (WTP) for specified benefits derived from marine environmental assets (wildlife in adjacent coastal areas). The outputs of such methods provide a monetised measure that can be incorporated into broader socio-economic evaluations of the benefits and cost of deep water corals and sandeel area MPAs.

2.7.8. "Target" audience/ users

- Marine Scientists
- MPA managers
- Policy makers

2.7.9. References

1. Carson, R.T., Flores, N.E., and Meade, N.F. 2001. Contingent valuation: Controversies and evidence. *Environmental and Resource Economics* **19**, 173-210.
2. Wattage, P., Mardle, S. and Pascoe, S. 2005. Evaluation of the importance of fisheries management objectives using choice-experiments. *Ecological Economics* **55**, 85-95.

2.8 BEMCOM: BioEconomic Model to evaluate the COnsequences of Marine protected areas

In order to consider the economic consequences of establishing MPAs, BEMCOM is being developed. It is a flexible modelling framework, which can be utilised to investigate different management strategies with respect to marine protected areas (for technical details on BEMCOM, please refer to Annex 2).

2.8.1. Main issues addressed - Non technical description

The main issues addressed with BEMCOM are:

- Estimation of the economic consequences of setting up MPAs
- Fleet behaviour following from changes in management
- Comparing different management scenarios, and determine the 'optimal' management strategy in order to obtain the best economic outcome

Besides the main issues, the model development will most likely lead to the revelation of other interesting issues, which will be analysed including topics such as investment behaviour, consequences of different biological scenarios (worst/best case scenario).

2.8.2. Model scales (space and time)

BEMCOM is working with four initial dimensions. These are:

- Fleet segments based on vessel length and primary gear type
- Fishing area (ICES-squares or some more aggregated areas depending on the case study)
- Species
- Two time dimensions: i) monthly to integrate short/intermediate run the behaviour of fishermen with respect to choice of gear type and fishing location; and ii) yearly to integrate the behaviour of fishermen with respect to long run investment decisions

Furthermore, the biological component of BEMCOM may require inclusion of for instance cohorts reflecting the age of fish. However, this is still under development. Also, at least within case study 2, it is necessary to include a nationality dimension.

2.8.3. Main model features

The intention is to program BEMCOM in a generic way in order to be able to handle different case studies reflected through the utilised dataset and parameter values. In PROTECT, BEMCOM will be used in Case study 1 – Top-down controlled ecosystems in the Baltic Sea and in Case study 2 – Wasp-waist ecosystems (sandeel).

Contrary to many models which focus on single species (e.g. sandeel and cod), the main feature in the economic model takes into account the revenues generated by catching other species. This is necessary to have a complete picture of the economic effects for the fleets modelled. Fleets may thus have other activities during a year besides catching the two species, which are the focus of biological modelling. Furthermore, BEMCOM operates through adjusting the number of days at sea and number of vessels in order to obtain the best possible outcome under the policies considered.

2.8.4. Main assumptions

Basic equations in BEMCOM have been developed, but actual quantification of these has not yet been undertaken. Therefore, the specific areas where assumptions are needed in order to for instance account lacking input data have currently not been identified.

2.8.5. Data requirements

Biological component

In order to quantify the biological relationships within BEMCOM, it is necessary to obtain information about natural mortality levels, recruitment functions, dispersion relationships and initial stock level. Also, the effect of increasing stock size on catch per unit effort is required. These figures will be available through the biological studies undertaken within PROTECT.

Economic component

For some nations, the availability of economic data for the different fleets is limited. The collection of economic data as defined in the EU Data Collection Regulation (DCR) has not yet delivered information which can be used in this type of modelling. Therefore, this project must rely on the availability of individual vessel information for the countries, where this can be obtained. By using this information, it is possible to estimate revenue and cost structures for the different fleets that are relevant to consider in each case study.

Management component
In order to obtain reliable and feasible solutions in BEMCOM, it is necessary when defining the management scenarios to be investigated also to include plausible restrictions with respect to for instance the development in fleet size and TACs.

2.8.6. Software and computer requirements

Software
BEMCOM is programmed in GAMS (General Algebraic Modeling System), which is not a freely available software. See more at www.gams.com

User manual
The user manual is available in English. See www.gams.com

Computer
Former experience with GAMS modelling do not indicate any problems with respect to running BEMCOM on any recent computer..

2.8.7. Outputs

BEMCOM can be used either as a simulation model or as an optimisation model. The former answers questions such as what happens if management changes, while the latter answers questions such as what is the best type of management.

Several different scenarios can be analysed using BEMCOM, by changing the restrictions and parameter values, which is included in the model. For each scenario, classical outputs of bioeconomic models such as biomass, catch, costs and revenues will be calculated. Because of the very detailed modelling level in BEMCOM, it may for presentation purposes be necessary to make some aggregations. However, what and how much to aggregate is of course dependent on the questions that needs to be answered/discussed.

2.8.8. "Target" audience/ users

Running of BEMCOM is complex and is therefore considered mainly to be done by scientists. However, defining the scenarios and restrictions included in BEMCOM may be done in collaboration with managers, fishermen etc. Furthermore, the results from BEMCOM can be presented in simple overview tables to motivate further discussions about the design of marine protected areas.

2.8.9. References

No specific references can at the moment be given for a description a BEMCOM, besides a draft working paper currently under development. However, bioeconomic models have been utilised in several papers, for broad overviews see for instance:

1. Arnason, R. and V. Placenti, 1997. *Bio-economic Fisheries Computer Models: An Overview of Existing Models*. Research Report FAIR-CT95-0561. IREPA, Salerno, Italy.
2. Frost, H. and J. Kjærsgaard, 2003. *Numerical allocation problems and introduction to the Economic Management Model for Fisheries in Denmark: EMMFID*. Report no. 159. Danish Research Institute of Food Economics, Copenhagen, Denmark.
3. Pascoe, S. (Editor), 2000. *Bioeconomic modelling of the fisheries of the English Channel*. Final Report FAIR CT96-1993. Research Report no. 53. CEMARE, Portsmouth, United Kingdom.

2.9 HABFISH - a habitat-fishery model

2.9.1. Main issues addressed - Non technical description

- Evaluate how habitat-fisheries interactions affect management options in fisheries, e.g. how MPAs should be implemented.
- Main focus on a non-renewable habitat (such as deep water coral), which is "harvested" in connection with an economic activity, and how this affects other economic activities that are either biologically and/or directly economically affected by the habitat.
- HABFISH is a theoretic bioeconomic model (at this point in time), where we study dynamic and static behaviour.

2.9.2. Model scales (space and time)

- Infinite, continuous time
- Space is not explicitly modelled

2.9.3. Main model features

Populations
- Lumped parameter, either biomass or numbers
- Schaefer growth model

Exploitation
- Gordon-Schaefer harvest function, lumped harvest
- Optimal economic behaviour
- Bioeconomic formulation of optimal exploitation path or condition
- Results are evaluated for interdependencies between a renewable resource and a non-renewable resource (i.e. habitat) through carrying capacity and cost
- Evaluating the externality effect of one harvesting activity (e.g. trawling) on another fishing activity (e.g. long-lining)

2.9.4. Main assumptions

- Three different connections between habitat and fishery: 1) habitat affects fish growth positively, 2) habitat has a positive effect on harvesting costs, and 3) habitat affects both costs and growth. These are compared to the case where there is no connection between habitat and the fishery.
- Habitat is negatively affected by some fishing activity (e.g. trawling, but not long-lining etc). Other fishing activities (e.g. longlining) are positively affected by habitat.
- Optimal profit maximising management
- No investment dynamics for fleets, only short-term reallocation of fishing effort

2.9.5. Data requirements

- None (theoretical model)

2.9.6. Software and computer requirements

- None

2.9.7. Outputs

- Understanding of how management of fisheries is affected by interactions between habitat and fish, or habitat and fishing activity.
- Understanding of consequences of not taking into account habitat-fisheries interactions.
- Policy advice on the likely consequences of current harvest activities (e.g. trawling on deep sea corals) on other fishing activity (e.g. longlining).
- Policy implications of how best to address habitat degradation and what the trade-offs are between harvesting activity and conservation.

2.9.8. "Target" audience/ users

- Scientists
- Fisheries managers

2.9.9. References

1. Armstrong, C.W. A note on the ecological-economic modelling of marine reserves. Forthcoming in *Ecological Economics*.
2. Clark, C.W., 2005. *Mathematical Bioeconomics. Optimal Management of Renewable Resources*. Second Edition, John Wiley & Sons, Inc., Hoboken, New Jersey.
3. Clark, C.W. and G.R. Munro, 1975. The Economics of Fishing and Modern Capital Theory: A Simplified Approach. *Journal of Environmental Economics and Management* **2**, 92-106.
4. Hotelling, H., 1931. The Economics of Exhaustible Resources. *Journal of Political Economy* **39**, 137-175.
5. Swallow, S.K., 1990. Depletion of the Environmental Basis for Renewable Resources: The Economics of Interdependent Renewable and Nonrenewable Resources. *Journal of Environmental Economics and Management* **19**, 281-296.

2.10 Production Function Approach model

2.10.1. Main issues addressed - Non technical description

- Evaluate whether there is a connection between prevalence of deep-water coral and redfish.
- Bioeconomic model of redfish, with deep water coral entering in carrying capacity.
- Statistical analysis of correlation between deep-water coral depletion and redfish stock depletion.

2.10.2. Model scales (space and time)

- Infinite, continuous time (steady state)
- Space is not explicitly modelled

2.10.3. Main model features

Populations
- Lumped parameter, biomass or numbers

- Deep water coral affects carrying capacity of redfish
- Schaefer growth

Exploitation

- Gordon-Schaefer harvest function, lumped harvest
- Open access management

2.10.4. Main assumptions

- Biological growth function of the fishery is modified to allow for the influence of deep water coral
- *Lophelia* is a function of carrying capacity
- *Lophelia* has a positive impact on carrying capacity, K.

2.10.5. Data requirements

- Harvest data (landings)
- Effort data (days at sea or number of vessels)
- Deep water coral data – area coverage (e.g. in km^2), measurement of coral destroyed or damaged per year

2.10.6. Software and computer requirements

- STATA

2.10.7. Outputs

- Understanding of whether habitats like deep-water coral are in any way important for the redfish stocks
- Understanding of consequences of not taking into account such habitat-fisheries interactions.
- Policy advice on the likely consequences of current harvest activities (e.g. trawling on deep sea corals) on other fishing activity (e.g. longlining).
- Policy implications of how best to address habitat degradation and what the trade-offs are between harvesting activity and conservation.

2.10.9. Target audience/ users

- Scientists
- Fisheries managers

2.10.8. References

1. Armstrong, C.W. A note on the ecological-economic modelling of marine reserves. Forthcoming in *Ecological Economics*.
2. Barbier, E.B., 2000. Valuing the environment as input: a review of applications to mangrove-fishery linkages. *Ecological Economics*, **35**, 47-61.
3. Barbier, E.B. & I. Strand, 1998. Valuing mangrove-fishery linkages: a case study of Campeche, Mexico. *Environmental and Resource Economics*, **12**, 151-166.
4. Fossa, J.H., Mortensen, P.B. & Furevik, D.M., 2002. The deep water coral *Lophelia pertusa* in Norwegian waters: distribution and fishery impacts. *Hydrobiologia*, **471**, 1-12.

2.11 Baltic Sea circulation model and Lagrangian particle tracking

2.11.1. Main issues addressed – Model description

This hydrodynamic model is used in the project to identify seasonally-resolved locations of Baltic cod nursery areas between 1979 and 2004, and to quantify exchanges of larval drifters between basins.

The numerical simulations of the circulation are performed by application of a three-dimensional (3-D) eddy resolving baroclinic model of the Baltic Sea (Lehmann 1995). The Baltic Sea Model is based on the free surface Bryan-Cox-Semtner model (Killworth et al. 1991) which is a special version of the Cox numerical ocean general circulation model (Bryan 1969, Semtner 1974, Cox 1984).

2.11.2. Model scales (space and time)

- Horizontal 5km, vertical 3m
- Simulation model: 5 minutes time steps, Lagrangian particle tracking: 6 hourly time steps

2.11.3. Main model features

- Inter- and intra-annual variability of drift and distribution of cod early life stages
- Exchange between spawning and nursery areas and between basins

2.11.4. Main assumptions

- No mortality of cod early life stages
- Homogenous initial distributions of particles in different basins representing larval drift release areas

2.11.5. Data requirement

- Highly temporally and spatial resolved atmospheric forcing data (e.g. wind data, air temperature, cloudiness)
- River runoff data
- Location and timing of larval hatching areas

2.11.6. Software

Simulation models are coded in Fortran, no further specific requirements

2.11.7. Outputs

- Calculation of three-dimensionally resolved current and hydrographic property fields
- Calculation of tracked particle positions and corresponding physical parameters along tracks.

Simulated three-dimensional velocity fields are extracted at 6-hourly intervals in order to develop a database for this Lagrangian particle tracking exercise. The data set then offers the possibility to derive drift trajectories by calculating the advection of "marked" water particles. The positions of the drifters varied over time as a result of the current velocities that they experienced. Furthermore, the data contain information on the temporal evolution of the hydrographic property fields (temperature, salinity, oxygen, current velocity, etc.) along the trajectories.

2.11.8. References

1. Bryan, K. 1969. A numerical method for the study of the circulation of the world ocean. J. Phys. Oceanogr. 15: 1312-1324.
2. Cox, M.D. 1984. A primitive equation 3-dimensional model of the ocean. GFDL/Princeton University, GFDL Ocean Group Technical Report 1, 144pp.
3. Killworth, P.D., Stainforth, D., Webbs, D.J. and Paterson, S.M. 1991. The development of a free-surface Bryan-Cox-Semtner ocean model. J. Phys. Oceanogr. 21: 1333-1348.
4. Lehmann, A. 1995. A three-dimensional baroclinic eddy-resolving model of the Baltic Sea. Tellus, 47A: 1013-1031.
5. Semtner, A.J. 1974. A general circulation model for the World Ocean. UCLA Dept. of Meteorology Tech. Rep. No. 8, 99 pp.

2.12 Model of the population dynamics of the Eastern Baltic cod

This model is a dynamic population model for a single-species (Eastern Baltic cod), which is based on statistical analyses (multivariate regression analyses).

2.12.1. Main issues addressed

Evaluate the consequences on population and exploitation of different management scenarios (in particular permanent and seasonal MPAs) and the influence of environmental conditions (climate change) on their performance.

2.12.2. Model scales (space and time)

- Spatially-explicit (ICES subdivisions 25, 26, 28, in ICES area IIId)
- Temporal resolution: time-step = 3 months
- Theoretically, spatial and temporal scale adaptable to data resolution

2.12.3. Main model features

Populations

- Recruitment: function of spawning stock size and environmental conditions (reproductive volume RV)
- Predation mortality (= cannibalism): function of spawning stock size
- Seasonal migrations: estimations based on qualitative descriptions of migration: spawning migration: function of spawning stock size and RV, density-dependent feeding migration.

Exploitation

- Fishing mortality is exogenous and constant, i.e., imposed by the “policy maker”
- Rough calculation of revenues of harvesting, based on management scenarios and constant fish price

2.12.4. Main assumptions

- Interspecific interactions are not considered, only cannibalism
- Constant natural mortality (cannibalism-independent)
- No connection with the Western Baltic cod stock
- Fishers perfectly comply to management policy
- Only oxygen and salinity matter for recruitment

2.12.5. Data requirements

Populations

- Area-disaggregated data of (MS)VPA: stock size, mortalities (natural, fishing, predation mortality), recruitment)
- Environmental/hydrographic data: dissolved oxygen, salinity → combined into one variable, i.e., the reproductive volume
- Biological parameters: age of maturity or maturity ogive, weight-at-age and catch-at-age data

Exploitation

- Fish prices
- Cost and revenue data

Management

- Represented by different scenarios, parameterized through fishing mortality

2.12.6. Software and computer requirements

- Model currently available in GAMS
- Statistical software useful for regression analysis

2.12.7. Outputs

- Future stock sizes, catches, yield, and revenues and costs, depending on the different management policies and environmental scenarios
- Results can be visualised directly in GAMS via Gnuplotxy

2.12.8. References

1. Röckmann, C., U.A. Schneider, M.A. St.John, & R.S.J. Tol. 2005. Rebuilding the Eastern Baltic cod stock under environmental change - a preliminary approach using stock, environmental, and management constraints, FNU-84, Hamburg University and Centre for Marine and Atmospheric Science, Hamburg.
2. Röckmann, C., M.A. St.John, U.A. Schneider, F.W. Köster, & R.S.J. Tol. 2006. Testing the implications of a permanent or seasonal marine reserve on the population dynamics of Eastern Baltic cod under varying environmental conditions, FNU-63-revised, Hamburg University and Centre for Marine and Atmospheric Science, Hamburg.

2.13. TEMAS

TEMAS (Technical Management Measures) is an Evaluation Frame based on a spatially and seasonally-explicit operational model. TEMAS is aimed at policy evaluation, in particular technical management measures.(Ulrich *et al*, 2002, Sparre, 2003, TECTAC, 2005, Ulrich *et al*, 2007). MPA is in this context considered a technical management measure. The TEMAS operational model also includes an economic description of the fishery, and a description of the behaviour of fishers, in particular the reaction of fishers to technical management measures.

2.13.1 Main issues addressed – Non technical description.

The overall content of TEMAS is illustrated by the data-flowchart in Figure 2.13.1. The system compares two management regimes, A and B, by simulating the fisheries system over a series of years for both regimes, and eventually it compares the performance of the two regimes during the time period. Thus the figure illustrates a dynamic system, where the arrows indicate the processes of one single time period (month, quarter or year).

The "operating system" (Figure) is a model simulation of the eco-system and the fisheries system. The boxes "Management regime A" and "Management regime B" indicates two models which can simulate the management processes (which may include simulation of ICES WG, setting of TACs, etc.).

The operating system generates ("fake" or "hypothetical") input data to the management models, and it predicts the effect of the management regulations on the eco-system and the fisheries.

Thus, one may consider TEMAS as a triple, model. Firstly, it executes the simulation of management regime A, using the operational model to produce input to the management simulation. Secondly, it does the same of management regime B, and thirdly it compares the two simulations.

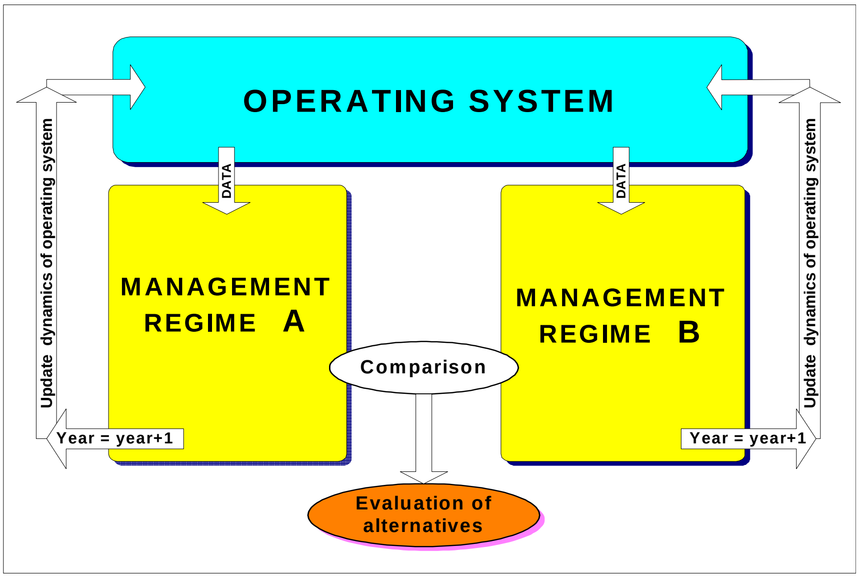

Figure 2.13.1 The principal components of TEMAS for one time period of a dynamic process.

In the context of evaluation of MPAs and closed seasons, the alternative management regimes could be: (1) the current management regime with no closed seasons and MPAs. (The current regime could be Eg. TAC and maximum number of sea days, mesh size regulations etc.) (2) The current management regime with closed seasons and MPAs

The TEMAS operational model (OP) is not a particular model, but is a tool box of (mainly traditional) models, which allows the building of user defined applications. The core of the TEMAS operational model is the ICES forecast model (or the Thompson & Bell model, 1934). The user may add various components to the core, either components already available in the tool box or the user may develop case specific new components. This is the case with the Baltic cod case study, where a special stock recruitment component has been added to the existing tool box.

The operational model is the same in all regime comparisons. The operational model simulates fish stocks, fishing fleets etc. and from the quantities it simulates input data to the pair of management models.

The TEMAS model can do single deterministic simulations or multiple stochastic simulations. The multiple stochastic simulations executes a number of single deterministic simulations (say 1000 simulations), each of which based on parameters drawn by a random number generator.

2.13.2. TEMAS OP scales (space and time)

- Spatially-explicit with a maximum of about 10 different areas (box-model).
- Optional time-step (year, quarter or month).

2.13.3. Main TEMAS OP features

Populations

- Growth, modelled by the von Bertalanffy model, and maturity by the logistic model. The growth and population dynamics of juveniles is described in more details, that that of adults, as the objective of technical management measures is to protect juveniles. Growth of individuals can be made s stochastic process.

- Seasonal migrations of adults between spawning grounds and feeding grounds. Migration of juveniles out of spawning grounds.

- Migration is based on the model by Quinn *et al.* (1990). Migration is time discrete, as instantaneous "jumps" at the end of a time period.

- Optional stochastic/deterministic relationship between spawning stock biomass and recruitment. Account is made for occasional years with a large "reproductive volume" in the stochastic model.

- Species interaction (predation/food competition) not covered by model.

Exploitation

- Fleets are divided into vessel size groups and vessel age groups (optional). These groups in turn are divided into "gear riggings" (or fisheries or metiers). The distribution of effort on riggings can be given as fixed input, or be determined by the model of fisher's behaviour, the RUM, Random Utility Model *Wilen. Et al,* (2002). That is, the reallocation of effort after introduction of MPA can be modelled, in terms of choice of fishing ground and choice of gear rigging. The choice of fishermen is based on the distribution of resources outside the MPA and the expected revenue. Thus, the model accounts for costs and earnings of fishing operations.

- The long term behaviour of fishers (the investment/dis-investment) is also modelled by the RUM. The model can account for decommission programs.

- Catches are split into discards and landings.

- Gear selection and discarding is modelled by the logistic model. This model can account for the introduction of escape windows, such as the BACOMA trawl.

2.13.4. Main assumptions of TEMAS OP

- No biological interaction (predation or food competition).

- Migration of fish is modelled in a time discrete manner. No migration takes place during a time period, and fish are assumed to be evenly distributed within an area. Migration takes zero time, and is allocated to the end of a time period. TEMAS can handle only a small number of areas (maximum 10 areas).

- Effort is assumed uniformly distributed within areas, and as migration of fish is time discrete, reallocation of effort is time discrete.

- Fishing mortality is assumed to be a function of effort only. In its simples form, effort is assumed to be proportional to effort. Effort is derived from the number of vessels combined with their activity levels.

- Errors are assumed to occur on several components of the model:
    - Measurement error. Errors in input data, such as catch at age data, caused by data being estimated from samples, and not from complete enumeration.
    - Estimation error. Errors caused by the method used to estimate parameters, or erroneous assumption about the data.
    - Model misspecification error. Errors caused by incomplete or wrong understandings of the mechanism behind the system dynamics. The assumed Stock/recruitment relationships may be candidates for model misspecifications.
    - Implementation error. The errors caused by regulations not being reacted to as assumed. The fishers may find ways to implement regulations, which do not lead to the achievements of the intensions of regulations.

- The model simulates the effect of errors and bias, by stochastic simulations.

2.13.5. Data requirements

- The data requirements depend on level of conditioning selected by the user. In theory, the model can be run without any input, if no conditioning is made, but that is not recommended.
- The conditioning is to make a “prediction” for a historical period, and then to compare the model predictions to the corresponding observations, e.g. landings either in weight or in number by age group. Many parameters are assumed to be estimated from external sources, e.g. from the ICES WG reports.
- Depending on the available data, as many input parameters as possible will be extracted from independent sources, and as few as possible estimated within the model.

Management

- The evaluation frame will simulate data collection, data processing, assessment by ICES WG and advice by ACFM. Subsequently, introduction of additional technical management measures by the STECF is simulated. This may be MPA, maximum number of sea days (effort based management) and gear regulations.

- The data collection can be simulated with various types of errors and bias, such as misreporting. Also bias in age reading of fish can be simulated.

- Lack of compliance with regulations can be simulated.

2.13.6. Software and computer requirements

- Software: Implemented in EXCEL/VISUAL BASIC. Apart from Microsoft Office, software is free and available on request to DIFRES.

- User's manual is available.

- Computer: Runs on any PC with MS Office and EXCEL 2003 or later.

2.13.7. Outputs

- Comparisons of two (or more) alternative management strategies, e.g. management with MPA and management without MPA. Both MPA measures can be combined with other technical measures.

- Alternative strategies are compared by "measures of performance", which may be biological as well as bio-economic measures. Measures of performance can be stakeholder specific, so that the performance can be evaluated in different ways, dependent on the priorities of the stakeholder.

- Output is in the form of tables and graphs in EXCEL worksheets, which can further processed with EXCEL functions.

- The Visual Basic code, of EXCEL macros. The Visual Basic code is heavily commented, to facilitate the understanding.

2.13.8. Target audience/ users

- Scientists
- MPA managers with the help of scientists
- Fishers and industry with the help of scientists

2.13.9. References

1. Quinn, T.J.II, R.B. Deriso and P.R. Neal.1990, Migratory Catch-Age Analysis. Con.J.Fish.Aquat.Sci, (47):2315-2327.
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8. Ulrich, C., Andersen, B. S., Sparre, P. J., Nielsen, J. R. 2007. TEMAS: A fleet-based bioeconomic simulation software for management strategies accounting for fishermen behaviour. – ICES Journal of Marine Science, 62: (in press)

2.14. IBM Model - SIMPL (Spatial, Individual MultisPecies model)

SIMPL (**S**patial, **I**ndividual **M**ultis**P**ecies mode**L**) is a spatially explicit individual-based-model (IBM) which tracks the feeding, growth, migration, reproduction and mortality of individual fish. The model has been constructed to test the impacts of various fishery-management options, including spatial closures in the North Sea, and in particular on the western edge of the Dogger Bank.

2.14.1 Main issues addressed – Non technical description

Traditional multispecies fisheries models within the North Sea (e.g. MSVPA) have not been spatially explicit and have therefore been of limited utility for testing the effect of MPAs on predators and their prey. The SIMPL model was parameterised for sandeels and their predators using dedicated field data as well as historical information. The range of management scenarios tested includes permanent closures, rotational closures, closed seasons as well as more traditional TAC management.

The model is highly flexible and can be set-up to replicate simple systems with one species living in one spatial location, ranging to multiple species, locations and fishing fleets. Individual fish exist in ‘shoals’ and the shoals interact at feeding time. Feeding itself is governed by the size ratio of the predator and potential prey, the diet composition of the predator and the predator’s maximum energy intake. Growth is implemented through either energetic functions (thus permitting “bottom up” food web control) or von Bertalanffy equations (“top-down” control). Migration is controlled by Levy-flight functions and fishing is implemented at the individual vessel level, on a swept-area basis. Each vessel carries certain gears with associated selectivity parameters.

2.14.2 Model Scales (space and time)

The temporal and spatial scales of the model have the potential to be fully flexible and can therefore operate on an annual scale and a single area, right down to daily time steps and thousands of patches.

- The model has currently been implemented at weekly time steps (52 time steps per year).
- As currently implemented the model includes 7 distinct ‘patches’ of sandeel habitat on the western edge of the Dogger Bank (F**igure 2.14.1**) as well as 1 ‘outside’ patch.

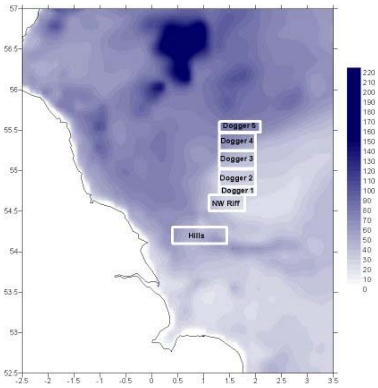


Figure 2.14.1. The 7 ‘patches’ currently implemented in the IBM model of the western Dogger Bank.

2.14.3 Main Model Features

As the name suggests, the model tracks individual fish as they eat, migrate, grow and, ultimately, die. This type of model is significantly more flexible than array based models and can mimic nature more closely, however this flexibility and realism comes at the expense of computational time which is vastly increased. Within any given time-step the model implements the following actions in sequence: (1) feeding, (2) growth, (3) migration and (4) fishing.

Habitat

- Although individual fish know their location in physical space, the environmental conditions for that space are held within the ‘patch’ object.
- Such conditions include depth, temperature etc. ‘Patches’ are also the spatial scale at which fishing and feeding occurs within each time-step of the model.

Feeding

- Each species carries a list of prey types thus enabling the model to assess a basic level of selectivity and restrict the dietary range of predators to that been observed in the field data.
- As each predator encounters a new prey object the basic decision process is as follows.
    - Am I still hungry?
    - Do I eat objects of this species type?
    - Is it the right size for me?
- The size function has been parameterised from field data and assumes that a predator of given size will have a range of preferred prey sizes.
- Feeding ceases once the predator is full. This is controlled by the maximum energy requirements of the fish.

Growth, Migration & Spawning

- Growth has been implemented as a bioenergetic function. Whenever a predator eats a fish, the energy content of that fish is passed to the predator. The energy requirements for basic metabolism and locomotion are deducted from the ingested energy, any energy left over is then put into somatic growth.
- Migration has been implemented as a function of Levy flight. This describes the probability of an individual moving a number of body lengths in a given time period.
- Fish can group together to form shoals of unlimited size. This allows the exploration of different shoaling behaviours that can potentially affect the feeding and fishing functions.
- Individuals spawn a number of recruits each year, in relation to their body mass and maturity. Recruits enter the model at 6 months, in the ‘patch’ where they were spawned.

Exploitation

- Fishing vessels are specified with size, operational speed, type and size of fishing gear.
- Fishing gears are specified with a mesh size so that the probability of capturing individual fish can be determined.
- Fishing works on the swept area basis. Within a time-step, a fishing vessel will operate for a number of hours (determined from satellite data), towing gear with a known door-spread.
- For each time-period within the model a vessel will fill in a logbook object which records where it has been fishing and the numbers and tonnage of fish caught by species.
- The model is currently set up to include 10 fishing vessels in the Danish sandeel fleet.

2.14.4 Main Assumptions

It is not anticipated that the full range of potential food items will be modelled, indeed for generalist predators such as cod it is simply impractical to model all individuals of all potential prey types. The model therefore utilises “otherfood” to generate “fish” objects to represent the biomass of all other potential prey. Fish of type “otherfood” are different in that they do not die and they do not migrate, i.e. the availability of ‘otherfood’ is constant throughout the time period.

2.14.5 Data Requirements

Data from field-work on the Dogger Bank (UK-Defra funded research programme M0323), in addition to published information on sandeel growth and the migration of predators (from tagging studies) has been used to parameterise and to validate the model outputs (see table). Information form VMS satellite monitoring of fishing vessels was also utilised.

Used for Parameterisation	**Used for Validation**
Predator prey links (who eats who, how much) [from UK - M0323 project]	Population abundance and size structure in heavily fished and lightly fished areas [from UK - M0323 project]
Prey size preferences of predators [from UK - M0323 project]	Prey quality impact on predators – do the predators have a higher condition index where sandeels are more abundant? [from UK - M0323 project]
Sandeel population parameters (bioenergetics data from EU-funded CORMA project)	
Vessel speeds determined from analysis of satellite tracking data from sandeel trawlers.	
Behaviour and movement rules for sandeels and their predators [based on tagging information]	
Spatial pattern in the aggregation of sandeel schools [from UK - M0323 project]	

2.14.6 Software & Computer Requirements

The model has been constructed using C++, an object orientated programming (OOP) language. OOP is ideal for individual based modelling as each individual is held in computer memory as a separate object. The evaluation of each management scenario takes around 35 hours for the model to run.

2.14.7 Outputs

- Fates of all individual fish (a logbook of why, where and when they die)
- Status of all individuals (e.g. a specified point in time)
- Fishery landings by ‘patch’, per year (summary statistics)

2.14.8 Target Audience/users

- Envisaged as a tool for exploring the management of sandeel fisheries
- Modelling work originally commissioned by the UK Department for Environment Food & Rural Affairs (Defra).
- The Dogger Bank has recently been proposed (by Germany, Netherlands and the UK) as a possible site for offshore SACs under the EU Habitats Directive. As such an evaluation of potential ecosystem and fishery effects of MPAs in this region would seem particularly useful.

2.14.9 References

Mackinson, S. et al. (2007) Multi-species fisheries management: a comprehensive impact assessment of the sandeel fishery along the English east coast. Final report on project M0323/01 to the Department for Environment, Food & Rural Affairs (Defra), March 2007.

2.15 The IBM model “PROTECT” for plaice and cod, North Sea

2.15.1. Main issues addressed - Non technical description

PROTECT is an individual based simulation model: Once a year fish hatches at different spawning area’s, migrates to different preferred area’s and then return to their spawning ground each year. Each fish has a probability to die due caused either by natural causes or by being caught. By defining various different MPAs one can vary the fishing effort and observe the impact on the population and yield.

2.15.2. Model scales (space and time)

The area used in the model is the whole North. It uses a grid size of about 10 by 10 nautical miles^2. For each cell the average depth has been. The MPAs are constructed by combining a number of cells at various locations.

The time step can be varied and normally is about one week (1/52 part of a year).

2.15.3. Main model features

Populations

- Fish in the model lives until it has been caught or dies otherwise.
- Each fish has a yearly migration pattern based on its age dependent preferred depth and spawning ground.
- Larvae migrate to the nursery areas.
- Fish younger than 3 years remain in the vicinity of nursery area’s and don’t’ migrate to their spawning ground.
- Size of the number of hatched larvae is a constant.
- Growth is modelled using the observed weights of the landings and surveys for the species (*Pleuronectes platessa*) over the period 2001-2006. The natural mortality is set to 0.1.
- Probability for a fish to be caught is based on the mean number of observed fishing in each cell over the period 2001-2005.

Exploitation

Fishing mortality is based on the Dutch fleet of beamtrawlers.

2.15.4. Main assumptions (of present version 1.0)

- There is no relation to other species.
- Every fish has a preferred depth range to which it migrates or where it remains when not homing to its spawning ground.
- Temporal fishing effort doesn’t change.

2.15.5. Data requirements

Populations

- Preferred depth for different ages is based on the observed distribution of the catches over the area.
- Locations of the spawning grounds is given by the observations.
- Growth of the species is based on the observed weights at age.

Exploitation

The exploitation is given by the average number of observed fishing hours in a cell.

Management

The different scenarios for having MPAs can be varied. Currently the following are used:

- No MPA at all.
- Current MPA's (Plaice box and Wind farms)
- Idem plus proposed MPA.
- Quarter of the area used as an MPA
- Whole area used as an MPA.

2.15.6. Software and computer requirements

- Software : using on the SWARM modelling environment: free software, no requirements, runs from Objective C. Everything is freely downloadable.
- The user's manual is available on the web site in English, but not very clear in all aspects.
- Computer : runs on Linux, Windows 2000/XP (using CygWin) and MacOS X.

2.15.7 Outputs

- Simulation designs can be run to test a range of model assumptions and run sensitivity analysis on model parameters.
- Results visualized in the software through an interface: depth, effort, location of the fish, age of the fish, yield, biomass, model parameters.
- Results exported in text format: for each timestep: yield, biomass, number of fish alive and caught.
- SWARM can export data to be used directly by R (no need to activate this yet).

4.15.8 Target audience/ users

- Scientists
- MPA managers with the help of scientists
- Can be used for discussions about MPA designs (either before MPA implementation, or after to evaluate if changes are needed)

2.15.9. References

Anon. (2000), Swarm User Guide, http://www.swarm.org/swarmdocs-2.1.1/userbook/userbook.html
Brody,S., (1945) Biogenetic and growth. Reinhold Publishing Corp., New York, N.Y. 1023 p.

3. Overview of PROTECT modelling components

3.1 Overview of modelling activities:

An overview of modelling actions to be achieved during the project is outlined in Table 3.1.

Table 3.1

		Baltic cod	**Sandeel**	**Coral reefs**
Empirical approaches / Statistical models	**Ecosystem and population indicators**	-Statistical analysis of CPUE and abundance series	-Statistical analysis of bird breeding rates -Statistical analysis of fish predator distribution and performance -Statistical analysis of CPUE series	
	Economic indicators Effort indicators	Partial Neural Networks Random Utility models	Contingent valuation	-Random Utility models -Contingent valuation
Dynamic modelling	**Ecosystem models**		IBM, sandeel IBM, Plaice	
	Population / Fishery models	ISIS-Fish TEMAS	SLAM+SPAM	ISIS-Fish
	Bioeconomic models	BEMCOM TEMAS	BEMCOM	HABFISH, PFA

Several models used in the project provide input parameters for other models which are directly used for evaluating MPA performance, as listed in Table 3.2.

Table 3.2

Model	**Output provided**	**Case study**
Hydrodynamic model	Delineation of nursery areas Estimation of larval dispersion Estimation of larval survival rates	Baltic cod
	North Sea wide distribution of larval drifters and their feeding conditions (ECOSMO)	NS Sandeel
	Western North Sea detailed drift pattern of larvae (HAMSOM new development)	Firth of Forth sandeel
Multispecies model (MSVPA)	Estimation of predation mortalities	Baltic cod
Correlation studies	Link between habitat quality and fisheries effort	Coral reefs
	Link between environmental variables and plankton, and survival rates of young stages	Baltic cod
	Relationship between spawners and egg production: link between maturity-fecundity and size-age-area and other effects	NS Sandeel

3.2 Interactions between modelling components:

Each case study is seeking to establish connections between modelling actions through a carefully planned work agenda. The work includes the following components and linkages:

Case Study 1 - Baltic Sea, cod

- Input fisheries data are common to BEMCOM, TEMAS and ISIS. Models operate at different scales and with distinct assumptions, but as far as possible, the types of vessels, fishing activities, gears, should be the same or compatible so that results may be compared across models.
- Scenarios tested should be the same, but are likely to slightly differ in details.
- ISIS focuses on the integration of biological knowledge and on the evaluation of MPA scenarios under a range of environmental conditions. The model will be retrospectively validated with respect to catch and effort data. In a first step, simulated catch and effort may simply be compared to catch and effort time-series.
- TEMAS focuses on the fleet response to MPA implementation and will consider economic parameters
- BEMCOM in addition considers MPA scenarios in a longer term perspective, accounting for the investment dynamics.
- Results from the RUM and PNN models are being integrated in each of the three models. In ISIS-Fish, they may be used to parameterize fishing effort allocation, or to fisher's behaviour.

Case study 2 - North Sea, sandeel, seabirds

- Hydrological output from ECOSMO is used for 3D currents, turbulence, salinity and temperature fields in SLAM
- Transport matrix from SLAM is base line for scaling SPAM recruitment part
- FRS maturity and fecundity relationships are input to SPAM
- IBM model on sandeel predation and used to generate predation rates scaled by IBTS data and used in SPAM
- Outputs from SLAM and SPAM will be used as inputs in BEMCOM
- Local bio-physical coupled model of sandeel drift is used to generate recruitment and sandeel availability for modelling of sandeel-seabird interlinkages.
- Contingent Valuation Models (CVM) of willingness to pay (WTP) surveys provide outputs for BEMCOM in that they will (ideally) help to define a functional form of valuation depending on protection level and on sandeel and seabird attributes.

Case Study 3 - Coldwater Coral reefs

- Contingent Valuation Models (CVM) of willingness to pay (WTP) surveys provide outputs for BEMCOM in assisting to define a functional form of valuation depending on protection level and on coral attributes

- Outputs from RUM models used to model fishing effort reallocation in ISIS-Fish.

- Population demographic rates obtained from the population dynamic model with ISIS-Fish may be used to build hypotheses about population renewal in the theoretical model HABFISH.

- HABFISH, a habitat-fishery model to evaluate how habitat-fisheries interactions affect management options in fisheries, e.g. how MPAs should be implemented.

- A 'Production Function Approach' model to evaluate whether there is a connection between prevalence of deep-water coral and redfish

3.3 Overview of ongoing PROTECT modelling development and outputs:

- Identification and evaluation of size and effective contribution of spawning areas. The study quantifies the spatial environmental heterogeneity of the Baltic cod spawning habitat in the Bornholm Basin. Station-based averaged environmental variables are calculated, describing the spawning habitat quality in order to identify longer-term spatial differences. It presents horizontal property fields associated with stagnation contrasted to more favourable spawning conditions as a result of inflow events into the Baltic Sea. See paper in press in **Section 2 – Annex 1.**

- A dynamic population model of the Eastern Baltic cod based on multivariate regression analyses to evaluate the consequences for populations and resource exploitation using different management scenarios (in particular permanent and seasonal MPAs) and the influence of environmental conditions (climate change) on their performance. See **Section 2 – Annex 2.**

- A hydrodynamic Baltic Sea circulation model and Lagrangian particle tracking used to identify seasonally-resolved locations of Baltic cod nursery areas, and to quantify exchanges of larval drifters between basins, and the implications for area-based management measures. See **Section 2 – 4.3.**

- A population dynamic modelling study to pilot future evaluations of the potential of MPAs to increase the size of the eastern Baltic cod stock under different management and environmental scenarios. The model is in essence a simulation model whose output (cod stock size) is randomized over different environmental scenarios and management options. See **Section 2 – 4.5**

- An Evaluation Frame based on a spatially and seasonally-explicit operational model. TEMAS is aimed at policy evaluation, in particular technical management measures, incl. MPAs. It includes an economic description of the fishery, and a description of the behaviour of fishers, in particular the reaction of fishers to technical management measures. See **Section 2 – Annex 3.**

- ISIS Fish–model for Baltic Cod. A new spatially and seasonally-explicit fisheries simulation model aimed at policy evaluation, in particular MPA, has been parameterised for model runs covering the Central Baltic. During the reporting period (i) the large amount of biological knowledge available about the cod population, and (ii) existing spatially-disaggregated fisheries data have been compiled and analyzed. An initial, basic parameterisation of the population model within ISIS Fish has been done. The parameterisation of the exploitation module is ongoing. See **Section 2 – 4.6.**

- A bioeconomic model is being developed to consider economic consequences of establishing MPAs. It is a flexible modelling framework to investigate different management strategies with respect to marine protected areas. Work on North Sea sandeel is progressing well, with future work on Baltic cod to follow. See **Section 2 – 4.8.**

- Statistical models for describing fleet movements in the Baltic Sea cod, sprat and herring fisheries; including identification of MPA-induced compensation actions of fishers, important factors affecting fishers' location choice, and socio-economic implications of MPAs enforced in the Central Baltic Sea. See **Section 2 – 4.9**

- A Sandeel Larval Advection Model aimed at identifying the consequences on sandeel recruitment dynamics of a stochastic hydrographical setup, particularly given a dynamic mosaic of fishing banks related to development of systems of MPAs in the North Sea. See **Section 2 – 5.2** and paper in press in S**ection 2 – Annex 5.**

- A Sandeel Population Analysis Model to address population dynamics interacting with an economic model of fishery, and spatially and temporally resolved fisheries issues in the North Sea. See **Section 2 – 5.3.**

- Bio-physical coupled hydrodynamic sandeel drift model used to simulate the relative survival of the spawning products of the North Sea (Scottish east coast) sandeel substocks for modelling of sandeel-seabird interlinkages. See **Section 2 – 5.4.**

- A spatially explicit individual-based model of sandeel and predator interactions in the North Sea, which tracks the feeding, growth, migration, reproduction and mortality of individual fish. The model has been constructed to test the impacts of various fisheries management options, including spatial closures in the North Sea, and in particular on the western edge of the Dogger Bank. See **Section 2 – 5.5.**

- An individual based simulation model for plaice and cod, North Sea. PROTECT is an individual based simulation model: Every year, fish hatch in different spawning areas, migrate to different preferred areas and then return to their spawning ground. Each fish has a probability to die from either natural causes or being caught. Through defining various MPA sizes one can vary the fishing effort in a given area and observe the impact on the population and yield. See **Section 2 – 5.8.**

- A study on the variation in the abundance of sandeels *Ammodytes marinus* off southeast Scotland: an evaluation of area-closure fisheries management and stock abundance assessment methods. See published paper in **Section 2 – Annex 6.**

- Statistical models to evaluate changes in seabird foraging ecology and demographic performance linked to the presence/absence of a sandeel fishery. Data from seabird (kittiwake) colonies along the UK North Sea coast, inside and outside the Firth of Forth sandeel fishery closure zone, is analysed to assess effects of fishery and/or the closures on breeding seabirds, incl. black-legged kittiwakes. See comprehensive report in **Section 2 – Annex 7.**

- A Production Function Approach model to evaluate whether there is a connection between prevalence of deep-water coral and redfish; the consequences of not taking into account such possible habitat-fisheries interactions; and to provide policy advice on the likely consequences of current harvest activities (e.g. trawling on deep sea corals) on other fishing activity, and policy implications of how best to address habitat degradation and what the trade-offs are between harvesting activity and conservation. See **Section 2 – 6.1.**

- A model to evaluate how habitat-fisheries interactions affect management options in fisheries, e.g. how MPAs should be implemented. Main focus on a non-renewable habitat (such as deep water coral), which is "harvested" in connection with an economic activity, and how this affects other economic activities that are either biologically and/or directly economically affected by the habitat. See **Section 2 – 6.1**

- Statistical modelling and simulation of fishing effort redistribution in relation to deep-water coral closures, including evaluations of biological effect of deep water closures on a commercial species, and how closures affect harvest displacement and levels in the short and long term. See **Section 2 – 6.1.**

- Environmental valuation techniques to assist decision makers in taking account of trade-offs between conservation effects and the cost implications of their implementation. Case studies include Nord Sea sandeel-seabird (kittiwake) linkages and protection benefits of Irish Sea coldwater corals. Theory and draft questionnaires are presented in the report. See **Section 2 – 6.3** and **Section 2 – Annex 9.** Increased activity in this component is expected in the second reporting period.

- A study to provide an overview of the bioeconomic modelling of marine reserves and to illustrate how economists have responded to the modelling results found in the ecological literature. See published paper in **Section 2 – Annex 8.**

For overview-diagrams of the different modelling components and linkages, please see **Section 2 – 4.1** and **5.1.** Overall, the modelling work is progressing well. It represents an ambitious undertaking and is expected to present a series of useful tools, as well as generic guidance and recommendations on future tool developments to evaluate effects of different types of MPAs. During the second reporting period, the modelling outputs will be integrated and synthesised in collaboration with WP6.

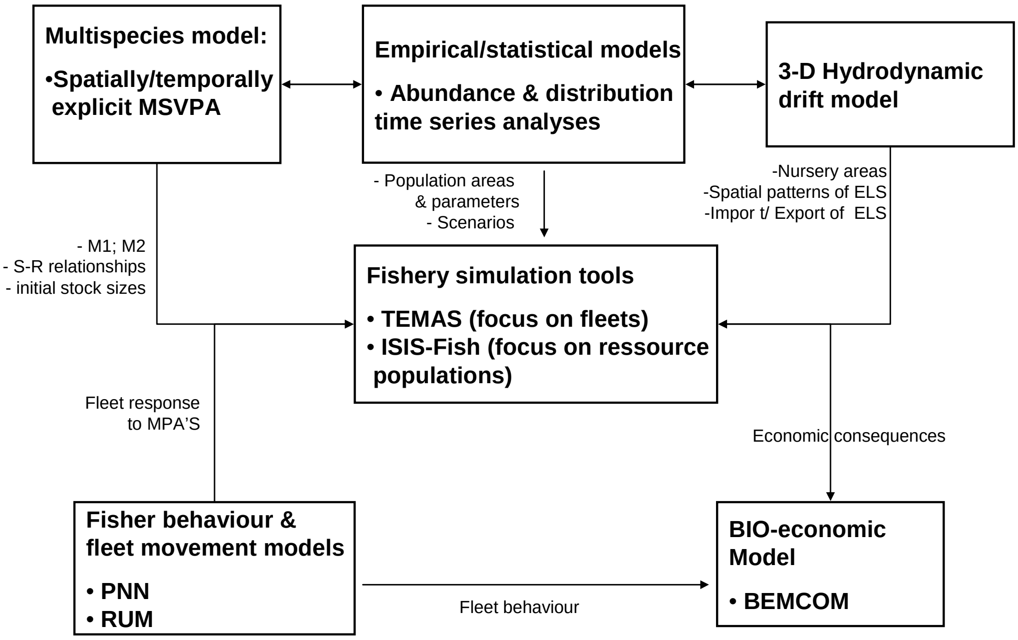
4. Progress in modelling work

Progress in modelling actions is reflected in the presentations made during the modelling session at the second thematic workshop in Hamburg, 14-15th Nov. 2006 (full details are provided in the workshop report and presentations are available at the internal project website)

Due to various degrees of timing of work the progress reports for each modelling action differ in length. According the workplan, some modelling actions are not being implemented yet and therefore no report is provided at this stage. For full details on models used please refer to Section 2.

4.1 Baltic Sea Case study - overview of modelling linkages

The Baltic Sea CS (1) has shown progress in several aspects of the modelling package of the project. First of all the flow of information and interfacing of model components has been established. The diagram below shows the major model components with their most important links.



Empirical approaches/statistical models are used to provide parameterisations and scenarios for the population models (MSVPA; populations models within TEMAS, ISIS and BEMCOM) and to test hypotheses about MPA effects, such as spatial differences in biological responses, but also to evidence effects of MPA on uses of marine resources and ecosystems, such as reallocation of fishing effort after MPA implementation. They may provide ecosystem indicators, community metrics, population-level indicators, probability distribution of fishing effort, and valuation of ecosystem services.

An area disaggregated Multi-Species VPA is used to provide initial area specific stock sizes for population simulations as well stock recruit data and natural mortalities for the ISIS model. A 3-D hydrodynamic model is used to identify areas that allow for successful cod spawning and to identify potential nursery areas, which serve as input to ISIS. On the Fisheries side two different approaches (parametric: RUM, non-parametric: PNN) are applied to simulate fisher behaviour and fleet movements used in the evaluation of MPA consequences within TEMAS and ISIS. In ISIS-Fish, they may be used to parameterize fishing effort allocation, or fisher's behaviour, whereas TEMAS will focus on the fleet response to MPA implementation and will consider economic parameters. The bio-economic BEMCOM will in addition consider economic consequences of MPA scenarios in a longer term perspective.

4.2 Identification and evaluation of size and effective contribution of spawning areas

H.-H. Hinrichsen, R. Voss, K. Wieland, F. Köster, K. H. Andersen, P. Margonski

Introduction

A prerequisite for the identification of spawning grounds as effective MPAs is to resolve egg stage distributions in response to the adult distribution and environmental conditions. For Baltic cod the locations of the earliest egg stage hardly varies from the distribution of the adults that are spawning, i.e. the horizontal distribution of newly spawned eggs provides a measure for the habitat selection of adult cod.

Historically, there have been three main spawning areas for the central Baltic cod stock: the Bornholm Basin, the Gdansk Deep, and the Gotland Basin (Fig. 1.1). Recent analyses of spatial and temporal heterogeneity of reproductive volumes (salinity > 11 psu, oxygen > 2 ml/l, temperature > 1.5 °C; hereafter RV) have recognized that conditions for successful cod-egg development are most likely to be found in the Bornholm Basin (MacKenzie et al. 2000). However, exceptions to this broad pattern do occur. During the longest recorded period without a major Baltic inflow of North Sea water (1977 – 1993) RV in the Bornholm Basin became very low. On the other hand, oxygen renewal events (e.g. inflows) can rapidly and significantly improve conditions in all basins (Matthäus and Lass 1995). However, the most recent years showed that the direct effect of a single water inflow is limited since i) unfavourable hydrographic conditions may already return in the following year, i.e. the improvement for cod is restricted to one spawning season only, or ii) inflowing oxygenated water replaces only lower parts of the bottom water resulting in an intermediate water layer with sufficient salinity to keep cod eggs floating, but not sustaining their development due to low oxygen concentration..

The present study describes the spatial environmental heterogeneity of the Baltic cod spawning habitat in the Bornholm Basin. Station-based averaged environmental variables are calculated describing the spawning habitat quality in order to identify longer-term spatial differences. It presents horizontal property fields associated with stagnation contrasted to more favourable spawning conditions strongly affected by inflow events. The approaches are empirically based, and involve besides environmental variables also egg abundance data. Finally, the approach attempts to characterize spatial and temporal variability in an eastern Baltic cod spawning areas and its habitat related environmental parameters from 1989 to 2003 in the light of the applicability of closed areas for fisheries to ensure undisturbed spawning as enforced until 2003 (ICES, 2004/ACFM:17).

The study is presented in its entirety in **Annex 1**.

References:

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4.3. Hydrodynamic modelling to describe transport and resolve physical environments

Hans-Harald Hinrichsen, IfM-GEOMAR

4.3.1. Introduction

A prerequisite for the implementation of cod nursery grounds as effective MPAs is to resolve the spatial and temporal dynamics of juvenile cod distribution. The primary aim of this study is to examine the influence of physical factors on the spatial distribution of early life stages of Baltic cod. In order to better understand the effects of physical forcing on the distribution of early life stages of cod, physical modelling activities have been focussed on the description of the circulation of the Baltic Sea and it's influence on the seasonal and the inter-annual distribution and transport of larvae originating in the deep basins (the centres of the stock's spawning activity). In order to investigate drift of fish eggs and larvae in the Baltic Sea with respect to variable atmospheric forcing conditions, Hinrichsen et al. (2001a) developed a transport index which allows the identification of changes in transport regime within and between spawning seasons. This index was based on drift scenarios calculated with the same hydrodynamic model of the Baltic Sea, which has been utilized also to clarify the potential mixing of early life stages between the western and eastern Baltic cod stocks (Hinrichsen et al., 2001b). These drift studies also allowed to identify the potential nursery areas of Baltic cod as well to describe the environment in which larvae and juveniles changed from pelagic to demersal habitat.

4.3.2. Material and Methods

Simulated three-dimensional velocity fields were extracted from the hydrodynamic model of the Baltic Sea (at a 6 hours interval) in order to develop a data base for a Lagrangian particle tracking exercise on larval cod. This data set offers the possibility to derive Lagrangian drift routes by calculating the advection of ”marked” water particles. The positions of the drifters varied over time as a result of the three-dimensional velocities that they experienced. Furthermore, the data contain information on the temporal evolution of the hydrographic property fields (temperature, salinity, oxygen, etc.) along the trajectories. Simulated drift routes were obtained from Eulerian flow fields by utilization of a Lagrangian particle-tracking technique. First, the hydrodynamic model on Baltic cod larval drift has been utilized for the time period 1979 to 2004 in order to obtain means of intra-annual variability in distribution and transport patterns. In order to consider its seasonal variability in relation to spatial and temporal variations in larval transports Lagrangian drifters were released at depth between 25 and 35 m (depths at which feeding larvae occur after vertical feeding migration) on a regular spaced grid enclosed encompassing the main cod spawning areas of the Baltic Sea (Fig. 4.3.1). Drifters, at their release representing first feeding larvae, were inserted into the modelled flow fields at 10 days intervals and were tracked for a period of 70 days. The release dates commenced April 1 and ended September 20 thereby encompassing the historic as well as the present main spawning period of eastern Baltic cod (Wieland et al., 2000a). Horizontal maps of modeled juvenile distributions were constructed by simple integration of larval/juvenile drift endpoints obtained from hydrodynamic model simulations in rectangles (representing quarters of ICES rectangles) of approximately 15x15 nm size covering the main spawning areas as well as the most likely larval and juvenile appearance. To allow direct relative comparisons of final larval/juvenile distributions between the sub-areas, numbers of drifters found in rectangles were normalized by dividing them with respect to the maximum number of larval/juvenile drift endpoints found in the sub-area.

To obtain a general impression of meteorological impacts on larval transport, we related larval and juvenile occurrence within the different basins of the Baltic to atmospheric conditions. Relevant for the atmospheric forcing are the local conditions over the Baltic Sea, which are embedded into the large-scale atmospheric patterns. With respect to local forces, Lehmann et al. (2002) defined a Baltic Sea Index, which is the difference of normalised sea level pressure anomalies between Oslo (Norway) and Szczecin (Poland). Daily mean sea level pressures at each position obtained from NCEP/NCAR re-analysis data (Kalnay et al., 1996) are normalised by deviding them by the long-term mean (1948-1999) standard deviation. For example, a positive BSI corresponds to an anomalous sea level pressure difference associated with westerly winds over the Bornholm Basin, in near surface layers leading to transport towards the east. In contrast, a negative BSI corresponds to easterly winds, favouring currents towards the west near the sea surface.

4.3.3. Results

Fig. 4.3.2 displays the mean distribution after 70 days drift periods for Bornholm Basin spawners if all available final endpoints of the drift patterns (18 annual releases times 26 years) were taken into account. The driftter endpoints are mainly concentrated in the center part of the basin, i.e. most juveniles have to settle in less optimal oxygen environment. Typically, adult cod have a minimum requirement of 4 ml/l oxygen content for survival (Chabot and Dutil 1999). Because there is no specific knowledge on oxygen tolerance of juvenile cod for settlement, we have assumed the same threshold for oxygen content for juvenile cod changing from pelagic to demersal stage. Fig. 4.3.3 represents the potential nursery areas for driftres released in the Bornholm Basin if areas with oxygen concentration < 4ml/l were excluded from the analyses. Compared to Fig. 4.3.2, juveniles settle mainly at the edges of the basin where the halocline hits the bottom, whereas settlement probability in the deepest part of the basin is low and was only possible during inflow and post-inflow years.

Figs. 4.3.4-6 display the potential nursery areas for juvenile cod initially started within different spawning grounds (Bornholm Basin, Gdansk Deep and Gotland Basin). The figures represent the mean final destinations of drifters for different decades (1979-1988 and 1989-1998). For drifters initially released in the Bornholm Basin, horizontal distribution maps clearly show higher concentrations of juveniles remaining in the Bornholm Basin during the first decade (Fig. 4). Only a low number of particles were transported out of the basin into easterly directions. Juveniles of Bornholm Basin spawners on average had distributional peaks in the majority of rectangles in the southern and northern shallower water areas of the Bornholm Basin. During the second decade, there is evidence for stronger eastward and northward transport of juveniles with some minor concentrations of juveniles found in the Gdansk Deep as well as in the eastern part of the Gotland Basin.

Potential nursery areas for the easternmost stock components of Baltic cod were identified to on average remain in the eastern part of the Gdansk Deep and Gotland Basin. Areas with highest average concentrations of juveniles were found along the Lithuanian- as well as along the Latvian coast lines. During the second decade, due to higher transport rates along the coast lines towards the north, there is evidence for intense contribution of Gdansk Deep drifter to the Gotland Basin (Fig. 4.3.5). Similar decadel variations in the location of potential nursery areas for the Gotland Basin stock component were not observed (Fig. 4.3.6). Generally, as obtained from the results of the longterm model runs, horizontal larval and juvenile distributions indicate higher dispersal of larvae and juveniles during the second compared to the first decade.

The overall changes in nursery areas of juveniles cod initially released in the Bornholm Basin is coupled to local atmospheric forcing conditions for larval drift (Fig. 4.3.7). Below the wind-induced Ekman layer, highest transport rates towards the southern shallower water regimes are related to high easterly winds (bsilo). In contrast, transport towards the north is mainly caused by high westerly winds (bsihi). Larval and juvenile transport towards the adjacent basins in the east (Gdansk Deep and Gotland Basin) is mainly due to strong easterly winds.

4.3.4. Discussion

In this study, the transport of eastern Baltic cod larvae spawned within the major spawning grounds was investigated by detailed drift model simulations for the years 1979 to 2004. We have analysed in which habitats larvae and juvenile cod potentially dwell and where larvae and juvenile are able to change from pelagic to demersal habitat. The results of these exercises on particles initially released within the Baltic cod spawning grounds yielded a clear dependency on wind-induced drift of larval cod, which is mainly controlled by the local atmospheric conditions over the Baltic Sea. Furthermore, there is some indication for the effect of climate variability on the final destination of juveniles in their nursery areas in terms of decadel variability.

In general, the information on temporally and spatially resolved Baltic cod nursery areas obtained here has the potential to design MPAs for juvenile cod. Our modelling approach is quite simplistic and should be seen as a baseline exercise, to be followed by more comprehensive investigations, e.g. comparison of potential nursery areas for different time periods during the Baltic cod spawning season (early vs.late spawners). Because the strong inter-and intra-annual variability of the locations of potential Baltic cod nursery areas, considerable effort has already concentrated on an easier parameterization of processes potentially responsible for the location of nursery areas. The identification of nursery areas of Baltic cod might benefit from simplified but ``online" accessible physical forcing parameters, e.g. the Baltic Sea Index.

Finally, as a starting point, the information on mean locations of potential nursery areas obtained from this study were used so far in the ISIS-fish model that aims in the development of a simulation model for assessing the impact of spatial management measures on resources and fisheries.

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4.3.6. Figures

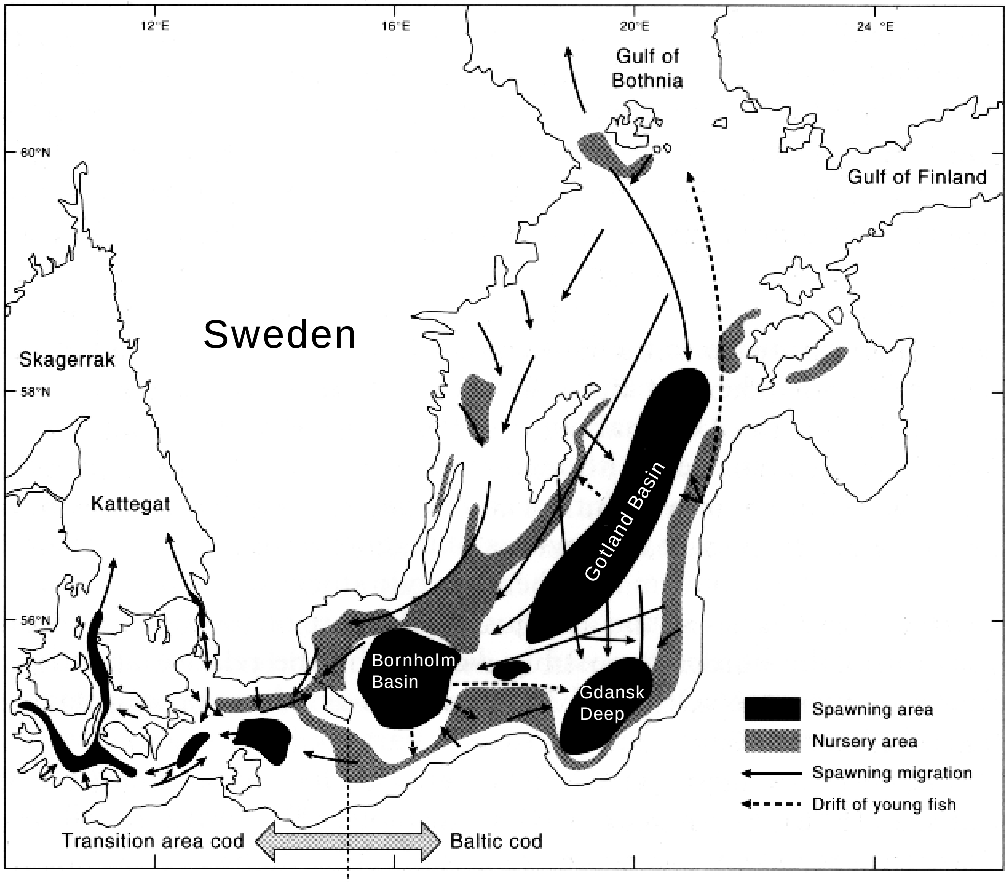

Fig. 4.3.1. Baltic cod spawning and nursery areas (after Bagge et al. 1994)

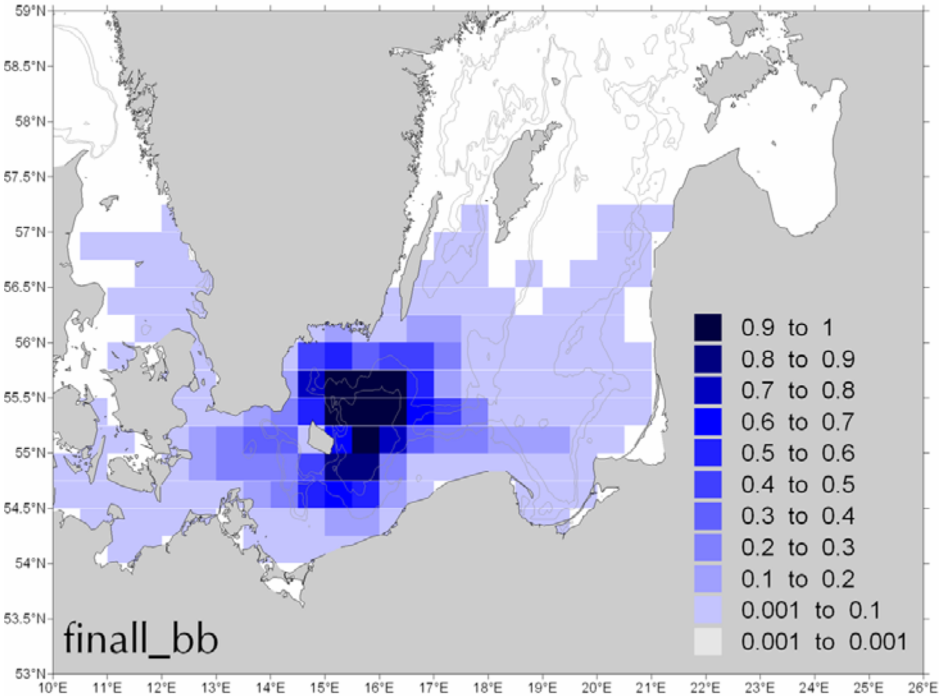

Fig. 4.3.2. Mean final destination of larval/juvenile drifter initially released in the Bornholm Basin for the years 1979-2004 (not accounted for minimum oxygen requirements of Baltic cod juveniles.

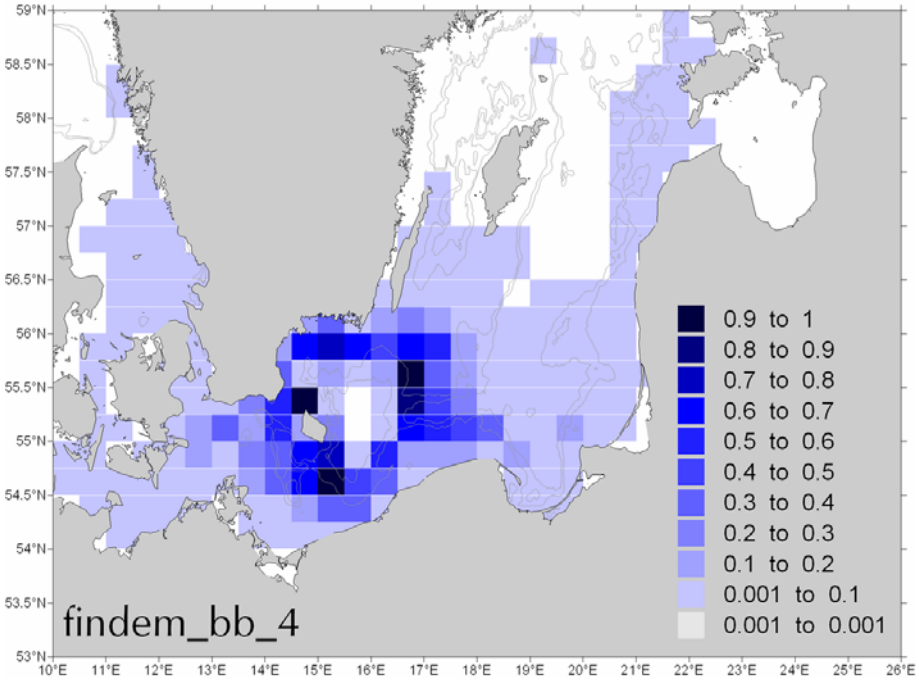

Fig. 4.3.3. Potential nursery areas of the Bornholm Basin cod stock component for the years 1979-2004 (accounted for minimum oxygen requirements of Baltic cod juveniles).

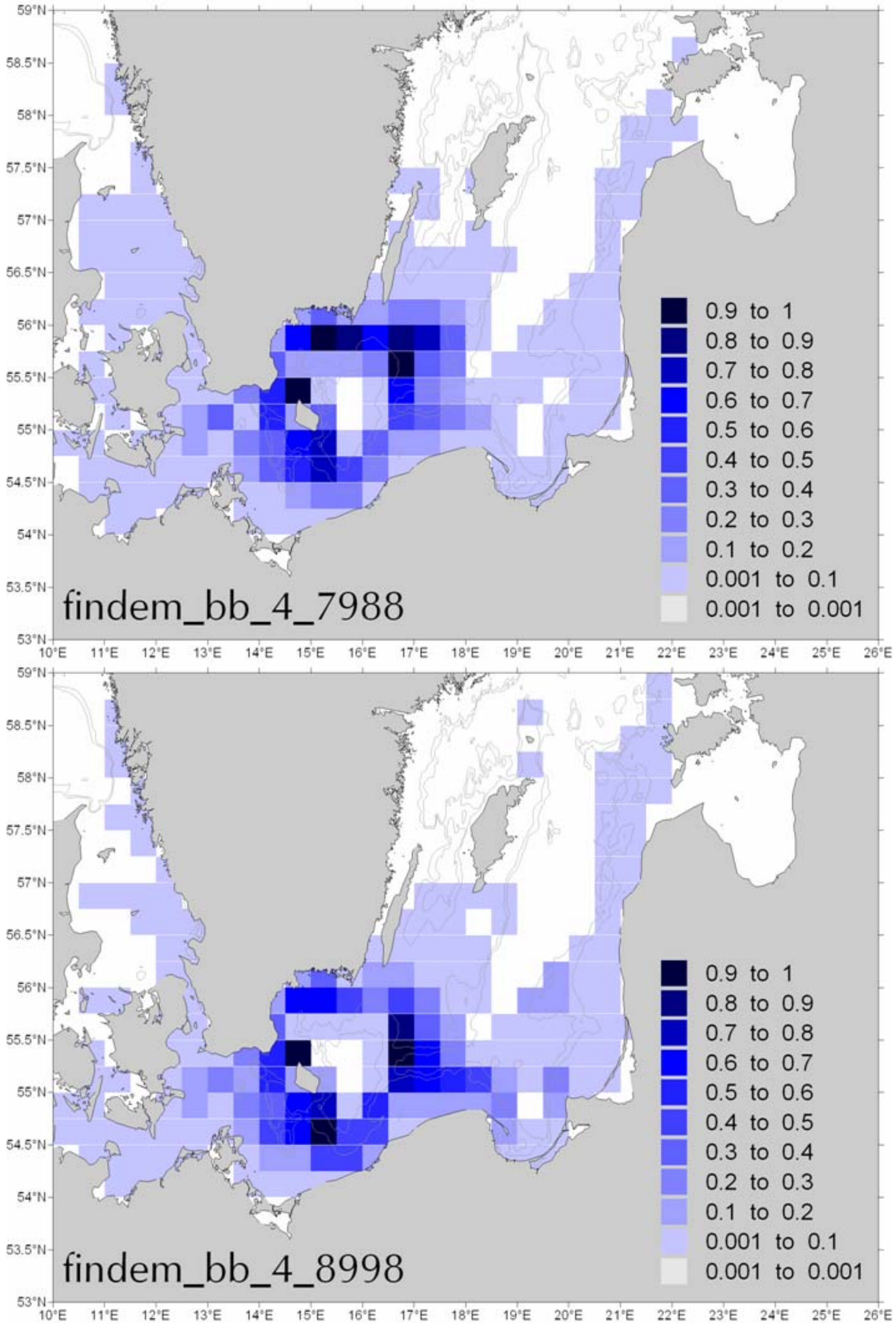

Fig. 4.3.4. Decadal variability of potential nursery areas of Bornholm Basin spawners, upper panel: 1979-1988, lower panel: 1989-1998

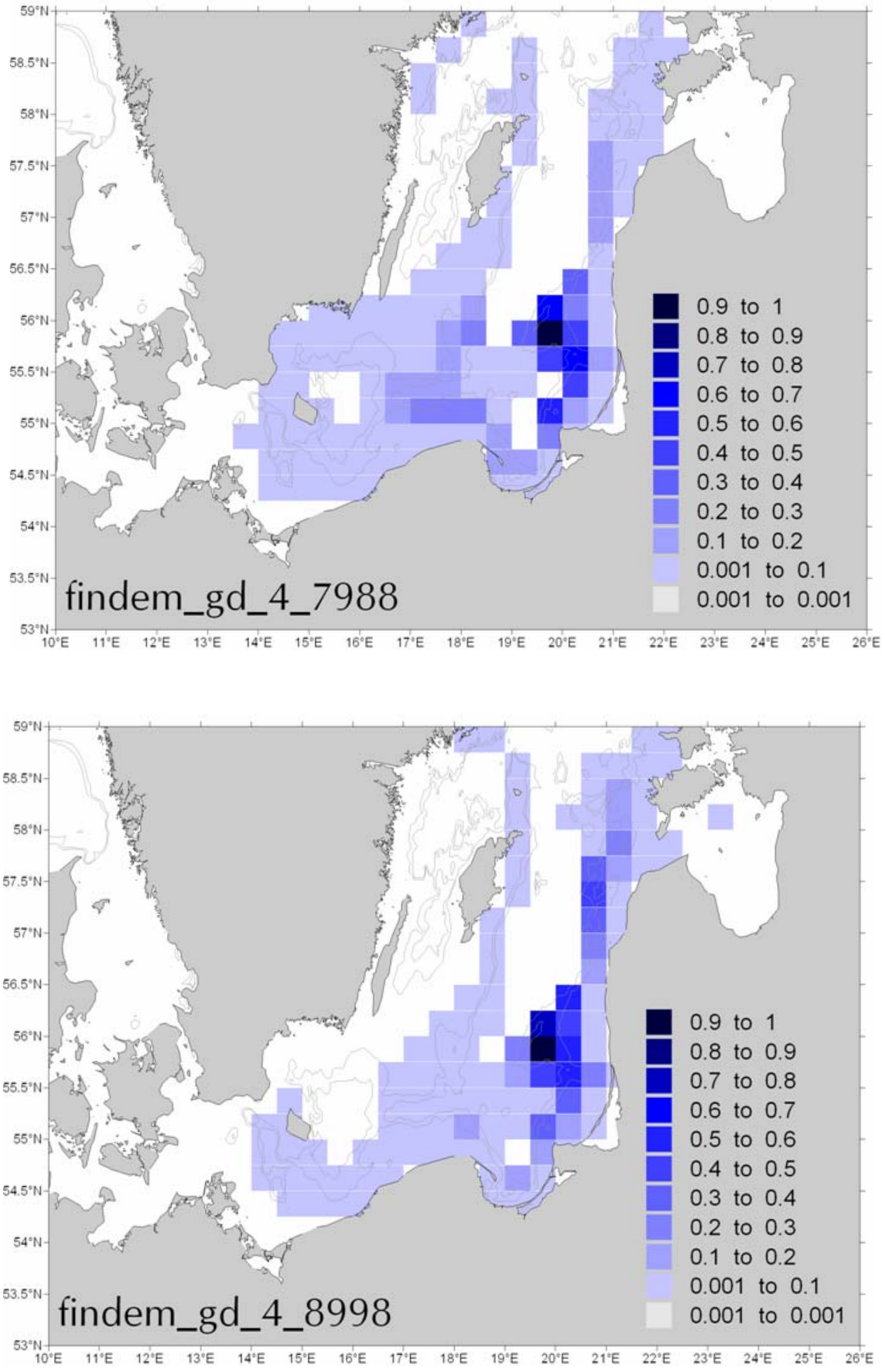

Fig. 4.3.5. Decadel variability of potential nursery areas of Gdansk Deep spawners, upper panel: 1979-1988, lower panel: 1989-1998

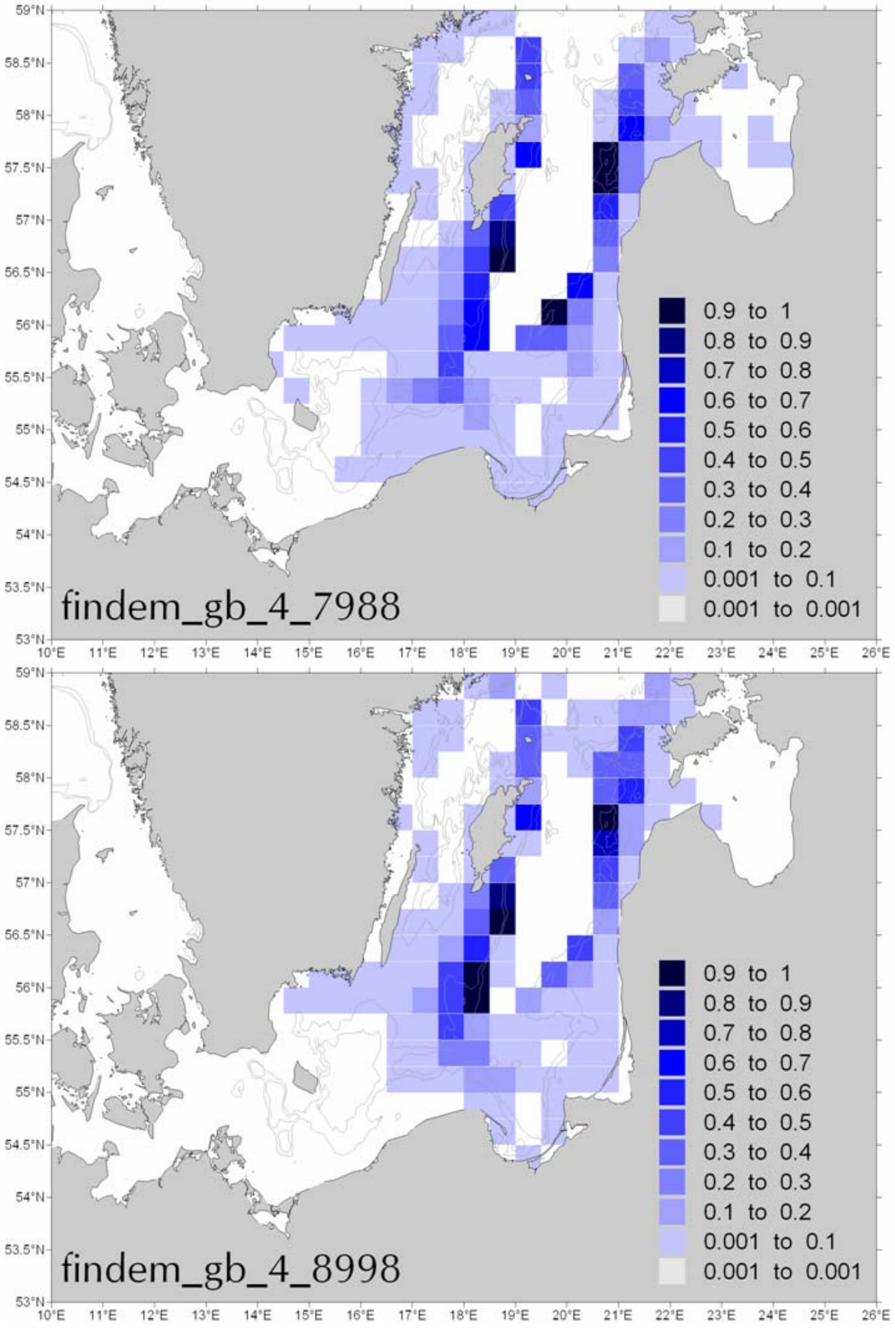

Fig. 4.3.6. Decadel variability of potential nursery areas of Gotland Basin spawners, upper panel: 1979-1988, lower panel: 1989-1998

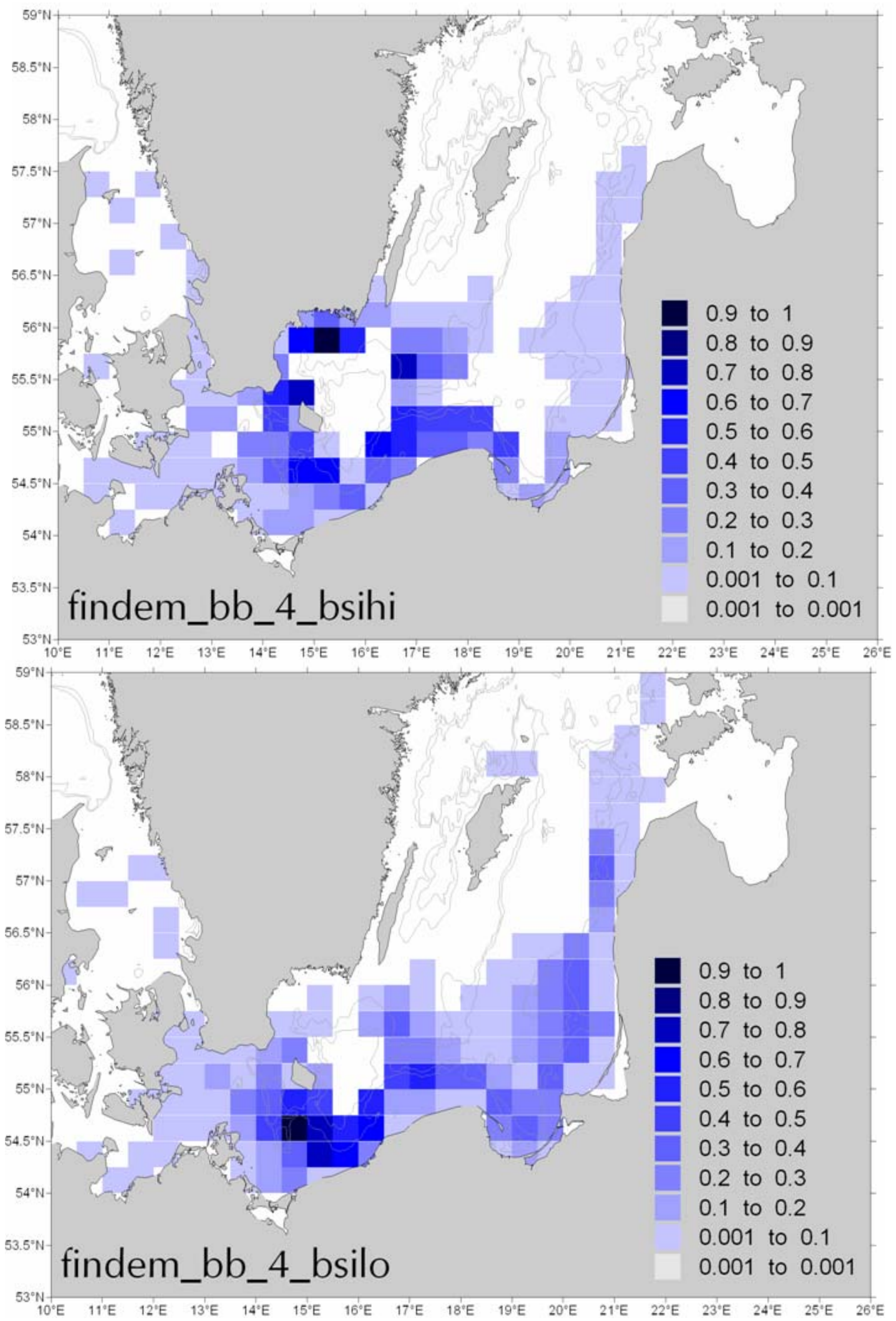

Fig. 4.3.7. Wind induced variability of potential nursery areas of Bornholm Basin spawners, upper panel: high BSI (high westerly winds), lower panel: low BSI (strong easterly winds)

4.4 Spatio-temporal stock trends of Baltic sub-components derived by disaggregated MSVPA

Eske Teschner and Gerd Kraus, IfM-GEOMAR

Introduction

In the Baltic Sea the spatial and temporal suitability of the spawning habitats of cod (*Gadus morhua*) vary dramatically with the oxygen conditions at the depth of incubation of the eggs (e.g., Wieland *et al.* 1994). As a consequence, different stock components of cod exhibit distinct trends in different areas of the Central Baltic (Sparholt and Tomkiewicz 2000), with a corresponding variation in predation pressure on its major prey species, sprat (*Sprattus sprattus*) and herring (*Clupea harengus*) (Sparholt 1994). In turn the population development of these planktivores determines the predation intensity on early life stages of cod (Köster and Möllmann 2000). Hence, a prerequisite for the implementation of effective MPAs is to resolve the spatial and temporal dynamics of cod and sprat as a basis for defining spatial and temporal windows for possible MPAs. This requires analyses on distribution and abundance patterns of adult fish relative to key environmental conditions and habitats. In the Baltic Sea, the MSVPA represents a suitable tool for calculating adult stock sizes of cod, sprat and herring taking into account species interactions. At present MSVPAs are run for two areas in the Baltic, a Western and Central Baltic component to match the stock units used in the regular stock assessments, with the Central Baltic component dominating in terms of biomass and abundance (ICES 1998/ACFM:16). Within these two regions, the abundance and biological characteristics of the three species are heterogeneous both spatially (between Sub-divisions) and temporally (inter and intra annually). For example, population sizes of Central Baltic cod, as resolved by international bottom trawl (Sparholt and Tomkiewicz 2000) and ichthyoplankton surveys (Köster *et al.* 2001a), have revealed distinct distributional trends.. The abundance and characteristics of herring and sprat have also been observed to vary spatially and temporally in the different Sub-divisions of the Central Baltic (e.g., Ojaveer 1989). The herring stock in the Central Baltic is comprised of a number of different spawning components exhibiting variations in spawning period and growth rates as well as meristic, morphometric and otolith characteristics (e.g., Parmanne *et al.* 1994). For sprat the existence of distinct populations is controversial as deviations in growth rates observed between sub-areas have been explained by immigration from the western Baltic and by migration between different basins (Parmanne *et al.* 1994). However, other authors state that sprat in the eastern Central Baltic form local populations (Ojaveer 1989), which can be separated, primarily by otolith characteristics (Aps 1981).

Consequently, the Baltic Sea area would particularly be suitable for establishment of a *spatially resolved* multispecies model to quantify the stock dynamics of cod, herring and sprat in the different Subdivisions, displaying distinct environmental conditions. These requirements are met by the area-disaggregated MSVPA, which has a spatial resolution of single Sub-divisions (SD), i.e. SD 25 (Bornholm Basin), SD 26 (Gdansk Deep) and SD 28 (Gotland Basin). Thereby the 3 historic major spawning areas, each with characteristic hydrographic conditions and stock trends, are separated.

A detailed description of the MSVPA model study is enclosed in **Annex 2**.

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4.5 Population dynamic modelling considering environmental and management options

Röckmann C., St. John, M. A., Schneider. A., Richard, S.J., Tol, U., IHF-Univ. Hamburg

The aim of this population dynamic modelling study is to pilot future evaluations of MPAs' potential to increase size of the eastern Baltic cod stock under different management and environmental scenarios. The model used is a single species dynamic population model, which is based on multivariate regression analysis (Röckmann et al., in press). The model is in essence a simulation model whose output (cod stock size) is randomized over different environmental scenarios and management options.

The model is separated into population and exploitation elements:

Population
Recruitment of the eastern Baltic cod is a function of spawning stock size and environmental conditions. Factors affecting environmental conditions and thereby reproductive volume are oxygen and salinity. While the model is in essence a single species model, cannibalism is taken into account but species interactions e.g. sprat predation on cod eggs is excluded in the model. That is, the cannibalism independent natural mortality is treated as constant. The seasonal migration estimation is a qualitative description whereas the spawning migration is a function of spawning stock size and reproductive volume. Feeding migration is assumed to be density-dependent. Mixing with the western Baltic cod stock is excluded.

Exploitation
The model assumes that fishers comply perfectly with the management measures. That is, fishing mortality is exogenous and constant i.e. it is imposed by the "policy maker". The model does rough calculation of revenues of harvests, based on management scenarios and constant fish price.

As all MPA related models, this population dynamic model is temporally and spatially explicit. The studied subdivisions in the Baltic Sea Main Basin are 25, 26, 28 and the temporal time-step of the model is 3 months. The output of the model is management and environmental dependent stock size, catch, revenue and cost of harvest. A more detailed description of the population dynamic model is found in the papers of Röckmann et al. (in press).

The policy analysis, focusing on different regulations of fishing mortality, was embedded into three environmental scenarios, assuming low, medium, or high climate and environmental change. The environmental assumptions were based on simulation results from a coupled atmosphere-ocean regional climate model, which project salinity in the Baltic Sea to decrease by 7-47% in the period 2071-2100 relative to the reference period 1961-1990.

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4.6. Development of an ISIS-Fish model for the Baltic Case Study

Gerd Kraus (IfM-GEOMAR/DIFRES), Christian Möllmann (IHF-Univ Hambrug), Hans-Harald Hinrichsen (IfM-GEOMAR), Dominique Pelletier (IFREMER)

4.6.1. Introduction

The development of an ISIS-Fish model for Eastern Baltic cod is a combined activity of project partners 13, Institut Français de Recherche pour l'Exploration de la Mer, 1, Danish Institute for Fisheries Research and 4, Leibniz Institute of Marine Sciences Kiel.

Eastern Baltic cod (*Gadus morhua callarias*) has collapsed due to climate-driven adverse hydrographic conditions and overfishing, remaining at historically low stock levels to date. A series of different spatio-temporal fishing closures (Marine Protected Areas – MPAs) has been implemented since 1995 as a management measure to reduce fishing mortality and restore the spawning stock. However, no signs of recovery have been observed yet, suggesting either MPAs to be an inappropriate management measure for this stock, or pointing towards a suboptimal design of the closures. We use the spatially-explicit fisheries simulation model ISIS-Fish (http://www.ifremer.fr/isis-fish), combining an age-structured population sub-model with a multi-fleet exploitation sub-model and a management sub-model to evaluate the performance of past and existing closures, and the potential of alternative MPA designs (different sizes and timings) to help the stock to recover.

During the reporting period (i) the large amount of biological knowledge available about the cod population, and (ii) existing spatially-disaggregated fisheries data have been compiled and analyzed. An initial, basic parameterization of the population model within ISIS Fish has been done. The parameterization of the exploitation module is presently ongoing, first simulation runs will be presented at the International MPA Symposium in Murcia, Spain in September 2007.

4.6.2. Model parameterization

4.6.2.1. Model scales (space and time)

- Fishery area: ICES Subdivisions 25-28
- Spatially-explicit with spatial scale: 1/4 ICES statistical rectangle
- Zones independently defined for population, exploitation and management: zones are defined as groups of cells on the grid
- 3 spawning areas: in ICES SD 25, 26, 28 delimited by the 60m depth line in SD 25, SD 26: 80m, SD 28: 90m (Figure 4.6.1)
- 3 nursery areas based on output of a hydrodynamic model (NA1, NA2, NA3) (Figure 4.6.2)
- 3 Feeding areas on the slopes of the three basins, i.e. starting on the borders of spawning areas up to the 40m depth line (may need to be refined) (Figure 4.6.3)
- Monthly time-step (cannot be changed)

4.6.2.2. Present assumptions and model parameterisation

Population

- The number of spawning grounds depends on hydrodynamic conditions:

hydrodynamic scenario	**spawning areas**	**nursery areas**
Good	SD25, SD26, SD28	NA1, NA2, NA3
bad	SD25, SD26	NA1, NA2, NA3

- Correspondence between spawning areas and nursery areas:

hydrodynamic scenario		**Comment**
good	SD25->NA1 (0.5), NA2 (0.3), NA3(0.2) SD26->NA1 (0.3), NA2 (0.5), NA3(0.2) SD28->NA1 (0.2), NA2 (0.3), NA3(0.5)	values may be refined
bad	SD25->NA1 (0.5), NA2 (0.3), NA3(0.2) SD26->NA1 (0.2), NA2 (0.5), NA3(0.3)	values may be refined

- Spawner-egg-recruit relationships:

  First run:
  - egg=linear function of spawner abundance with fecundity coefs and 0.5 to get females only (independent of climate)
  - egg and larvae mortality of 0.9999 to be refined from MSVPA (does not depend on climate)

  Second run alternatives:
  - Sptially explicit stock recruit relationships fitted to area-disaggregated MSVPA output for two different hydrographic regimes
  - Spatially explicit environmentally sensitive S-R relationships from BECAUSE (e.g., Köster et al. 2003)
  - Sage-based model with different mortality parameterisation for egg, larval and juvenile stages

- Reproduction seasons:
  - Spawners migrate to spawning areas from February until May. Old individuals move in first because they spawn earlier:
    cod7 and 8 migrate in February
    cod6 and 5 migrate in March
    cod4 and 3 migrate in April
    cod2 migrates in May.
  - Spawners reproduce from April until June. Migration out of the spawning area:
    All age groups from migrate to feeding areas from July to September:
    July : coef=0.3 for each mature group.
    August: coef=0.6 for each mature group.
    September: coef=1 for each mature group.
    NB: coef values have to be tested to see the patterns they generate.

- Recruitment:
  - Number of months between reproduction and recruitment = 6
  - Recruitment lasts from October to December
  - Distribution of recruits from spawning to nursery grounds:

repro zone\nursery	**NA1**	**NA2**	**NA3**

SD25	0.5	0.3	0.2
SD26	0.2	0.5	0.3

- Catchability:

  First run:
  - Te same value for each group and each season: 10^{-4}
  - NB: need to discuss the fact that fish is more dense and schooling in the spawning areas.

- Other biological parameters:
  - Grth (Von Bertalanffyl curve fitted on length converted weight at age data from most recent assessment (ICES 2006)),
  - Ntural mortality (density dep. natural mortalities from area-disaggregated -MSVPA incl. predation mortality, alternative 0.2 for all ages as in standard assessments),
  - Age or length at maturity (so far sex ratio of 0.5 and knife edge maturity at
  - age 3, to be refined)

Exploitation

- Vessels: 3 groups: <12m; 12-24m; >24m
- Gears: suggestion (TEMAS): 3 groups, i.e. gillnets, otterbord, other.
- Metiers (=fishing activities) = gear X target species X fishing grounds: To be defined
- Strategies (=groups of vessels that practice the same metiers throughout the year): To be defined
- Exploitation costs and revenues (calculation of economic indicators and impact on effort reallocation): Not yet included
- Fisher's response to management and economic conditions (fisher reallocate effort to other metiers or strategies): Not yet included. Full compliance to management is assumed.

Management scenarios

Evaluation of:

- Summer ban on targeted cod fishing introduced in 1995 and in 2004 enforced from 15th of April to 31st of August; enforced from 1st of May to 15th of September in 2005, and from 1st of June to 15th of September in 2006.

- Special "spawning" closures for all Fishery from 15. May to 31. August in an area east of Bornholm, extended in dimensions in 2004. Similar closures established the Gdansk Deep and southern Gotland Basin since 2005 (Fig. 4).

- Alternative closure scenarios that lead to stock recover under different environmental scenarios.

4.6.3. Expected outputs

- Simulation of catch, effort, abundance at any level and possibly summed over stages, metiers, months
- Results exported under text format: catch, effort, abundance at disaggregated level.
- Available R scripts for importing these text files and building graphics from results (R package in construction for Spring 2007).

4.6.4. References

1. Köster, F.W., Hinrichsen, H.-H., Schnack, D., St. John, M. A., MacKenzie, B. R., Tomkiewicz, J., Möllmann, C., Kraus, G., Plikshs, M., Makarchouk, A. and Eero, A. 2003. Recruitment of Baltic cod and sprat stocks: identification of critical life stages and incorporation of environmental variability into stock-recruitment relationships. *Scientia Marina* **67** (Suppl.1), 129-154.
2. ICES 2006. Report of the Baltic Fisheries Assessment Working Group. ICES CM 2006 / ACFM:24.

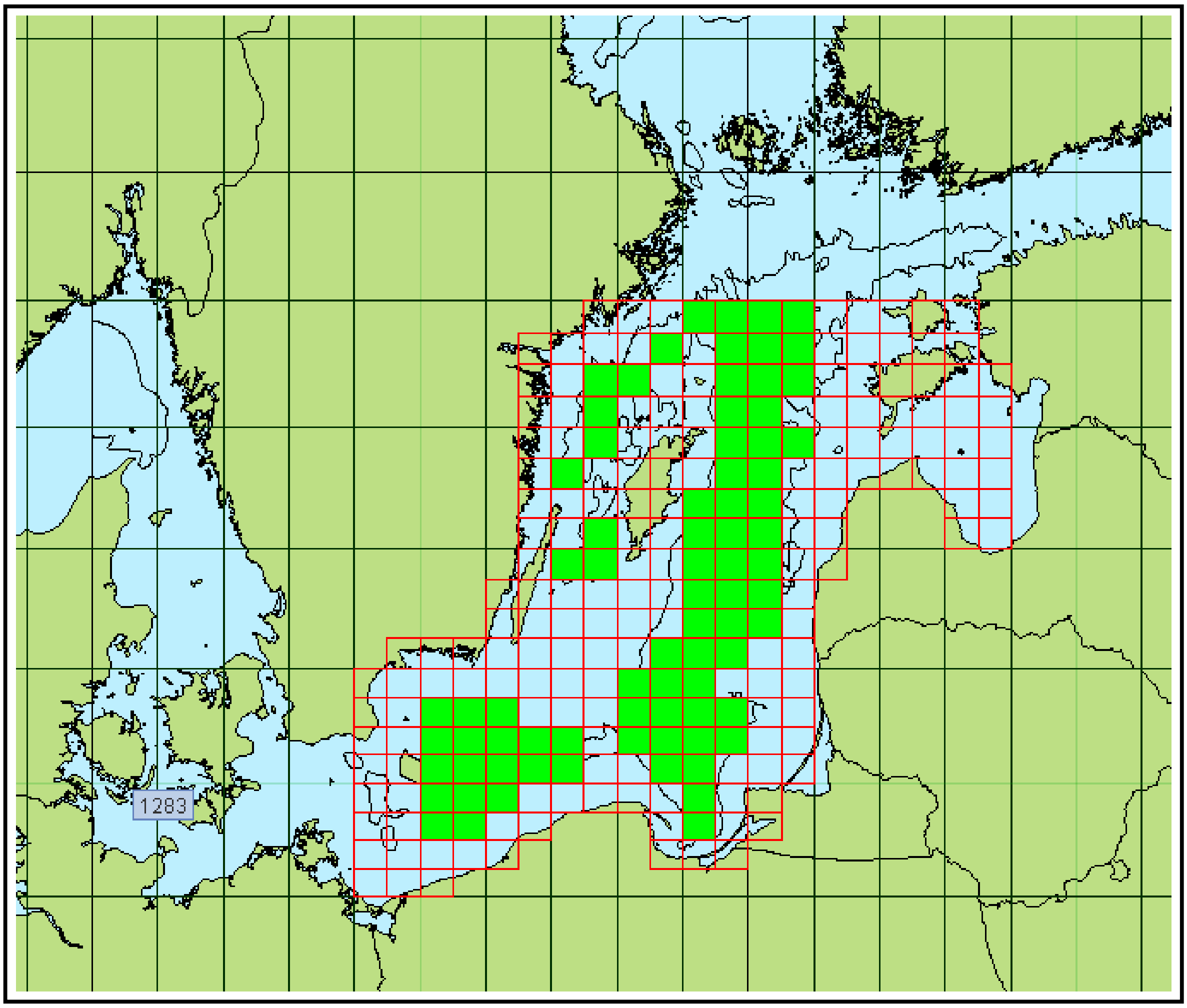

Figure 4.6.1: Spawning areas of Eastern Baltic cod as defined in the spatially explicit fisheries simulation tool ISIS-Fish. Spawning areas are approximately limited by the 60 depth line in SD 25, 80m depth line in SD 26 and 90m depth line in SD 28. Definition in relation to depth is based on long term average observed egg distribution.

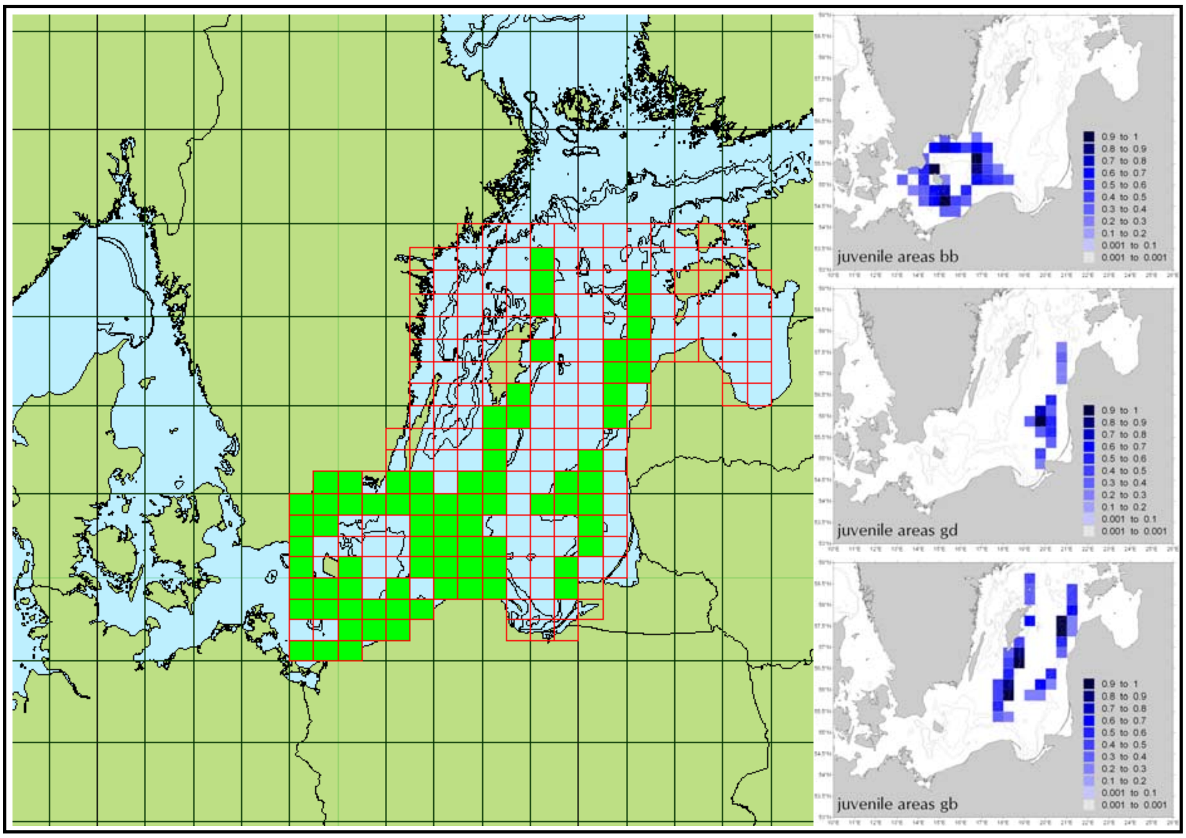

Figure 4.6.2: Long term average nursery areas for Baltic cod simulated by a hydrodynamic drift model for stock components in ICES SD’s 25, 26, 28 (right panel) and its implementation into the population sub-model of the spatially explicit fisheries simulation tool ISIS-Fish (left panel).

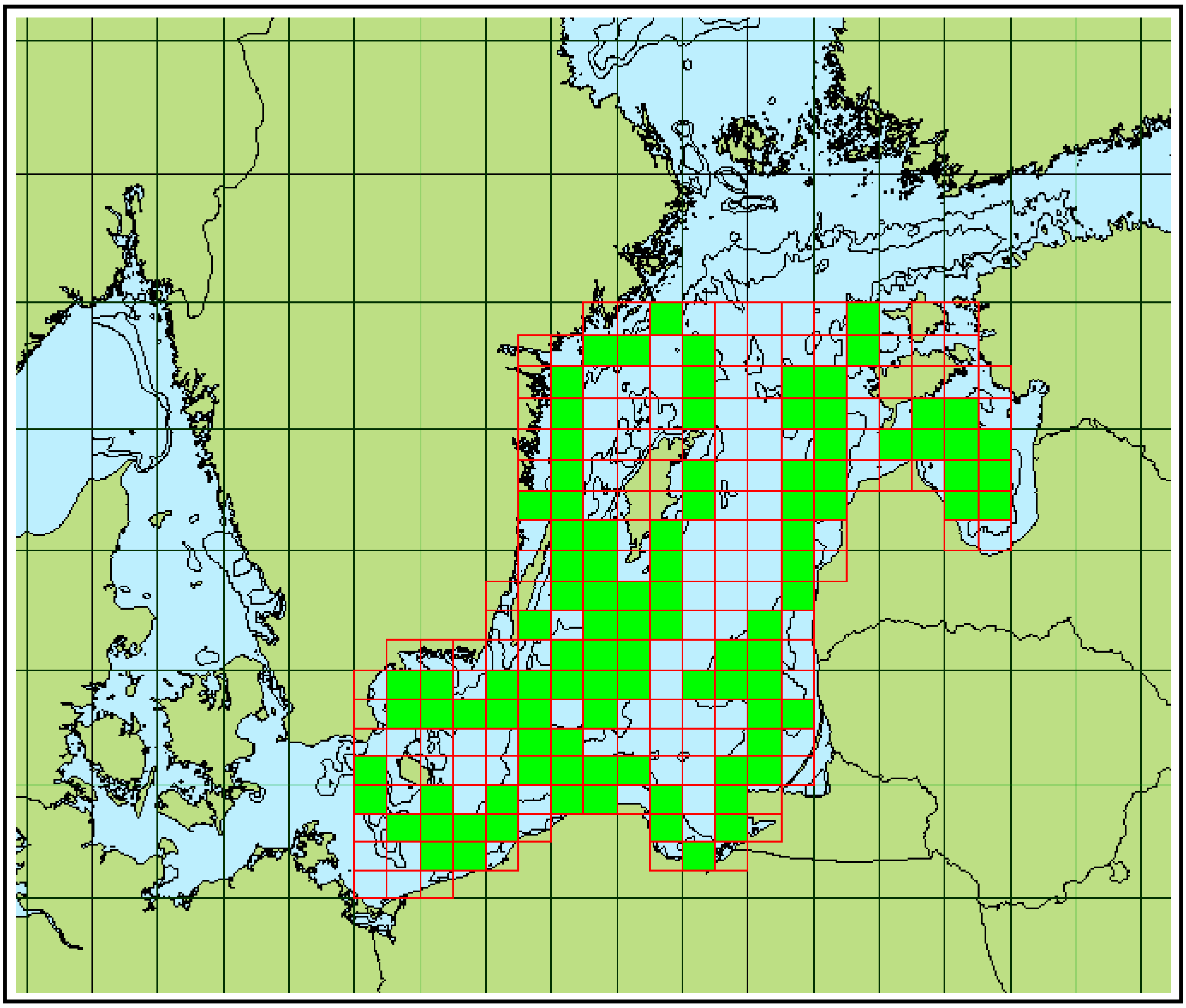

Figure 4.6.3: Feeding areas of Eastern Baltic cod as defined in the spatially explicit fisheries simulation tool ISIS-Fish. Feeding areas are defined as slope areas of the deep basins approximately limited by the 40m and 80m depth lines.

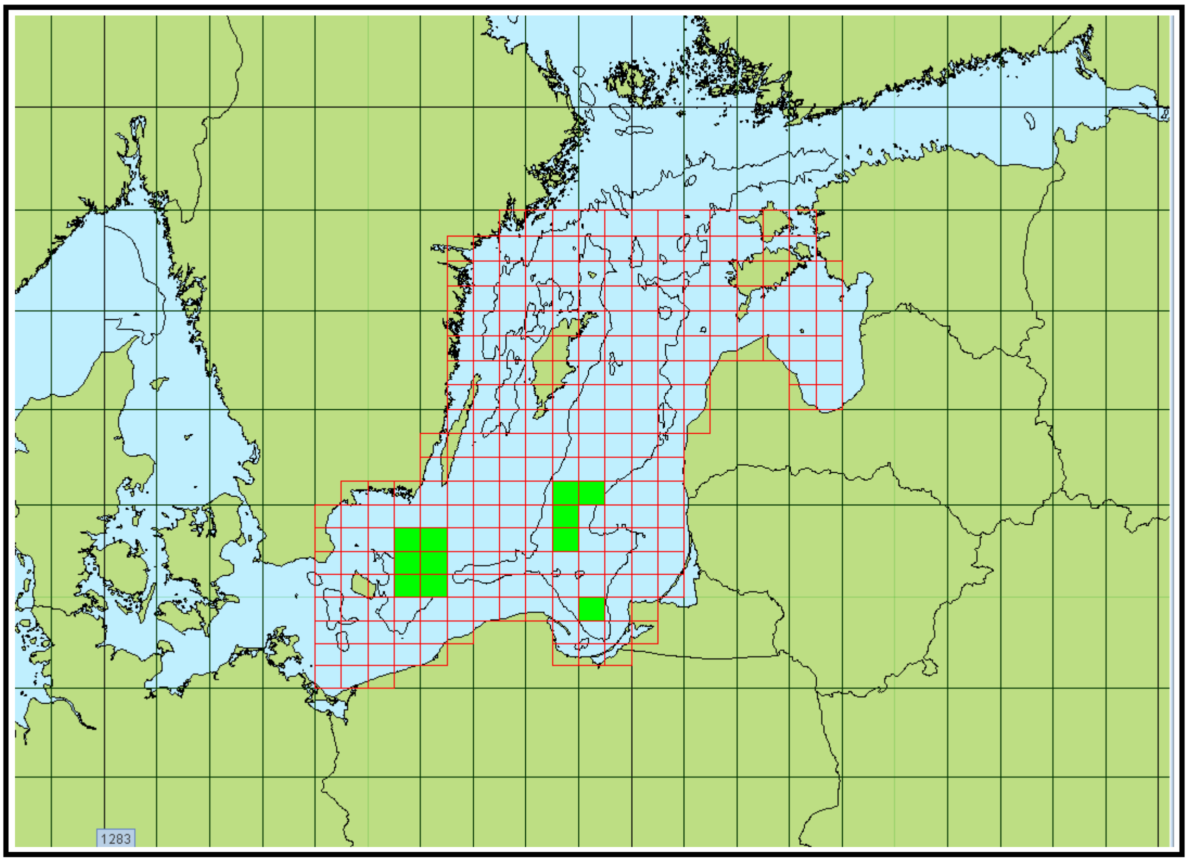

Figure 4.6.4: Implementation of spawning closures as enforced in 2005 into ISIS-Fish to simulate and evaluate the performance.

4.7 Assessing the effect of Marine Protected Areas by the TEMAS model

Per J. Sparre, DIFRES

4.7.1. Introduction

To make a complete assessment of the effect of marine protected areas, which has the purpose of improving the production of Baltic cod recruits, it is required to model a long suite underlying relationships, such as.

1) The relationship between spawning stock biomass and recruitment
2) The relationship between environment and recruitment, including the impact of the environment on egg and larvae survival.
3) The temporal and spatial distribution of spawners (distribution of egg production) and juveniles, including spawning migration and migration of juvenile.
4) The relationship between fishery and recruitment, including gear selection and spatial /temporal distribution of fisheries.
5) The spatial/temporal reallocation of fishing effort after closure of an MPA, including, e.g. the impact of economy on the behaviour of fishers.
6) Predation on cod larvae and juveniles, including cannibalism.
7) Food availability for cod larvae and juveniles.

More fundamental mechanisms could be listed, but even these 7 items makes one almost give up making a complete model for the effect of MPAs. Some theories and some parameters estimation exists for all the items listed, but none are believed to be fully understood or fully documented with observations and estimations of model parameters.

The word “assessment” is used here conceptually as used by ICES working group. An assessment is composed of two parts (1) Estimation of parameters from historical data (2) Prediction based on the parameters estimated under (1). The main thing to predict is the recruitment, and needless to say to any worker with more than one year of experience in fisheries science, this is “next to impossible”. What may possibly be concluded from any model on recruitment are statements like “It is believed that the regulation (e.g. an MPA) is likely to improve the future recruitment”. Only the novice in fisheries science can hope to make quantitative prediction of recruitment. This is needless to say to the experienced fish stock assessment worker.

So when a model for the recruitment of Baltic cod, which can be used to assess the effect of MPAs, as presented here, there is no expectation from the side of the author, that it can ever be used for quantitative predictions. To underline this fact (which applies to any other recruitment model for any fish stock in the world), the model is formulated as a stochastic model, giving output in the form of probability distributions, rather than single figures.

For a full description of the model work undertaken in association with PROTECT, see **Annex 3** The model presented deals with only items 1 to 3 in the list above, although the TEMAS model can handle 4 and 5, whereas TEMAS does not cover items 6 and 7, as it assumes constant natural mortality and growth rates of larvae, juveniles and adults. The handling of 4 and 5 is a multispecies, multi fleet model of fisheries including a module for fishers behaviour (in particular reaction to regulations) based on random utility model combined with a microeconomic model. However, that technical/economic module is not covered here.

The background of the model, as presented in **Annex 3,** is an extract from an extensive report on the complete model. Many aspects of the biological model are ignored. The interested reader can obtain the full report from the author.

4.7.2. The biological frame of TEMAS

The biological model behind TEMAS, is the traditional model by Thompson and Bell (1934), which has been discussed in many textbooks on dynamics of fish stocks The major part of the biological model behind TEMAS is the traditional model, or generalizations of the traditional model. TEMAS extends the traditional models with a spatial model, accounting for, e.g. migration using the approach of Quinn et al, (1990). All these models originally were thought of as "fish stock assessment model", where parameters were estimated by (e.g.) VPA or "Cohort analysis" (Virtual Population Analysis, Derzhavin, 1922). In general, TEMAS has inherited all the unsolved problems of traditional fish stock assessment as implemented by ICES.

It is generally accepted that there are two separate Baltic cod stocks, the Western Stock in ICES Areas 22-24, and the Eastern stock in ICES Areas 25-32. The definition by the ICES areas, however, is rather problematic, and there is no doubt that mixing of the two stocks takes place. Needless to say the cod do not respect the borders defined by the ICES areas. The cod do not respect the sub-divisions of the Baltic as defined by ICES, which are not defined relative to the cod distribution.

With a few rare examples, the identification of the relationship between parent stock (SSB, spawning stock biomass) and subsequent recruitment (R) has remained elusive for marine fishes (Gilbert, 1997, Hilborn, 1997, Myers, 1997). The precautionary approach dictates that unless it is scientifically demonstrated that there is no relationship between the parent stock and subsequent recruitment, such a relationship should be assumed to exist, even if the data are ambiguous. Observations of stock and recruitment show large variation around any SSB/R curve, so scientists are not in a position to predict future recruitment with any accuracy. They are only able to tell the probability distribution of the future recruitment, and only then, if a long time series of SSB/R observations is available.

There is a suite of special theories on the factors that determines the recruitment of East Baltic cod. The spawning success is linked to the spatial and temporal distribution of the cod spawning. There is an extensive literature on the spawning of Baltic cod (Kuster *et al*, 2001,2004, 2006, Andersen.. & Mollmann, 2004). Section 7 of. the 1999 Report of the Baltic Fisheries Assessment Working Group. (ICES, 1999) summarises the knowledge basis. A more comprehensive contribution from ICES is the Report of the Study Group on Closed Spawning Areas of Eastern Baltic Cod (ICES,2004a and b). The following text is extracted from these reports. The success of recruitment is considered the key to the recovery of the Baltic cod, and the MPA's are designed to improve the success of recruitment. Therefore, special attention is given to this aspect of the cod biology.

The Bornholm Basin, the Gdansk Deep and the Gotland Basin cod are the principal spawning areas of the eastern Baltic cod stock (Figure 4.7.1). The salinity and oxygen conditions mainly define the spawning habitat of this stock as well as the water volume suited for egg and larval development. Salinity levels above 11 PSU are necessary to enable cod eggs to reach neutral buoyancy and an oxygen content above 2 ml/l in the water volume in which the eggs float is further required for successful egg development. These conditions define the so-called "reproductive volume", (RV), which has been shown to be positively related to the recruitment of Central Baltic cod.

The processes affecting the RV are:

i) The magnitude of inflows of saline oxygenated water from the western Baltic,
ii) Temperature regimes in the western Baltic during winter affecting the oxygen solubility

prior to advection (which normally takes place during winter months),

iii) River runoff and

iv) Oxygen consumption by biological processes.

The Baltic Sea is characterised by a series of deep basins separated by shallow sills, and an inflow will usually fill up the first basin (the Bornholm Deep) only, with little or no transport in an eastern direction. Only if the inflow is very large or more likely if the advected water is replaced by an even denser water mass in a subsequent inflow or a subsequent inflow of less dense water glides over the earlier inflow water, the eastern Baltic basins will benefit from the water exchange. Thus, hydrographic monitoring and the unique topography make predictions of RV in a given year possible when conducted after the inflow period in January to March. The largest problem in the prediction is whether the inflow will turn south into the Gdansk Deep or north into the Gotland Deep, a process depending on local forcing conditions.

As a secondary effect of large inflows into the Bornholm Deep is that there is an increased likelihood of a potential inflow the following year will reach the eastern spawning areas.

The conditions for reproduction are potentially met in the Bornholm Basin deeper than 60 m, in the Gdansk Deep deeper than 80 m and in the Gotland Basin deeper than 90 m, where cod spawning takes place. However, the oxygen conditions in the eastern spawning areas are unfavourable for egg survival and development during stagnation periods. The conditions for successful egg development have been very limited in the Gotland Basin and Gdansk Deep since 1986.

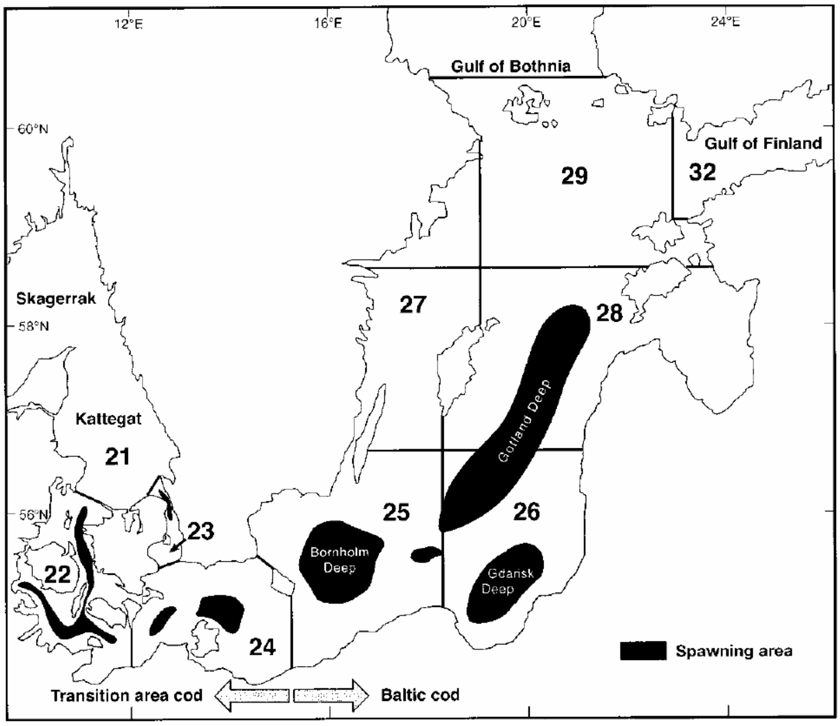

Figure 4.7.1. Historical spawning areas for cod in the Baltic Sea (from Bagge et al. 1994), modified by Aro (2000).

The size and distribution of the spawning stock component and thus the potential egg production in the various areas has also changed over time. The change in spawning stock distribution is evidenced by abundance indices from Baltic International Trawl Survey as well as from SSB estimates based on a spatially dis-aggregated multispecies VPA. Both show a very low spawning stock in Subdivision 28 (central Gotland Basin) at present, while the adult population components in Subdivision 25 (the Bornholm Basin) and 26 (the southern Gotland Basin and the Gdansk Deep) have remained at similar levels. A seasonal shift in the spawning stock distribution between areas seems also to occur. The proportion of the spawning stock increased in the Bornholm Basin during the spawning period while it decreased in the eastern spawning area. This pattern indicates spawning migration into the Bornholm Basin. Cod spawning migrations have previously been described from tagging experiments and from analyses of commercial catch rates with the migration intensity depending on the oxygen conditions in eastern spawning areas.

The hydrographic conditions may not only affect the horizontal distribution of cod spawning aggregation, but also the vertical distribution. Thus, lack of oxygen at the bottom can result in pelagic aggregations of spawning cod in the mid water layer just below the halocline. During the recent stagnation period pelagic aggregations of spawning cod have been abundant in all spawning areas. The combination of decreasing egg production and low egg survival explains the low abundance of egg and larval in the Gdansk Deep and especially the Gotland Basin throughout the 1990s as well as in most recent years. As a result, the Bornholm Basin is at present the main spawning area of the eastern Baltic cod stock.

The spawning time of the eastern Baltic cod stock is very extended, i.e., from March to August – in some years extended into September. The main spawning season lasts approximately 3 months. Peak egg abundance were observed in May / early June in the 1970–80s, while a successive shift to later month was observed in the 1990s with highest egg abundance encountered from late June to late July. The timing of spawning seems to be relatively similar in the three main spawning areas. The females generally started spawning in April and continued at least into August with the majority being in spawning condition in June-July. Males reach generally spawning condition earlier and aggregate also earlier in the spawning areas than females, which means a high fishing intensity on pre-spawning aggregations of cod will result in increased male fishing mortality rates.

A special version of TEMAS applied to the Baltic cod has been developed. This Baltic cod version attempts to account for some of the basic features of the theory for Baltic cod recruitment presented above, but a full account has not been attempted. TEMAS is primarily a model that describes fisheries, it is not the hydrographical model, that would be required to match the full theory outlined above.

4.7.3. The spatial frame of TEMAS

TEMAS offers the opportunity to account for spatial aspects, in the sense that fish and fleets can be allocated to a number of areas in a given time period. TEMAS uses a simple "box-model" to handle spatial aspects (Quinn II et al., 1990). This will require a number of additional input parameter, for example "migration coefficients", the concept of which will be explained below. TEMAS is not suited for handling of a large number of areas. It is not anticipated that TEMAS applications will use more than, say, 10 divisions of the total area. TEMAS is not constructed to deal with a division of the area in small squares (say, 30 by 30 Nm, or smaller). A division of the sea area in TEMAS is relevant only when each division differs conspicuously in terms of distributions of resources and fleets. Furthermore, some knowledge (or at least some opinions) on the distributions and movements of fleets between the selected areas and stocks must be present.

For a theoretical discussion of migration in connection with age based fish stock assessment the reader is referred to Quinn II et al. (1990). These authors also discuss the estimation of migration parameters. In principle their model is the approach planned for this version of TEMAS.

The migration is modelled in a time discrete manner:

a) Migration takes place at the end of each time period and the process of migration takes zero time.
b) During a time period the fish/shrimps are assumed to be homogeneously distributed within the area.

The "Migration Coefficient", (MC), from area A to area B is defined as the fraction of the animals in area A which moves to area B. In this definition, the "movements" include the "move" from area A to area A, i.e., the event that the animal does not move. The migration coefficient depends on (or has the indices): FAr: Starting area TAr: Destination area

Note that the sum of migration coefficients over destination areas always becomes 1.0, as the starting area is also considered a destination area: $1.0 = \sum_{TAr} MC(FAr, TAr, q, a)$

where a = age group and q = time period (division of year).

The spatial set-up for the the Baltic cod used in TEMAS, will include the spawning areas of cod (for example 1: Bornholm deep, 2: Gotland deep and 3:Gdansk deep, see Figure 4.7.1). TEMAS will be used to simulate the migration of spawners into the MPA, as well as the migration out by juveniles and adults after spawning. For that purpose we will need 4-5 areas. Furthermore the cod resource will be divided into a western stock and an eastern stock, and mixing of the stocks will be simulated. The MPAs may be considered one area (to make calculations simpler) or it may be considered 2 or 3 separate areas. The areas are composed of ICES rectangles Figure 4.7.1) and ICES areas, 22-32 (Figure 4.7.2). ICES statistical rectangles are used here because the basic data (logbook data) are by statistical rectangle.

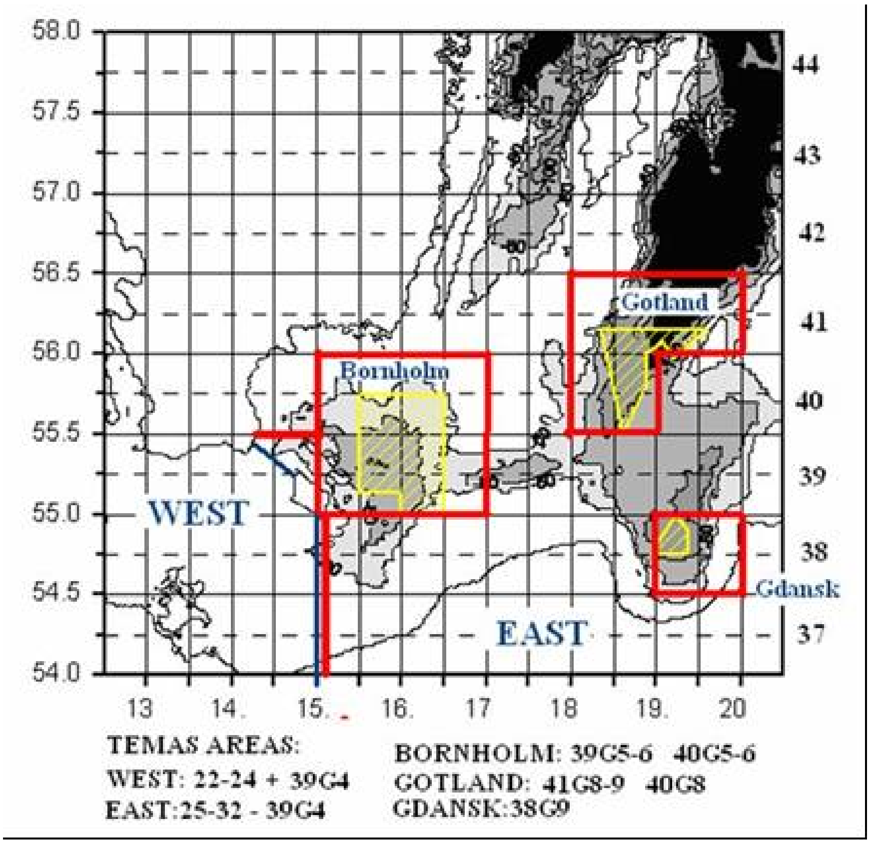

Figure 4.7.2. Tentative definition of Areas of the TEMAS simulation for the Baltic cod. MPA currently in force are the hatched areas.

The selection of areas is always a compromise between conflicting objectives or conditions. As mentioned above, the availability of data (by statistical rectangles) is one condition. The importance of an area in terms of landings is another example. Figure 4.7.3 shows the landings of cod 1993-2003 by areas (composed of ICES rectangles) of Figure 4.7.1. The Gdansk area turns out to be inferior in terms of cod landings, and it should be considered it is worthwhile to include it in the simulation of the Baltic cod. Figure 4.7.3 shows landings in the period 1995-2003 only. Had the time series gone back to the eighties the picture would be different. In the days when the cod stock was a lot bigger than in 2003, the cod would have a wider distribution, extending into the northern areas. It is believed that currently it is only the Bornholm deep that contributes to the spawning, whereas the Gotland and Gdansk deeps also contributed substantially in the eighties.

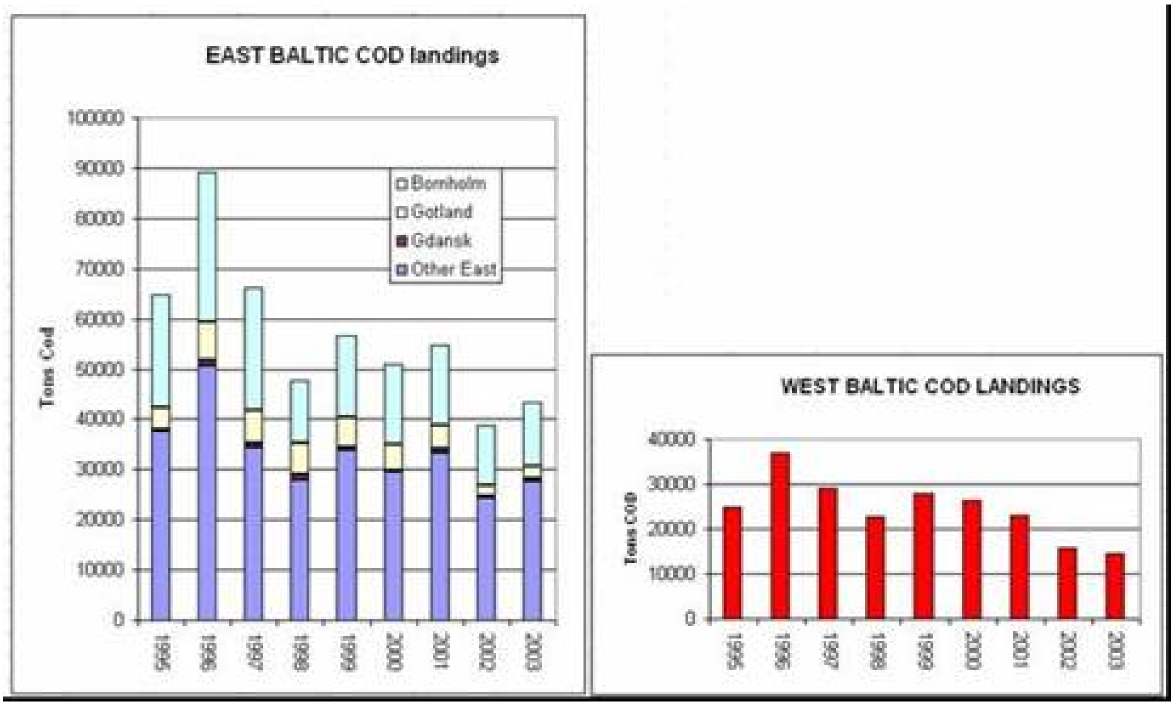

Figure .4.7.2. Landings of cod by areas (composed of ICES rectangles) of Figure 4.7.1, by Denmark, Germany, Latvia, Poland and Sweden.

For full details on Assessing the effect of Marine Protected Areas by the TEMAS model, please refer to **Annex 3.**

4.8. BEMCOM model and applications for top-down controlled ecosystems (case study 1) and wasp-waist ecosystems (case study 2).

Jesper L. Andersen, FOI

Two main activities have been undertaken in the first 24 months of the PROTECT-project: 1) data collection, and 2) building of a bioeconomic model. The former is related to WP4, while the latter is a part of WP5.

4.8.1. Collection of data

In order to parameterize the equations used in the bioeconomic model, collection of economic data has been undertaken.

Case study 1 covers several nations. FOI has collected the required economic data for Denmark, but data needs to be compiled before a general description of the data can be made.

Collection of economic data is simpler in case study 2, because Danish fishermen are more or less the only one fishing commercially for sandeel in the North Sea. Economic data have therefore only been collected for Denmark.[2]

Economic data is collected at a yearly level, where the included vessels catch other species than the ones sought to be protected using marine protected areas. The biological part of the bioeconomic modelling will not include these other species, but they will be included in the economic part. The reason is that if they are left out, the included vessels will obtain a lower catch value, and thus have lower earnings, which may not be sufficient to cover the fixed costs. The catch of other species may however also change, dependent on the activity performed by the included vessels.

4.8.2. Building of a bioeconomic model

In order to consider the economic consequences of establishing marine protected areas, a bioeconomic model is being developed. The acronym of the bioeconomic model is BEMCOM (**B**io**E**conomic **M**odel to evaluate the **CO**nsequences of **M**arine protected areas), and it is programmed in GAMS (General Algebraic Modeling System).

BEMCOM is a flexible modelling framework, which is programmed in a generic way in order to be able to handle different case studies reflected through the utilised dataset and parameter values. Different management strategies can thus be investigated with respect to marine protected areas.

All model equations has been specified and programmed in GAMS, but some work is still required in order to adapt the general model framework to the two case studies i.e. case study 1 and case study 2, where BEMCOM is going to be used. For case study 2, several meetings have been held together with biologists in order specify the biological and economic interactions within BEMCOM. Based on the experiences obtained from case study 2, the expectation is that this will facilitate a more straightforward bioeconomic modelling of case study 1.

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[2] In 2003, Denmark had 94 % of the EU sand eel quota, and caught 92 % of the total landings weight taken by EU countries . In 2004, the figures were 97% and 89%, while in 2005 they were 98% and 100%.

4.8.6. Detailed descriptions of progress in work

With respect to the basic modelling framework, an unpublished working document has been prepared, which explains the basic variables, interactions and equations in BEMCOM, cf. **Annex 4**.

A more precise description of the biological equations is still needed in order to account for the complex biological relationships, which have a major influence on the economic outcomes. Issues such as the level of natural mortality and relationship between recruitment and stock level are two primary ones. It is also necessary to obtain information about the dispersion of fish from a closed area to the surrounding areas and the effect on the size composition of the fish stocks.

Case study 1 – Top-down controlled ecosystems in the Baltic Sea
Data collection in case study 1 covers several nations. Data covers vessels, which primarily catch cod in the Baltic Sea. The basic data has been collected for Denmark, but this still needs to be compiled and aggregated to the required levels, cf. **Annex 1**. Therefore, it is currently not possible at the moment to make a general description of the included data.

Case study 2 – Wasp-waist ecosystems (sand eel)
In order to parameterize the general modelling framework, economic data has been collected for the years 2003-2005. Only commercial vessels with at least 10% of their catch value consisting of sand eel were included in the dataset. In total, 120 vessels in 2003, 125 vessels in 2004 and 70 vessels in 2005 had at least 10% of the catch value consisting of sand ell. All these vessels are above 18 metres. From the Danish Fisheries Analysis Database (DFAD) information is obtained about the activity in form of number of days at sea, the catch compositions and the physical characteristics for all the included. However, due to confidentiality issues, it is necessary to aggregate the vessels into fleet segments.

Six fleet segments have been identified in form of: 1) trawlers 18-24 metres; 2) industrial trawlers 24-40 metres; 3) consumption trawlers; 4) mixed trawlers 24-40 metres; 5) industrial trawlers above 40 metres; and 6) mixed trawlers above 40 metres.[3]

Furthermore, yearly economic information has been obtained from the FOI Fisheries Account Database (FAD). FAD only covers a stratified part of the total number of vessels in Denmark, and therefore only around 50% of the total number of included vessels is represented in the economic dataset. Adaptation of the collected economic figures, so they can be used to reflect the vessels included in case study 2 has be done using the proportion between the average catch revenue in the Account Statistics and the one observed for the included vessels. This is of course an approximation, but the economic data has not been collected in order to specifically reflect the economic situation of the vessels included in case study 2.

Basic description of vessels included in Case Study 2

Data for the analysis conducted in Case Study 2 is obtained from two sources: 1) the Danish Directorate for Fisheries and 2) the Food and Resource Economics Institute in Denmark. The former has data related to vessel characteristics, activity and catches, while the latter collects economic data on a selected part of the commercial Danish fishing vessels.

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[3] A industrial trawler is defined as one with at least 80% of its catch value consisting of industrial species (sand eel, horse mackerel, sprat, blue whiting or Norway pout). A consumption trawler is on the other hand defined as one with at least 80% of its catch value consisting of consumption species, i.e. all species besides industrial species. Finally, the mixed trawlers are the ones not included in one of the other segments.

Based on the data from the Danish Directorate of Fisheries, 173, 199 and 97 vessels caught sand eel in 2003, 2004 and 2005, respectively. However, several of these vessels did only have sand eel as a minor part of their overall catches. It has therefore been decided only to include vessels for which sand eel comprises at least 10% of their overall catch value in the bioeconomic model. The number of vessels was thus reduced with around 30%. The included vessels catch the major part of the Danish catches of sand eel in the North Sea. In 2003, 2004 and 2005 the share of total sand eel catch value was for these vessels 94%, 92% and 91% respectively.

The economic data is for the Danish vessels compiled in order to reflect the economic situation of the complete Danish fleet. However, in this analysis, we only use a selected part of these vessels. It is therefore necessary to adjust the cost figures. This is done by scaling each of the cost figures. As scaling factor, the proportion between the average revenue in the Danish Fisheries Account Statistics and the average revenue observed for the vessels included in the present analysis is used for each fleet segment.

4.8.4. Possible deviations from project setup and work plans

Generally, the bioeconomic modelling is considered to be going forward as planned in the PROTECT timetable. Data collection is completed for Denmark, but it needs to be worked through and described for case study 1. For the next half year, the bioeconomic modelling framework will be finalised and parameterized. Modelling results and preliminary conclusion will be compiled into at least one paper covering case study 2, which will be presented at the MPA conference in Murcia, Spain in September. Case study 1 will be postponed until the primary part of case study 2 has been completed. No problems are at the moment expected with respect to finalisation of the project.

4.9. MPA-induced fleet movements of the Baltic cod, herring and sprat fishery: Results based on a Probabilistic neural network model.

Pekka Jounela, FGFRI

The Baltic CS evaluates MPAs' potential to protect top-down controlled ecosystem from the adverse effects of fishing. Hence, it is fundamental to know fishers' location choice with respect to MPAs. Below is presented an overview of models used to predict fleet-movements, followed by initial results and a preliminary discussion of their implications for MPAs.

Modelling approach

Fleet movement models describe socio-economic implications of MPAs enforced in the Baltic Sea Main Basin in 1996 - 2005. Fleet movement models investigate fishers' temporal and spatial adaptation to MPA regulations imposed in the Baltic Sea Main Basin in 1996 – 2005. Danish, Polish and Swedish economic and catch-effort logbook data from the cod, sprat and herring fishery are analysed using two different approaches: a non-parametric discrete choice model (PNN, probabilistic neural network; Specht, 1990) and a parametric discrete choice model (RUM, random utility model; Holland and Sutinen, 1999). The assembled logbook database consists well over 1.5×10^6 rows of input data for the trip-by-trip based fleet movement analyses. The outcome and performance of these two statistically different model types are compared with each other. The main issues addressed in the fleet movement modelling are:

- Identification of MPA-induced fleet movements in the Baltic Sea cod, sprat and herring fishery
- Identification of MPA induced compensation actions of fishers'
- Identification of most important factors affecting fishers' location choice
- Identification of socio-economic implications of MPAs enforced in the Baltic Sea

A general description of the two fleet movement models is described below. A more detailed statistical description of the models is found in the papers of Specht (1990) and Holland and Sutinen (1999). It is of note that the fleet movement analyses also provide input for the TEMAS and ISIS-Fish models.

PNN model

The PNN is a neural network classifier introduced by Specht (1990). The theoretical framework for the Probabilistic Neural Network (PNN) is an old but very powerful theory of how to build classifiers, based on Bayes, and termed "Bayesian Classification" (see Duda and Hart, 2001). In general, the PNN procedure includes cross-validation i.e. in the testing phase of the network is provided with unseen logbook data (i.e., logbook data that were not used in its training phase) and the network is providing an answer (prediction) for the class (ICES-square) to which these unseen data most likely belong to. By this way, the model parameters and the output as well are not blind with respect to data from which it was optimized. In PNN model, ICES rectangle is treated as a categorical dependent whereas the continuous independents are Julian day (from day 1 to 365), distance from the fishing location to the landing port (km), catch (kg), effort (min), vessel's length (m), engine power (kW), and weight (GRT). The categorical independents are homeport, vessel type, gear type, and target species (cod, sprat and herring). These factors are the usual ones applied in the almost all other fleet movement models.

The PNN based fleet movement analyses were done on a trip-by-trip basis and by using ICES-square of each trip as a dependent variable i.e. the output was non-parametric Bayesian posterior probability density of fishers' location choice in the Central Baltic Sea cod, sprat and herring fishery.

Random utility model

The other fleet movement model used in the Baltic Sea case study is a classic random utility model (RUM). In general, it is a traditional parametric nested logit model aiming to predict MPA induced redistribution fishing effort by the expected returns to individual fishers from the alternative fisheries and locations. The output of the model is the parametric probability density of fishers' location choice. A detailed statistical description of the RUM model is found in the paper of Holland and Sutinen (1999). At the moment, the parameterization of the RUM model is ongoing and results will be presented during the latter stage of PROTECT.

Results and preliminary discussion of their implications

We studied Swedish and Polish cod, herring and sprat fishers fleet movements in relation to MPAs enforced in 1996 – 2005 using Probabilistic neural networks. Catch-effort logbook data consist of 1.5×10^6 logs and analyses were done on a trip-by-trip basis. The study area covered the whole Baltic Sea Main Basin (Figure 4.9.1).

The long-term cod fishers' fleet movement patterns reflect poor status of the eastern Baltic cod stock during the last decade. During 1996-2005 the eastern Baltic Sea lost its economic importance for the cod fishers and thereby the average steaming distance in between the landing ports and fishing grounds reduced approximately by one half (Figure 4..9.2- 4).

Our fleet movement analyses implied major MPA induced forth-and-back fleet movements in between the eastern Baltic and the western Baltic Sea. This forth-and-back fleet movement appeared to be dependent on the prevailing MPA regulations enforced in the eastern and western Baltic Sea. That is, protection of cod using MPAs in the eastern Baltic Sea increased effort towards the western Baltic cod stock, and vice versa (note that a temporal spring ban for cod targeted fishing was introduced in 2005 in the Western Baltic Sea and was enforced from 1st March to 30th April, Figure 4.9.3 and 4.9.4).

To be effective in increasing the absolute and relative size of the eastern Baltic cod stock in relation to sprat and Baltic herring stocks, MPAs should be enforced in the areas having high cod targeted effort and/or high reproductive potential of cod. The fleet movement results implied that Swedish and Polish cod targeted fishery was not active in the Gdansk Deep and in the Gotland Deep (Figures 4.9.4 and 4.9.7), which were protected year-round using MPAs in 2005. These MPAs actually displaced and reduced effort of pelagic fishery thereby protecting sprat and herring stocks from effective exploitation (Figure 4.9.8-12). This is somewhat contradictory with respect to goals of MPAs enforced in the Baltic Sea i.e. rebuilding a cod dominated ecosystem using MPAs in the Baltic Sea.

To be effective in reducing the by-catch of cod in the pelagic fishery, overlapping areas of high cod abundance and active sprat/herring fishery should be protected using MPAs. Although the main pelagic Swedish fishery occurs in the eastern Baltic Sea, substantial pelagic effort was observed also on the western side of the Bornholm MPA during the spring (Figure 8 and 9). This implies that by-catch of cod could be high in the more shallow waters west of the Bornholm MPA.

Policy makers are often interested not just in the size of economic gains and losses but also in how those gains and losses are distributed within society. Our fleet movement results suggest that in 2004 (and 2005) the enlarged Bornholm Deep MPA led to unevenly redistributed economic benefits in between Swedish and Polish fishers. After the Bornholm MPA was enlarged to the east, Polish fishers did not steam northwards to the eastern and northern side of the Bornholm Deep that used to be the high catch area for them (Figure 4.9.12). That is either due to direct MPA induced effort displacements in the eastern Bornholm Deep or due to too long steaming time to the northern side of the Bornholm MPA. On the other side of the MPA, Swedish fishers did not steam southwards to the eastern side of the MPA anymore, which otherwise used to be a high catch area for them (Figure 4.9.3 and 4.9.4). That is, a number of Swedish and Polish demersal trawlers and gillnetters were induced economic losers due to temporally enlarged closure area. In addition, the excess "race to fish" a global TAC increased in the remaining open areas and seasons. On the other hand, the excess "race to fish" a global TAC decreased between Swedish and Polish fishers. This separation of two national fleets does not, however, suggest any protection value for the cod stock.

To be effective in increasing rule compliance of fishers, MPA regulations should be designed in concert with other fisheries management measures. In 2005, the MPA regulations were year-round in the three Deeps and the summer ban was extended as well (Figure 4.9.4). However, cod targeted harvests were legal also during the summer spawning months i.e. the regulations permitted 200 kilos of cod landings as "by-catch". Compared to earlier years, this combination of technical regulations induced increased cod targeted effort during the summer spawning months (Figure 4.9.3, 4.9.4 and 4.9.7), which in turn implied increased misreporting of catches. It suggests that the very restrictive regulations enforced in the 2005 were not economically sustainable in the most fishery-reliant communities and hence the circumvention actions during the summer season.

The magnitude of MPA induced effort displacements was extremely variable between different fleets. In general, fishers location choice was primarily determined by the distance in between the fishing ground and landing port as well as vessel's characteristics. This suggests that MPAs nearby most fishery-reliant communities should be avoided.

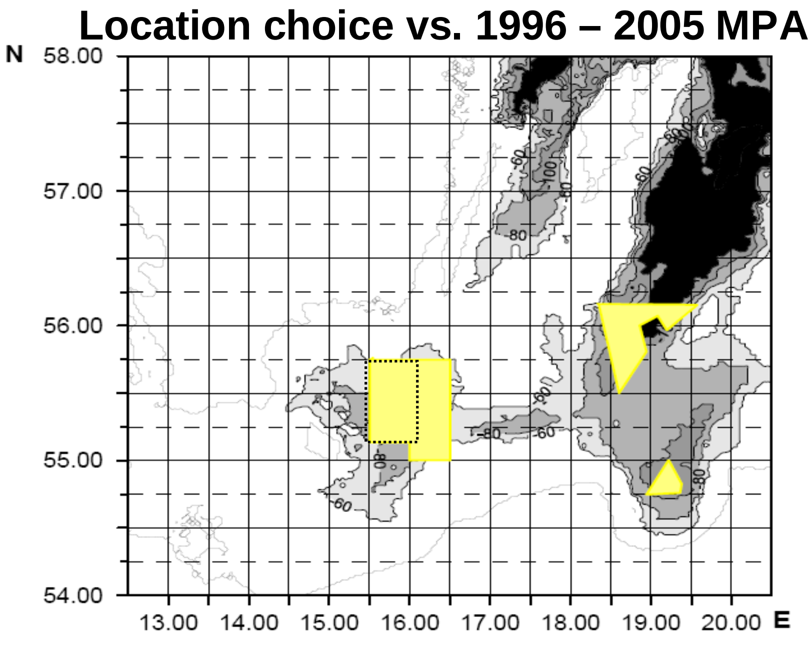


Figure 4.9.1. The year-round MPA network for all fishing enforced in the Baltic Sea in 2005 (light yellow) and a temporal MPA for all fishing enforced from 1997 – 2004 (dark yellow). In addition, in the eastern Baltic Sea cod targeted harvest were prohibited during the summer ban in 1995-2005 and, in the western Baltic Sea a winter-spring ban for cod targeted harvests was introduced in 2005.

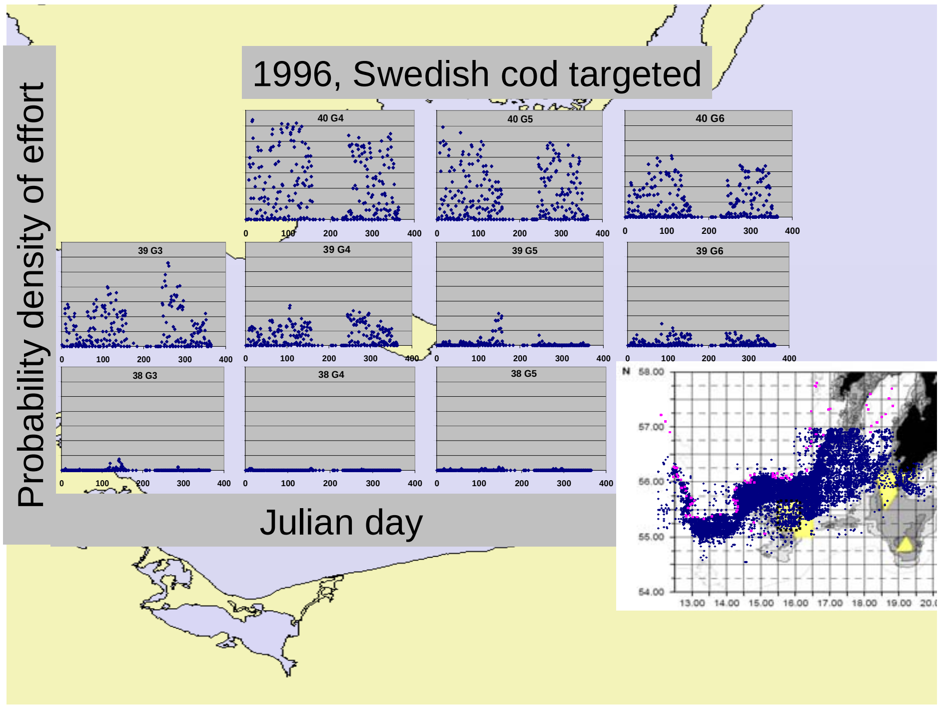


Figure 4.9.2. The probability density of Swedish cod fishers' location choice in 1996 (Julian days 1-365, large figure). The spatial distribution of Swedish cod fishers' location choice in 1996 in relation to year round MPA network enforced in 2005 (small figure).

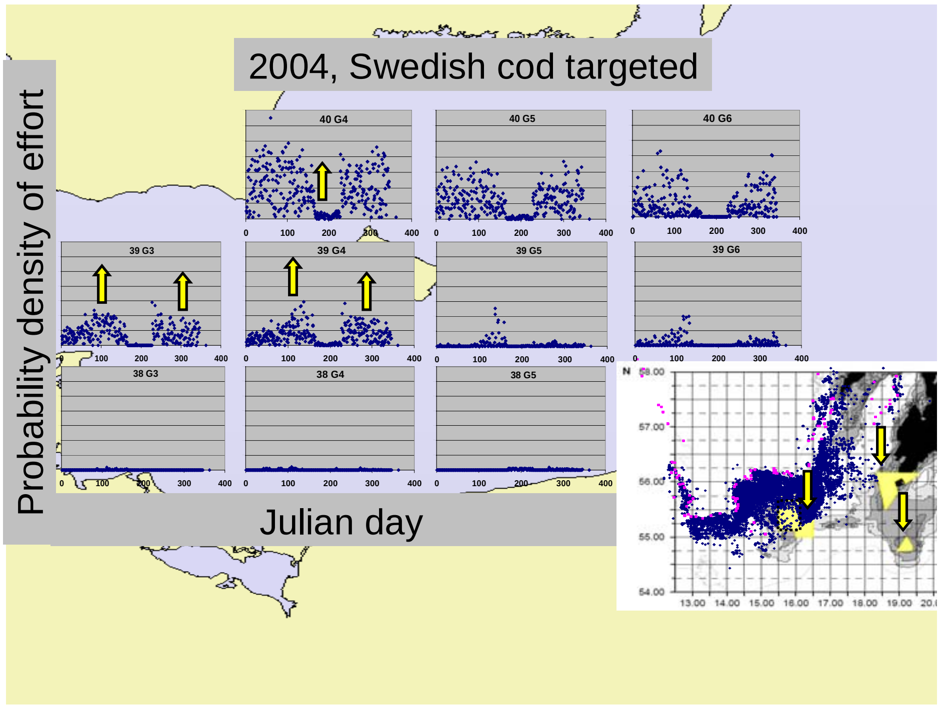


Figure 4.9.3. The probability density of Swedish cod fishers' location choice in 2004 (Julian days 1-365, large figure). The spatial distribution of Swedish cod fishers' location choice in 2004 in relation to year round MPA network enforced in 2005 (small figure).

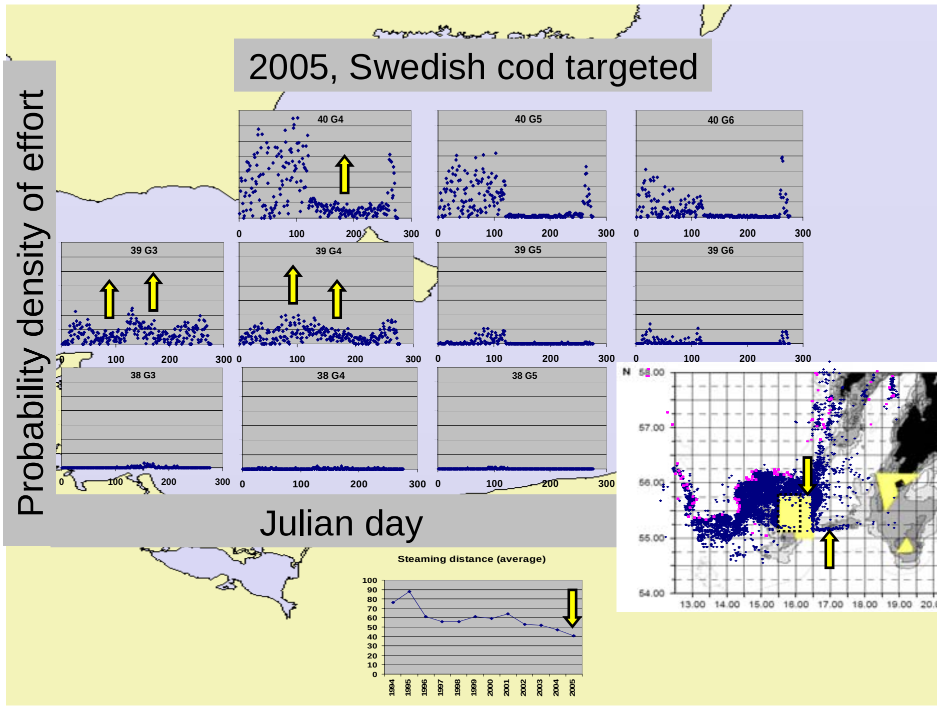


Figure 4.9.4 The probability density of Swedish cod fishers' location choice in 2005 (Julian days 1-365, large figure). The spatial distribution of Swedish cod fishers' location choice in 2005 in relation to year round MPA network enforced in 2005 (small figure).

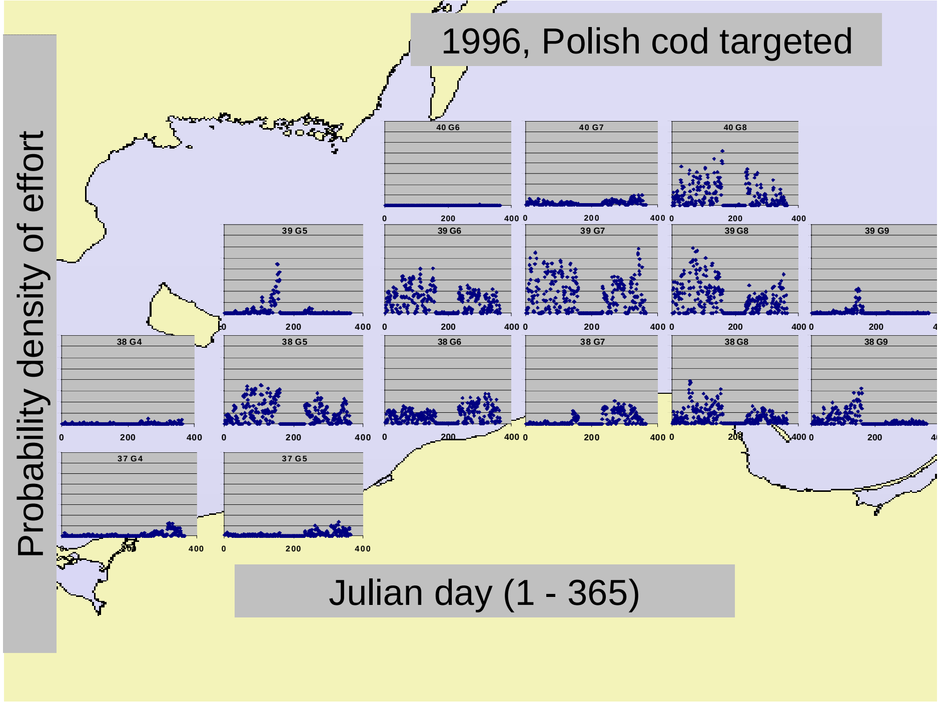


Figure 4.9.5. The probability density of Polish cod fishers' location choice in 1996 (Julian days 1-365).

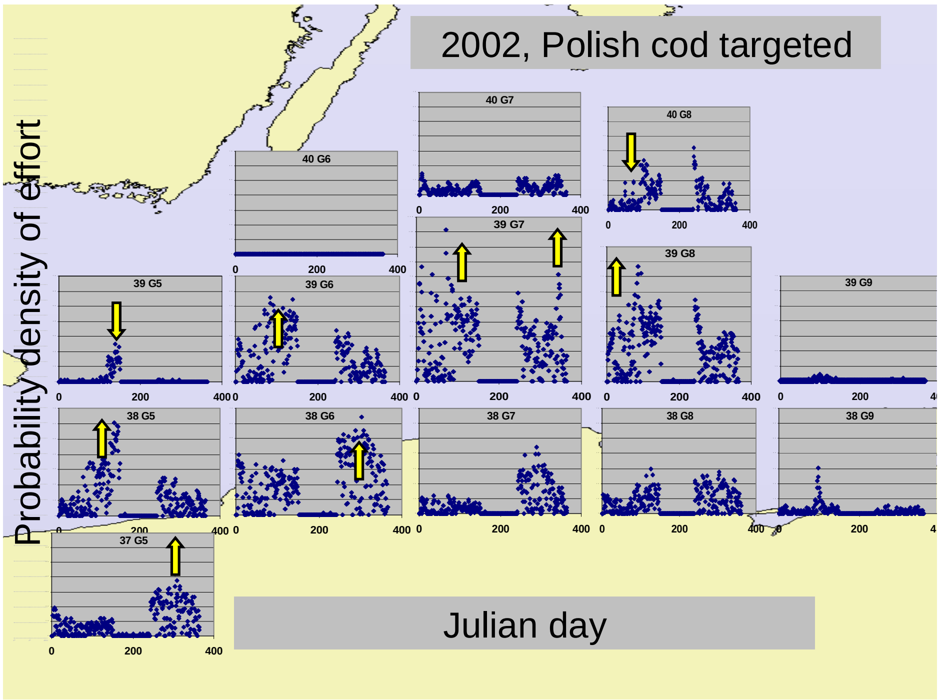


Figure 4.9.6. The probability density of Polish cod fishers’ location choice in 2002 (Julian days 1-365).

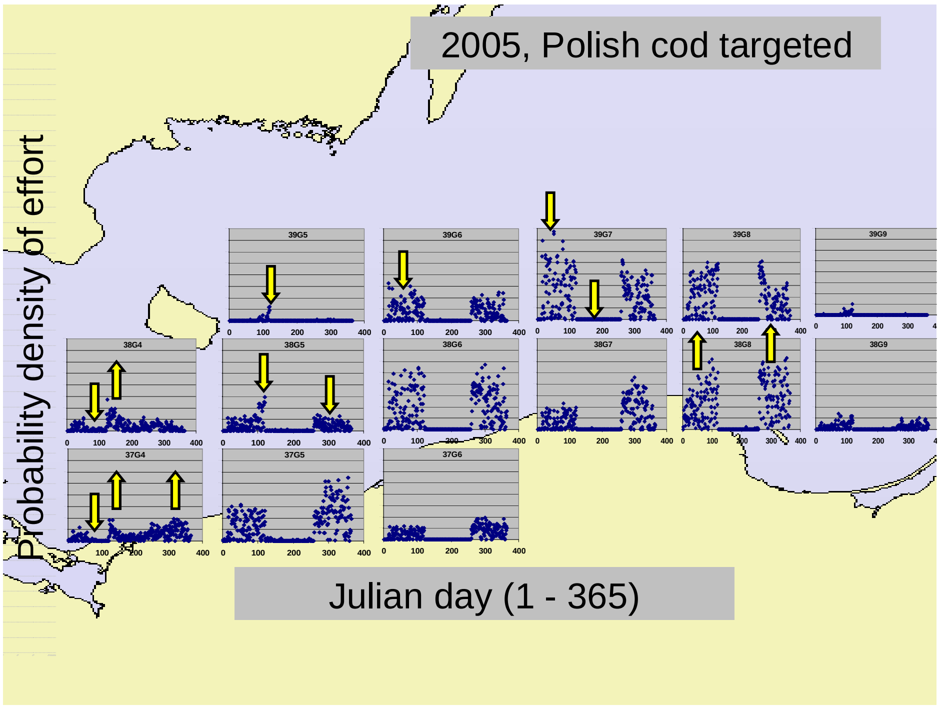


Figure 4.9.7. The probability density of Polish cod fishers’ location choice in 2005 (Julian days 1-365).

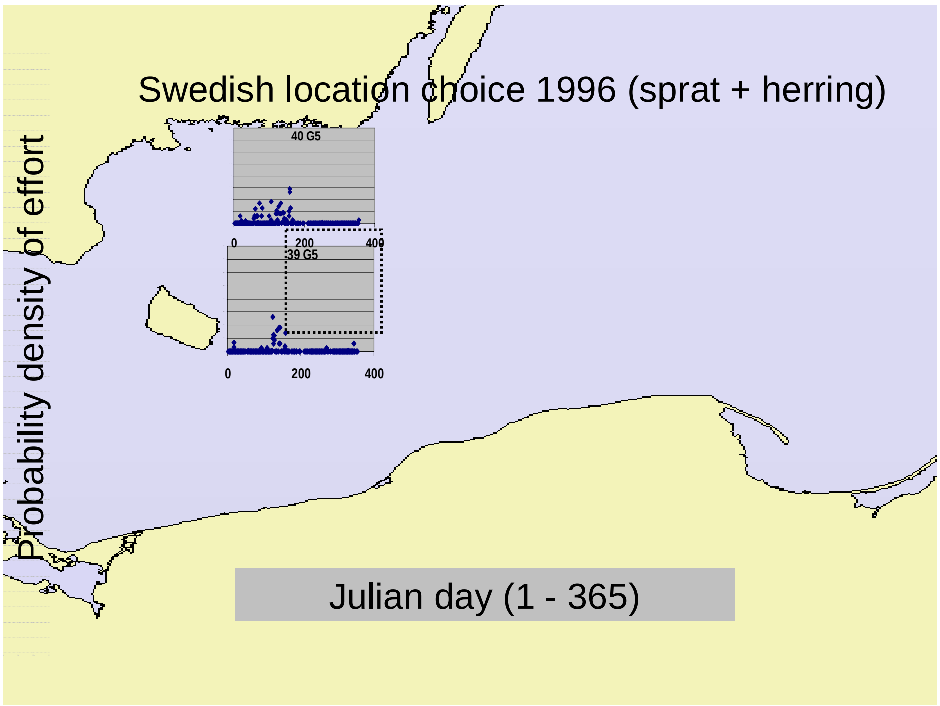


Figure 4.9.8. The probability density of Swedish sprat and herring fishers' location choice in the region of Bornholm MPA in 1996 (Julian days 1-365).

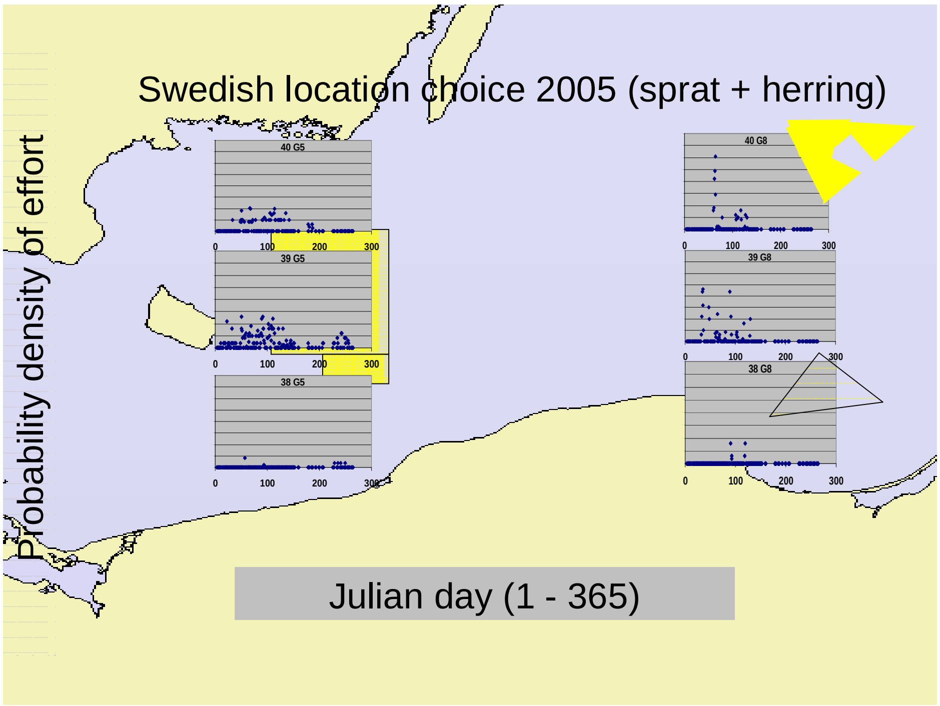


Figure 4.9..9. The probability density of Swedish sprat and herring fishers' location choice in relation to year round MPA network enforced in 2005 (Julian days 1-365).

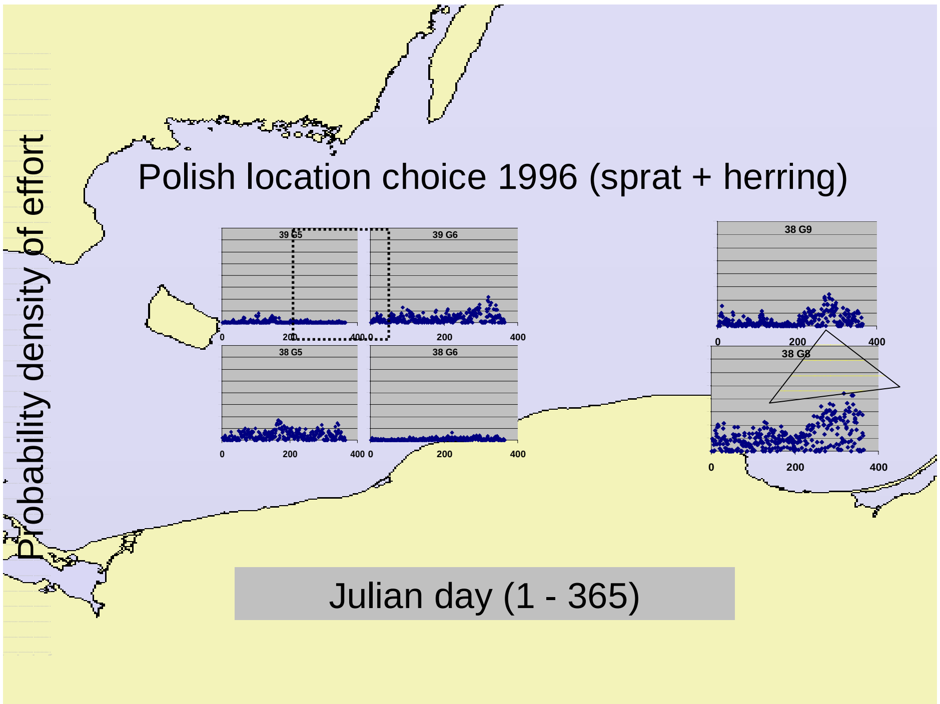


Figure 4.9.10. The probability density of Polish sprat and herring fishers' location choice in 1996 in the region of Bornholm MPA and Gdansk MPA that was enforced later in 2005 (Julian days 1-365).

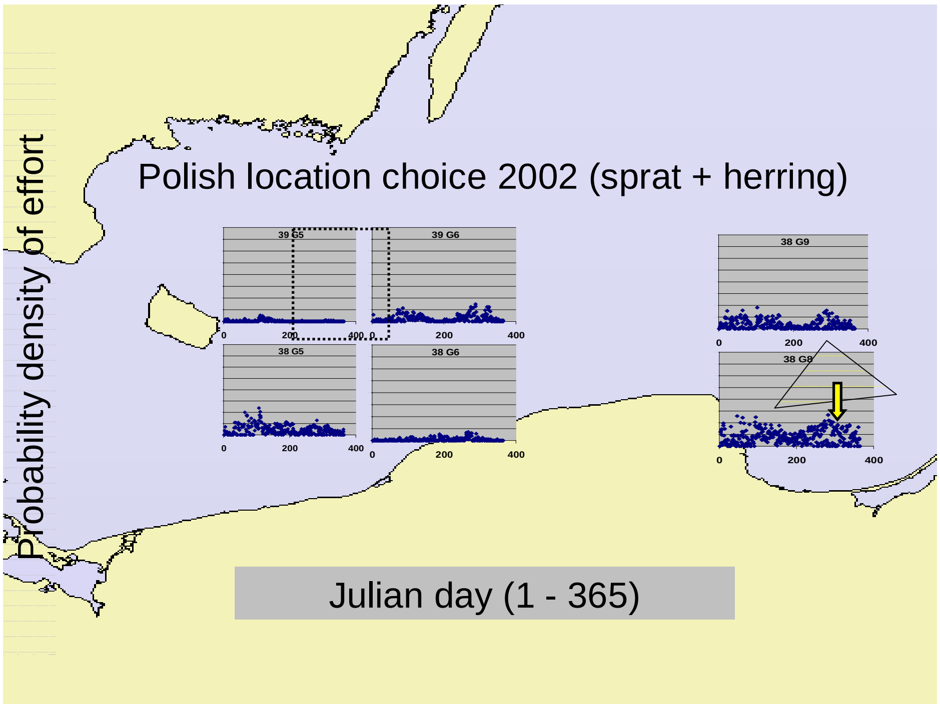


Figure 4.9.11. The probability density of Polish sprat and herring fishers' location choice in 2002 in the region of Bornholm MPA and Gdansk MPA that was enforced later in 2005 (Julian days 1-365).

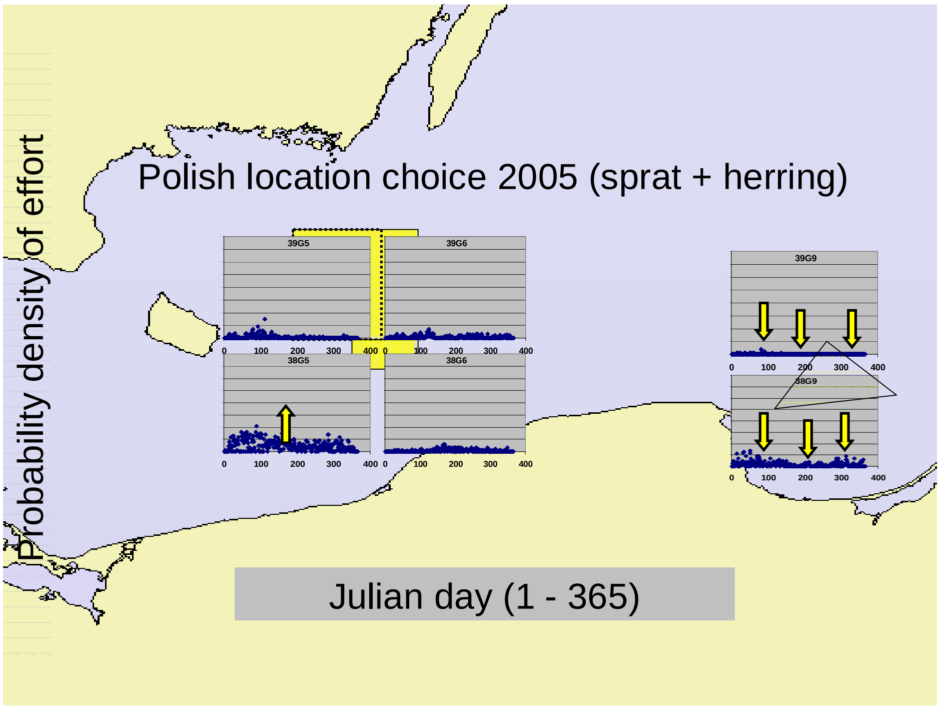


Figure 4.9.12. The probability density of Polish sprat and herring fishers' location choice in 2005 in relation to year round MPA network enforced in 2005 (Julian days 1-365).

References

1. Duda R.O., Hart, P.E., Stock, D.G. 2001. Pattern classification, John Wiley & Sons Inc., 2nd edition.
2. Holland, D.S., Sutinen, J.G. 1999. An empirical model of fleet dynamics in New England trawl fisheries. Can. J. Fish. Aquat. Sci. 56: 253–264
3. Specht, D. F., 1990. Probabilistic Neural Networks, Neural Networks, Vol. 3, pp. 109-118.

4.10 Deviations from the modelling work programme, Baltic Case study

Generally, the Baltic case study modelling work is considered to be going forward as planned in the PROTECT timetable. The hydrodynamic, single species, MSVPA and PNN fleet movement modelling has been done and parameterization of the TEMAS, ISIS, RUM and BEMCOM models are underway. BEMCOM modelling of the Case study 1 will be postponed until the primary part of case study 2 has been completed. Economic data collection is completed for Denmark, but it needs to be worked through and described for case study 1. No problems are at the moment expected with respect to finalisation of the project.

Summary of modelling activities

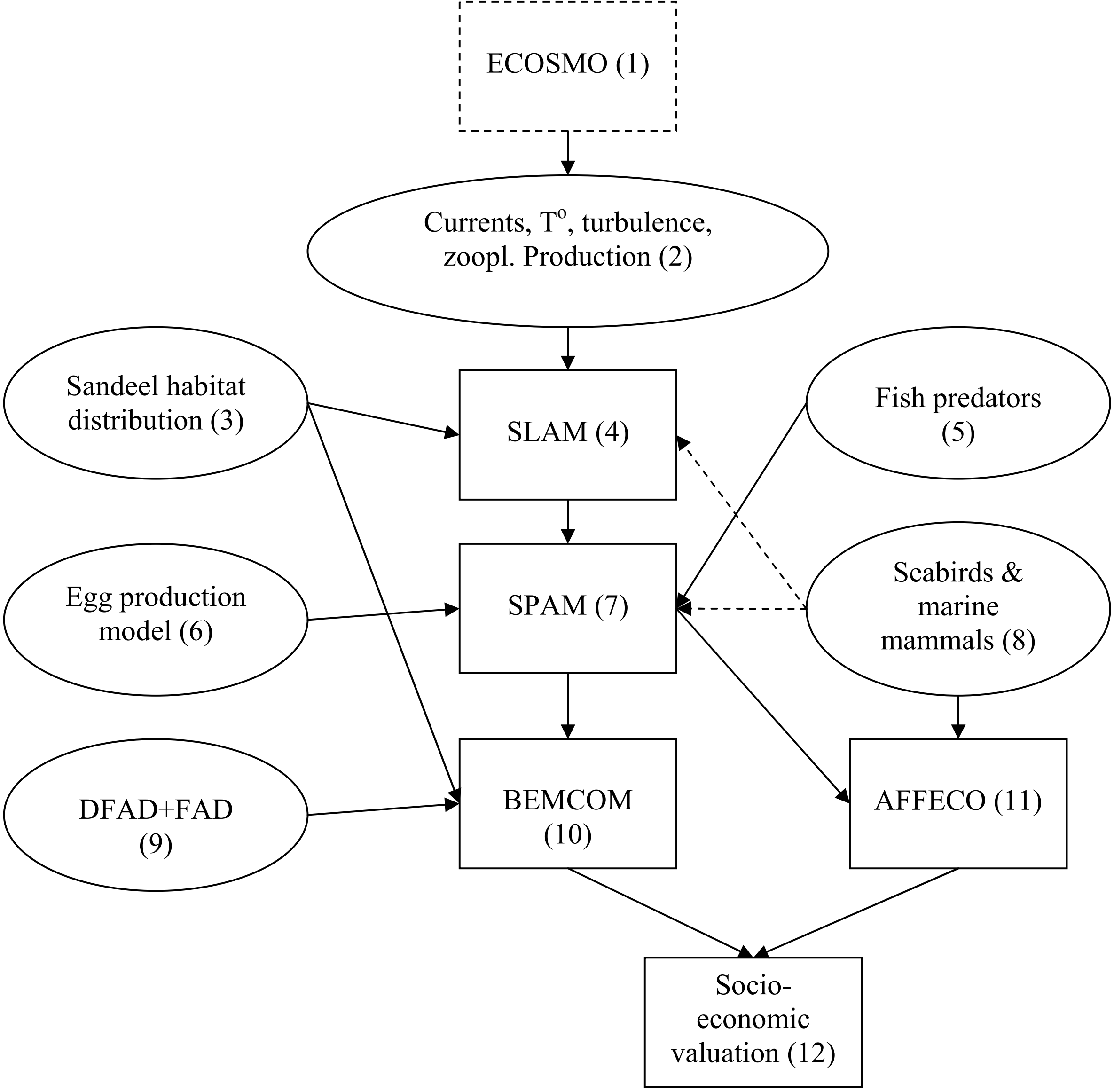
Cod egg and larval survival was modelled. Hydrodynamic modelling to describe transport and resolve physical environments was finalized. Identification and evaluation of size and effective contribution of spawning areas was finalized. Lipid analysis of cod and clupeids was conducted. The area-disaggregated MSVPA analyses were done. Distribution patterns of cod and sprat based on hydro-acoustic surveys were analysed. A single species population dynamic modelling considering environmental scenarios and management options was done that consists of: (1) Development of a spatially disaggregated, discrete time, age-structured model of the population dynamics of the Eastern Baltic cod stock, (2) A 50-years simulation analysis, projecting the potential development of the cod stock size, yield, revenues, and costs for the fishermen under various management policies and environmental change scenarios, (3) Development and investigation of management policies for stock recovery. A preliminary bio-economic modelling framework of the potential Baltic Sea MPAs was formulated. MPA induced fleet movements of cod, sprat and herring fishery were assessed.

5. Progress in work: North Sea Case Study

Henrik Mosegaard, DIFRES

5.1 Overview of progress, modelling linkages, preliminary results, and future needs in continuation of WP5 activities

The North Sea sandeel CS (2) has shown progress in several aspects of the modelling package of the project. First of all the flow of information and interfacing of model components has been established. The diagram below shows the major model components with their most important links.



(1) ECOSMO as described in the literature (Schrum et al. 2006), has delivered on a stratified grid scale (2) current flow fields, temperatures, turbulence and secondary production (see SLAM 2.2). (3) Based on an

array of all established habitats for adult sandeels the (5) SLAM component has used an approach of IBM larval drifters and produced a transportation probability matrix among all the individual sandeels banks.
(7) The population model SPAM simulates growth, and with input of (6) egg production from a size, age and area resolved maturation and fecundity equation. SPAM then initiate density distributions of larval survivors to be multiplied into the transportation matrix. Resulting recruitment to a specific bank is a function of year y÷1 density distributions of egg-production, transportation probability, and density dependent larval survival. The bank specific population numbers at age and size are exponentially decreasing as a function of fishing-, predation- and natural mortality. Fishing pressure is either taken as an observed quantity or as an output from BEMCOM (10). Predation mortality is dominated by fish (5) but sandeels are also to a lesser degree eaten by seabirds and marine mammals (8).

BEMCOM gets its economic data (9) from the Danish Fisheries Analysis Database (DFAD) and from the Fisheries Account Database (FAD). Since the SLAM SPAM components are constructed to be generic and generally flexible it was decided that these are to be fit into the BEMCOM fish population module to deliver sandeel stock dynamics for the fisheries exploitation in the bio-economical modules. Further it has been decided that a version of SPAM will be constructed to interact with novel formulations of general seabird community reactions to local sandeel availability in scenario modelling in AFFECO (11). Finally the scenario modelling of MPA closures from BEMCOM and the resulting effects of sandeel availability on seabird performance from AFFECO will be analysed in respect to the society values (12) concerning preferences and priorities of resources and ecosystem attributes.

5.2 Bio-physical coupled Sandeel Larval Advection Model SLAM

Asbjørn Christensen, Henrik Mosegaard and Henrik Jensen, DIFRES

SLAM is coupling outputs from ECOSMO (Schrum et al. 2006) of North Sea currents, turbulence, temperature and zooplankton production and biomass with an IBM sandeel egg and larval stage development and drift model as indicated in Figure 5.2.1.

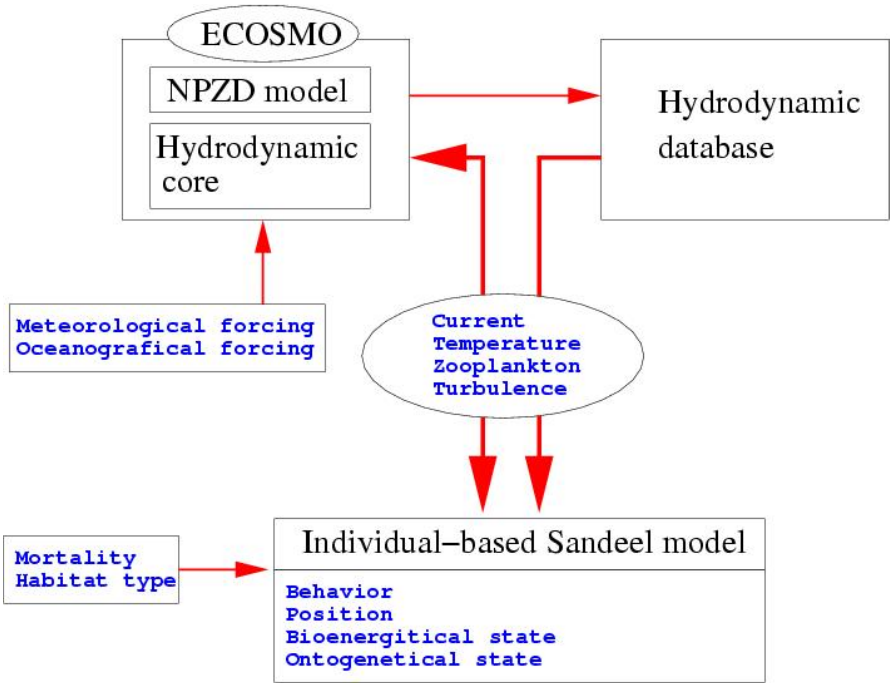


Figure 5.2.1: Overview of the SLAM-model.

The potential egg producing habitats are closely connected to a mosaic of sandbanks with specific characteristics (see section 3 on “Knowledge required to model and evaluate MPAs“). Figure 5.2.2 shows a map of the present perception of North Sea sandeel habitats. The map is constructed by combining Danish fishers’ historic records of sandeel catch locations with information on bathymetry and bottom substrate, as such the map is only a minimum of possible habitats.

Based on the overlap between the hydrographical grid and these habitats a number of locations have been defined as start and ending points for larval drift.

For further details on this study, please refer to **Annex 5.**

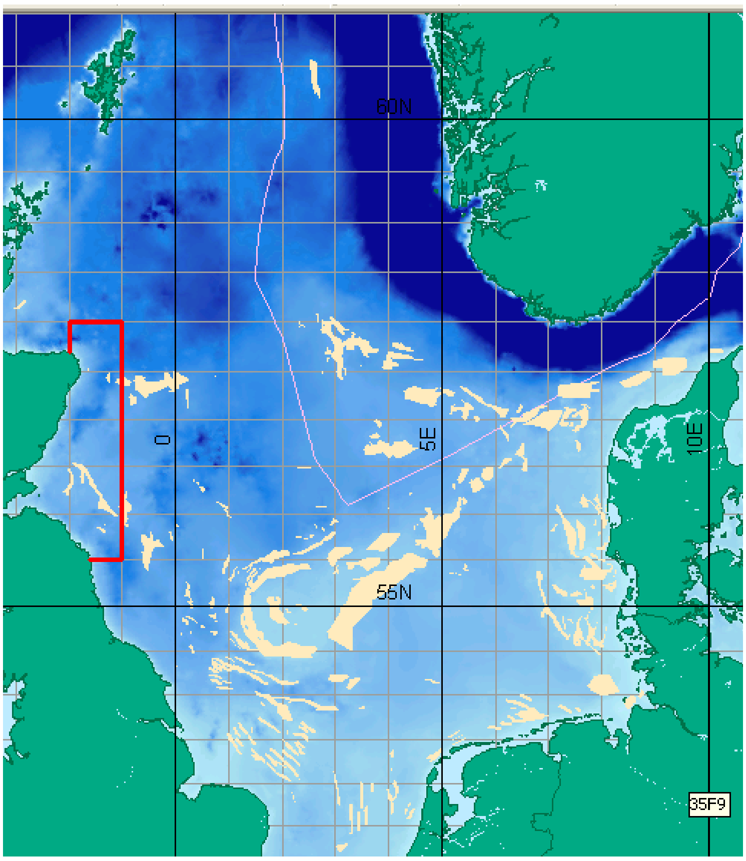


Figure 5.2.2. Map of North Sea potential sandeel habitats (yellow) with an indication of the closed area of Firth of Forth.

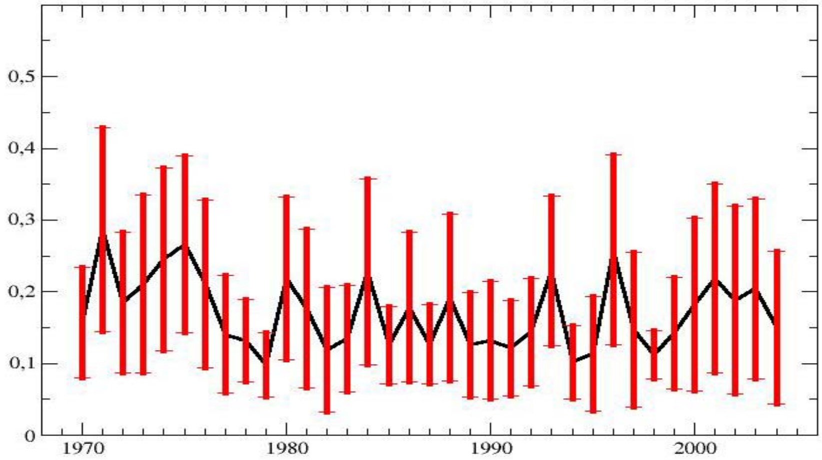


Figure 5.2..3 Interannual variability in transport survival σ for all North Sea banks. Average transport survival is indicated by the full line. Bank variability for a given year (referred to bank division in colours) is indicated by red bars (Model cx_h20febw01_crw)

5.3 Report on the Sandeel Population Analysis Model, SPAM

Asbjørn Christensen and Henrik Mosegaard, DIFRES

Population analysis is performed on a bank specific basis where the transportation probability matrix from SLAM deliver the basics for recruitment at age 0, and the bank specific population component is considered resident within the closed system of the bank boundaries.

Recruitment to the bank population is further assumed density dependent with exponential mortality coefficients for drifting larvae as well as settling larvae calculated from the area based total biomass in relation to area carrying capacity.

The major state variables of the model are the cohort abundances represented as numbers at age and size at each specific bank. These numbers are assumed to be exponentially declining due to natural- predation- and fishing- mortalities on a sub yearly time scale (e.g. weekly).

Growth is assumed density dependent with an individual based stochastic L-infinity and a k dependent on total biomass in relation to carrying capacity. The growth-density response is parameterized as $\lambda(\rho) = \lambda_0 \cdot \exp(-\lambda_i(\rho-1))$ i.e. same relative response over all length classes, assuming all sandeels to compete for the same resources. Resources are modelled in SLAM and transferred by size and age dependent bioenergetics to the body constituents protein and lipids (oil).

Maturity is logistically dependent on length with an area effect. The fecundity relationship to sandeel length $F = a \cdot L^b$ is at present taken from Macer (1966) but will eventually be replaced by a size, age and area based equation being developed by FRS.

A central output of SPAM is the catch, resolved on a bank level. Because mortality is high and growth significant along the season, it is important to include these factors when assessing the catch from the model. For each catch day the population length distribution is calculated and the harvested mass is transformed into proportion active sandeel mass at age and length which is then subtracted from the total age - length distribution. With a given effort the CPUE will be a function of size-, age- and time- dependent sandeel activity level times the remaining bank specific biomass. The accumulation of protein and oil in the growth equations constitute the major value components of the catch and will feed into the earning parts of the bio-economic model BEMCOM.

SPAM will be used in combination with SLAM or an average transportation matrix with stochastic variation both to analyse the time series of sandeel population dynamics and catches at the highest possible level of resolution as well as in scenario modelling with MPA closures at different temporal and geographical scales.

Figure 5.3.1 demonstrates the present SPAM β-version output of sandeel population dynamics as total biomass and spawning stock biomass in each of the five major areas as defined in section 6.6.1. An uncomplicated setting has been applied where an average transportation matrix with and without stochastic variation drives dispersion and recruitment starting from low population abundances and allowing for a constant fishing mortality corresponding to the last three years average from the latest ICES stock assessment.

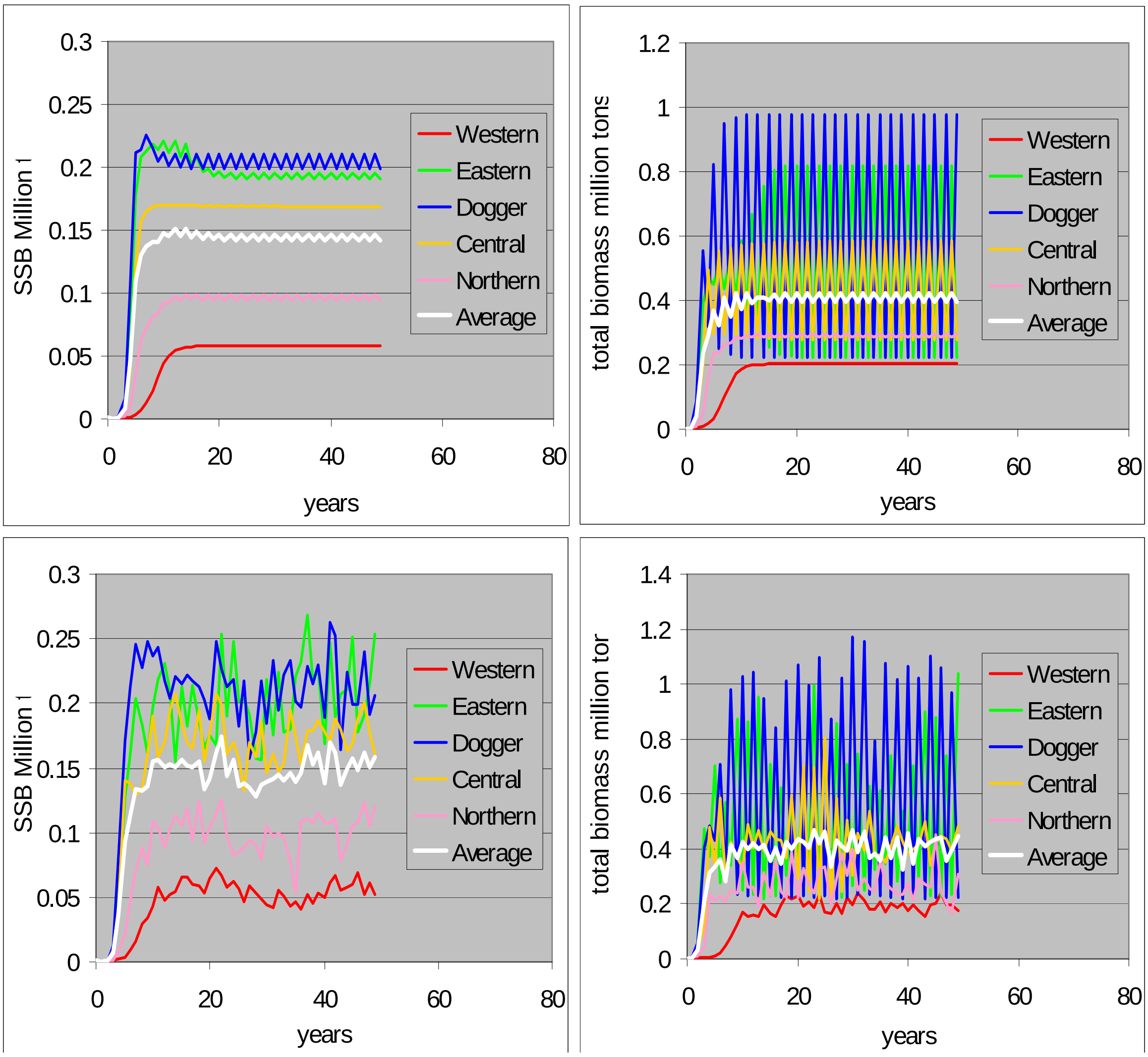


Figure 5.3.1: SPAM ß-version output of sandeel population dynamics as spawning stock (left) and total biomass (right) without variation in transport (upper) and with stochastic variation in transport for each of the five major North Sea areas (plus the average).

5.4 Bio-physical coupled HAMSOM - Sandeel drift Model

Alejandro Gallego, FRS-MarLab

5..4.1. Overall status of progress

Work in 2006 has centred on finalizing the bio-physical modelling code specific to a new fine scale version of the HAMSOM hydrodynamic model and carrying out some preliminary simulations. Although the bulk of the modelling work has been completed and tested, the spatially resolved egg production estimates are not yet available so the fine scale simulations and the coarser scale simulations will have to be somewhat delayed until egg production estimates are available in early 2007.

5..4.2. Detailed descriptions of project results and outputs

To simulate the relative survival of the spawning products of the North Sea (Scottish east coast) sandeel sub-stocks we used the output of a new version of HAMSOM (Logemann, K., Harms, I. (2006): High resolution modelling of the North Icelandic Irminger current (NIIC). Submitted to Progress in Oceanography) available for in two recent years (2002 and 2003), plus a “climatological” year. The coarse base matrix of this model has a horizontal resolution of approximately 150 km in the subtropical Atlantic Ocean and 75 km in the Arctic Ocean. This model covers the whole North Atlantic in a spherical coordinate system and the Arctic Ocean in polar stereographic coordinates. Six sub-domains were nested into the coarse domain, at increasing resolution to a maximum 1.5 km eddy-resolving fine scale in coastal waters. The year-specific runs used atmospheric input data from the NCEP and ECMWF databases and included full thermodynamic exchange at the sea surface based on bulk formulae and freshwater fluxes resulting from precipitation, evaporation and actual river runoff data from Scotland. The climatic year was the result of a 10 year climatological run performed on the coarse base grid and continued for another 2 years on the fine resolution grid. The bio-physical simulations used the horizontal and vertical velocities, as well as the temperature fields, simulated by the hydrodynamic model.

Following hatching, zero-drag particles are passively transported by the flow in the horizontal plane (including M2 tidal velocities). Horizontal diffusion velocities can be applied to particles in the bio-physical simulations but this feature can be switched off if necessary. Vertical transport is not passive but the result of an ontogenetically-varying vertical migration pattern derived from the analysis of the published location of sandeel larvae in the water column, although vertical HDM velocities are applied to particles tracked in the model. Particles are tracked with time steps of 10 min, and particle positions are saved at daily intervals. Each particle represents a super-individual i.e. an assemblage of identical individuals with an initial weighting that gets reduced as a function of mortality. Only pelagic individuals are tracked, so particles remained on a fixed position once they reach the settlement stage. No grounding of particles is permitted. The particles are released as larvae (following a temperature-dependent demersal egg period) of a given length at hatching, and their length is updated following a temperature-dependent growth function (which will be possibly updated as a temperature and food growth function in the future, on the basis of ongoing statistical modelling of sandeel growth). Once the larvae reach the settlement length, they are no longer pelagic and remain “anchored” to their 2-d position. Initially, all larvae are allowed to settle once they reach settlement length, regardless of the suitability of the substrate and other spatial characteristics (e.g. water depth) but it is possible to vary the settlement scenarios to restrict settlement to suitable areas (banks) exclusively.

5.4.3. Possible deviations from project setup and work plans

The simulations planned for Nov. 2006 and Jan. 2007 have been delayed slightly by a delay in the availability of egg production data. As soon as these are available, the bio-physical simulations will be carried out (first quarter of 2007). Test simulations carried out with a version of the model parameterised for cod early life stages (where full forcing datasets were already available) demonstrated that the model configuration works properly.

5.5 Individual-based model of sandeel and predator interactions in the North Sea

John Pinnegar and Ewen Bell, CEFAS

Extensive effort has been dedicated to constructing an Individual-Based Model (IBM) of interacting sandeel and predator populations in the central North Sea, as part of case study 2. These models, will be used (in year 3 of PROTECT) to evaluate a range of potential management options including permanent and rotational fisheries closures. This work is co-supported through a parallel project funded by the UK government. The resources from PROTECT have been used to provide model documentation, in anticipation of a detailed manuscript submitted to the PROTECT/EMPAFISH symposium in September 2007.

Field data recently collected (and available to other PROTECT partners) includes information collected between 2004 and 2006 (8 research cruises) on the Dogger Bank. The western Dogger Bank has recently been nominated (by the UK) as a candidate Special Area of Conservation (SAC) under the EU Habitats Directive. A number of other EU member states are also looking into potential offshore closed areas in this region. In the case of Germany, for example, this includes an area known as the Weisse Bank and part of the eastern edge of the Dogger Bank.

The multispecies model developed is a radical departure from those previously used within the ICES community. Traditional models such as MSVPA (MultiSpecies Virtual Population Analysis) are array-based, each array holding the numbers of individuals at a given age. For a given species and age, each individual is assumed to be of average size and has the same dietary requirements and dietary consumption. Growth is assumed to be constant and therefore there is no effect upon the population trajectory of food limitation (i.e. animals do not starve). Dietary composition is determined by “suitability” which is a fixed parameter for each combination of age classes of predator and prey, therefore a cod age 6 will have a set preference for whiting age 2. The parameterisation of this type of model is heavily data dependent and relies upon the ICES “Year of the Stomach” sampling programmes conducted in 1981 and 1991. Analyses of these data have shown that whilst the suitability parameters for some combinations of predator and prey remained constant between the two years, others varied considerably.

The modelling approach taken within this project is that of Individual Based Modelling (IBM). As the name suggests, the model tracks individual fish as they eat, migrate, grow and, ultimately, die. This type of model is significantly more flexible than ‘array-based’ models and can mimic nature more closely, however this flexibility and realism comes at the expense of computational time which is vastly increased.

The model has been constructed using C++, an object orientated programming (OOP) language. OOP is ideal for individual based modelling as each individual is held in computer memory as a separate object. The model has a number of basic object types which are described below.

1. Fish. These are the basis for the whole model and an implementation can generate as many fish objects as is required subject to the memory capacity of the computer. In order to conserve computer memory, the amount of information each fish carries with it must be kept to a minimum. Individual fish know what species they are, how old they are, what size they are and where they are.

2. Species. This object carries all the parameters which control the behaviour of the individual fish. These behaviours include spawning,, growth, feeding and migration. Each species carries a list of prey types thus enabling the model a basic level of selectivity and restrict the dietary range of predators to that which has been observed in the field data.

3. Shoal. Fish can group together to form shoals of unlimited size. This allows the exploration of different shoaling behaviours which can potentially affect the feeding and fishing functions.

4. Otherfood. It is not anticipated that the full range of potential food items will be modelled, indeed for catholic predators such as cod it is impractical to model all individuals of all potential prey types. The model therefore utilises “otherfood” to generate “fish” objects to represent the biomass of all other potential prey. Fish of type “otherfood” are different in that they don’t die (so the availability of otherfood objects is constant) and they don’t migrate.

5. Patch. Although individual fish know their location in physical space, the environmental conditions for that space are held within this object. Such conditions include depth, temperature etc. Patches are also the spatial scale at which fishing and feeding occurs within each time-step of the model.

6. Vessel. Fishing vessels are specified with size, operational speed, type and size of fishing gear. Fishing gears are specified with a mesh size so that the probability of capturing individual fish can be determined.

7. Logbook. For each time-period within the model a vessel will fill in a logbook object which records where it has been fishing and the numbers and tonnage of fish caught by species.

Within any given time-step the model is required to perform the following actions, feeding, growth, migration and fishing, which is does in a fixed order. The temporal and spatial scales of the model have the potential to be fully flexible and can therefore operate on an annual scale and a single area, right down to daily time steps and thousands of patches. However, care needs to be taken when defining the spatial and temporal scales as they need to match the ‘foraging range’ of fishing vessels and predators alike such that a “patch” should be roughly equivalent to the area a typical fishing vessel would operate in for a given time-step.

The operational functions are as listed below.

1. Feeding. All individuals (including all the “otherfood” type fish) within the patch are temporarily merged into one supershoal. From this list, all the fish which are currently hungry are identified and each of these hungry fish sequentially works through the supershoal to identify which item to eat. As each predator encounters a new prey object the basic decision process is as follows.
    a. Am I still hungry? If not then move to the next hungry predator.
    b. Do I eat objects of this species type?
    c. Is it the right size for me?

The size function has been parameterised from field data and assumes that a predator of given size will have a range of preferred prey sizes. The model has been constructed such that predators can carry a number of prey-size functions so that where data are sufficient to parameterise a prey-selection curve for a particular predator-prey pairing this can be used and for other prey types a more generic prey selection curve can be used.
Feeding ceases once the predator is full. This is controlled by the maximum energy requirements of the fish.

2. Growth. Growth has been implemented as a bioenergetic function. Whenever a predator eats a fish, the energy content of that fish is passed to the predator. When it comes to growing, the energy requirements for basic metabolism and locomotion are deducted from the ingested energy. Any energy left over is then put into somatic growth. For the purposes of this model we have ignored the portion of energy which goes into reproduction, but this could be added at a later date. Growth occurs in both weight and length of individuals, however if there is insufficient energy to supply the basic metabolic requirements of the predator then its weight will decrease whilst the length will remain constant.

3. Migration. Migration has been implemented as a function of Lewy flight. This describes the probability of an individual moving number of body lengths in a given time period. Three parameters control the shape of this function, but essentially for most individuals and short time steps there is a high probability that it will only move a short distance and a low probability that it will move a long distance. Different migration patterns have been constructed for different species, life stages and seasonalities. Adult cod, for instance, will undergo directional migration in order to spawn, reverse the migration path to return to the feeding grounds and then undergo short-scale random migrations as they move about to feed. Adult sandeels however will only undergo very short scale random migrations and always remain within a given patch.

4. Fishing. Fishing works on the swept area basis. Within a time-step, a fishing vessel will operate for a number of hours (determined from satellite data), towing gear with a known door-spread. From this we determine the swept area within the time step and divide this by the total surface area of the patch the vessel is currently operating in. This gives a probability of covering any particular point within the patch (point probability). We assume that shoals have a random uniform distribution within the patch and therefore the probability of encounter between vessel and shoal is the same as the point probability. The shoal is then filtered using the size selectivity parameters of the specified gear.

We intend to explore the implications of a variety of potential management options:

- Total closure of the sandeel fishery (the base-case).
- TAC. (the traditional management system for sandeels)
- Local TAC.
- Global effort limitation.
- Local effort limitation.
- Permanent MPAs closed to all fishing.
- Permanent MPAs closed to sandeel fishing.
- Rotational MPAs closed to sandeel fishing.
- Closed seasons. (to prevent fishing on the recruiting ages).

5.6 Assessment of the effects of the Firth of Forth sandeel fishery closure on breeding seabirds

Morten Frederiksen and Sarah Wanless, NERC

The industrial fishery for sandeels started in the North Sea in the 1950s and gradually developed into the largest single-species fishery in the region, with landings exceeding 1 million t in some years (Furness 1999). The complex of sand banks off the Firth of Forth in SE Scotland (Wee Bankie, Marr Bank etc.) was not exploited by the sandeel fishery until 1990, when Danish vessels started to fish here. Landings then quickly grew to more than 100,000 t in 1993, a level that was considered to have negative effects on local sandeel stock size as well as breeding productivity of black-legged kittiwakes (Rindorf et al. 2000). In the late 1990s, concern arose after several years of very poor kittiwake breeding productivity on the Isle of May and other colonies in the Firth of Forth area. A zone along the east coast of Scotland and Northern England, including the Wee Bankie, was therefore closed to the sandeel fishery from 2000 (Camphuysen 2005). The closure was initially for a three-year period, but was later extended and will be re-evaluated in 2006. A limited-scale survey fishery by commercial fishing vessels has been maintained throughout the closure period (Camphuysen 2005). NERC analysed the available data from seabird colonies along the UK North Sea coast, inside and outside the closure zone, to assess whether the fishery and/or the closure has had an effect on breeding seabirds, including, but not restricted to, black-legged kittiwakes.

The assessment concentrated on two questions: i) Did the fishery affect breeding seabirds on the Isle of May, NERCs primary field site? ii) Did the fishery affect breeding kittiwakes throughout the closure zone? Data were available for the period 1986-2005, i.e. before, during and after the period when the fishery was active.

1. Effects on Isle of May seabirds. This analysis tested for fishery effects, controlling for three environmental covariates previously shown to be important: sea surface temperature in the previous year, sandeel size, and the biomass of sandeel larvae in the previous year. A negative effect on breeding productivity was found for the surface-feeding black-legged kittiwake, but not for four other species, all of which feed by diving (Fig. 5.6.1). No effect was found on the condition (fledging mass) of chicks for the two diving species where this is monitored (common guillemot and Atlantic puffin).

2. Effects on black-legged kittiwakes in the closure zone. This analysis compared breeding productivity of kittiwakes at seven colonies in the closure zone and five colonies further south along the UK North Sea coast, using a replicated BACI (before-after control-impact) design. A statistically significant interaction between zone and period was found, indicating that kittiwake breeding productivity was lower in the closure zone when the fishery was active, whereas there was no difference between the two periods in the control zone (Fig. 5.6.2).

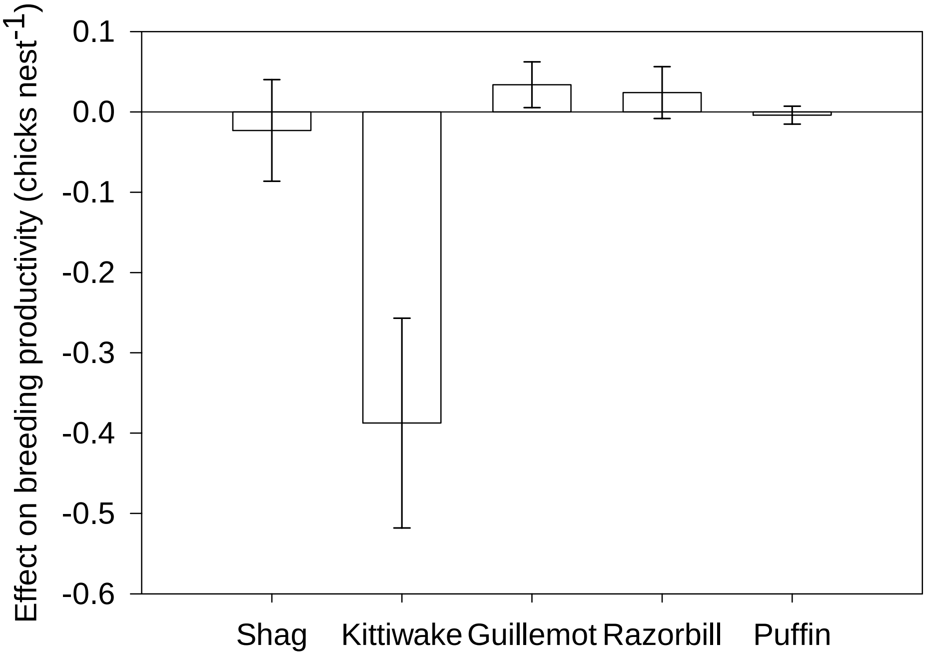


Figure 5.6.1. Fishery effects on breeding productivity of Isle of May seabirds. Error bars indicate ± 1 standard error.

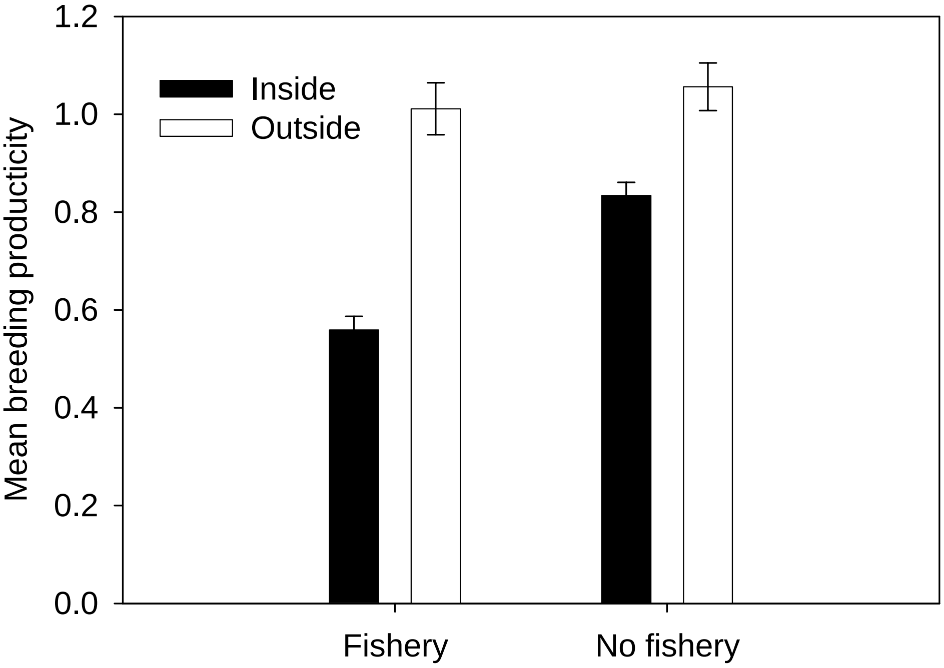


Figure 5.6.2. Mean breeding productivity of black-legged kittiwakes inside and outside the closure zone, in fishery and non-fishery years. Error bars indicate ± 1 standard error.

The results consistently indicate that the sandeel fishery in the Wee Bankie area reduced breeding productivity of black-legged kittiwakes (by 0.28-0.39 chicks/nest), whereas no effects were found for other species. The most obvious interpretation is that kittiwakes are more vulnerable than other species to a reduction in sandeel abundance, because they as surface feeders only can access a fraction of the fish that actually occur in the water column. However, it is also possible that the vertical distribution of sandeels was affected by the fishery, making a smaller fraction available to kittiwakes.

These analyses have been described in a detailed internal PROTECT report attached in Annex 7, and are currently being written up for publication in a high-ranking scientific journal (Ecological Applications).

References:

1. Camphuysen, C. J., editor. 2005. Understanding marine foodweb processes: an ecosystem approach to sustainable sandeel fisheries in the North Sea. IMPRESS final report. Royal
2. Netherlands Institute for Sea Research, Texel, The Netherlands. Rindorf, A., S. Wanless, and M. P. Harris. 2000. Effects of sandeel availability on the reproductive output of seabirds. Marine Ecology Progress Series **202**:241-252.

For full details on this study, please refer to **Annex 7.**

5.7 Environmental valuation study, North Sea, Sandeels

Prem Wattage, CEMARE

For a description of the environmental valuation study of North Sea sandeels, please refer to **Section 6.3**.

5.8 Model for the effect of MPAs on migration and dispersal of commercial fish stocks

Charlotte Deerenberg, Willem Dekker, Frank Storbeck, Bert Brinkman and Niels Daan, IMARES

5.8.1 Objectives

To evaluate the effects of MPAs on a migratory fish population and the fishery on that species and to provide reference points against which system dynamics can be gauged. To this aim a model has been developed specifically based on the dispersal and migration characteristics of individuals of the modelled species. The two variations of the model are tailored to simulate behaviour of Plaice and Cod and their fisheries on the North Sea.

The following questions related to management of protected areas will be addressed via the model:

- What is the effect of establishing marine protected areas on the total population and the commercial catch of the target species?
- What surface area is required to achieve a reasonable protection?
- Where should the protected areas be located?

5.8.2 Role in PROTECT

- The IMARES model constitutes a methodology to assess the potential of different MPA regimes (first and second main aim of PROTECT) to protect and manage commercial fish stock and their fisheries.
- The IMARES model is a typical example of a stock specific spatial model (second objective of WP5).
- The biological characteristics of the model – other than growth and survival – specifically reflect the dispersion and migration characteristics of individual fish. The physical characteristics of the model are tailored to reflect the North Sea and several of its physical conditions.

5.8.3 Rationale and work strategy

Protected areas are a tool for environmental protection, directed at specific species or habitats. Area closure may also act as a positive tool for fisheries management by exhibiting some degree of protection to commercial fish stocks that suffer from overexploitation. Although a whole range of potential benefits of protected areas to commercial fish stocks has been identified, e.g., stock recovery and more predictable and higher catches (PROTECT 2006), little empirical evidence exists to demonstrate such effects.

Closing an area to fishery undoubtedly has a local effect on fish and other organisms. This effect may quickly reduce to zero when the extra fish produced is harvested the moment they leave the protected area. Likewise, the commercial fishing industry suffers from marine protected areas, unless fish migrate out of the protected area. The main factors that determine the effect of protected areas can be grouped into three issues: characteristics of fish (spatial and temporal dynamics), of protected areas (size, location, protection level) and of fishery (spatial and temporal patterns).

To capture the tendency of a fish species to move around and the resulting distribution area, we constructed an individual-based model (IBM) in which key processes driving movement of an individual fish are modelled. This allows us to produce patterns of distribution of fish in both space and time. Superimposing distribution patterns of fishing effort on those of fish determines local fishing mortality F. These two – spatially distributed – factors (numbers of fish occurring somewhere and fishery mortality F) suffice to infer temporal variation in population size and population composition of the modelled species. The emergent properties of the population are determined by the sum of all individuals, not by imposing trends on abundance and distribution on the collection of all individuals. We add a degree of realism to the model by using as much as possible real behaviour and real effort and effort distribution. With such an individual-based model we are able to investigate the effects of size and location of a protected area on the modelled fish population and the modelled catch.

5.8.4 Results

5.8.4.1 IBM model

Key aspects and processes of the model comprise explicitly the spatial processes of migration and homing relative to the location of spawning, nursery areas and preferred habitats. To this end we need a good insight in the ecology of the selected species, based on field research. We initially selected Plaice *Pleuronectes platessa* as our model species, because of the extensive research carried out on this species, and because Plaice is one of the most important target species of the Dutch beam trawl fishery. In a later phase, we will adapt the model to reflect Cod *Gadus morhua* behaviour and fishery in the North Sea, because this is one of the species subject to a recovery plan for which the EU contemplates MPAs as a management strategy.

Conceptually, the model consists of two layers: a physical and a biological one. The physical layer represents the North Sea (including the Wadden Sea) as a rectangular grid. Each grid cell has relevant physical characteristics, such as mean depth, position, size, temperature and fishing effort. The latter is derived from the average annual effort distribution of a relevant fleet and the selected management scenario (see below). The effort distribution was derived from that of a selection of Dutch beam trawls, divided into Eurocutters and trawls of ~2000hP, raised to represent the effort of the total beam trawl fleet, whose main target species are Plaice and Sole *Solea vulgaris*.

The biological layer contains fish, reacting to their physical environment. These fish spawn in designated areas (the spawning areas), that is: new animals are introduced at the start of each year in these grid cells, whereas the adults have to return to these areas during the spawning season. In their first year of life, young fish migrate towards nursery grounds. At the end of the first year, and in subsequent years, the fish migrate through the sea, while they grow in size, get older, undergo natural mortality, and are being fished. The migratory behaviour for the first selected species, Plaice, depends on depth preferences (that is: on depth of the grid cell where the fish resides, and on depth of the adjacent cells), the age of the fish, and a generalized migration pattern with a stochastic variation on swimming direction. Parameter values are derived from literature and the resulting distribution pattern is compared to the actual distribution. The implementation is based on discrete time steps of one week.

The model is minimalistic, in the sense that we model the absolute minimum number of processes for an individual fish that may show potential effects of area protection on survival, distribution and fishing yield. In particular, dependence on food sources and mutual interactions between individuals have not been included, which contrasts with conventional Individual Based Models (Railsback and Grimm 2006). Since the objective is rather focused (analysis of effects of protected areas), we preferred a traceable and tractable minimalist model.

The model was implemented in SWARM 2.2 (Anon. 2004), an open-source simulation environment for individual based modelling (IBM).

5.8.4.2 Management scenarios and model runs

The management scenarios examined cover a control, and realistic and experimental scenarios, ranging from all of the North Sea open to fisheries to an area that results in substantial positive effects on the target species and the fishery. To gauge the degree of realism of the model, the simulated natural distribution of fish under the current fishing regime was compared to information on the spatial distribution of catches. Subsequently, a range of scenarios (8) for protected areas was imposed, in which fishing was reduced or set to zero in protected areas and the corresponding fishing effort was redistributed over the remaining areas.

Each of the model scenario's was initiated at t=0 with a small number of fish reproducing at the spawning place. Following this initiation, the model quickly converges to a stable state after some 600 time steps (weeks), characterising each scenario. Results will be presented for the full series (6000 weeks, i.e. >100 years) starting at t=0, with the initial 600 weeks shaded.

A comparison of scenario results is presented below, in a Box-Whisker plot of the long-term averages (and variation) in abundance, fishing yield, and individual weight (Figure 5.8.1). Separate 'time series' are presented in Figure 5.8.2, describing the evolution in abundance, fishing yield, and individual weight for each of the scenarios, as well as a map of the spatial distribution.

5.8.4.3 Model results

We present a comparison among scenarios for Plaice in the North Sea. The first two scenarios (the whole North Sea open to fisheries, respectively the Plaice box closed for beam trawls > 300 hp) effectively set the standard for the current conditions. The results obtained characterise the model assumptions, with parameters tuned to realistic values. The third scenario (Plaice Box only open to Eurocutters and wind farms closed to all fisheries) adds another 100 nm^2 of closed area. Results closely resemble those of the first two scenario's (Figure 5.8.1), that is: closing the wind farm areas has a negligible effect on the simulation runs. The fourth scenario encompasses all proposed protected areas, with a total surface area of 5,000 nm^2. As for scenario 3, the effects on fish stock and fishery appear to be minimal, although a slight increase (3%) in catch weight is found. Protecting an even smaller area, only Doggerbank and Cleaverbank (2,600 nm^2), thus has a negligible effect too, and results in a smaller (2.5%) increase in catch weight. Replacing the proposed protected areas by a single continuous area of the same size (scenario 6) does not alter the results, indicating that the small size, and not the fragmentation of the protected areas determines the overall lack of response. In contrast to the previous two, this scenario does not show an increase in catch weight; that is: the positive effects on the stock are counterbalanced by less efficiency in the fishing, which is due to the large protected area. Closing a substantial part of the North Sea (25%, scenario 7) does indeed have a considerable effect. Stock abundance increases by 50%, mean individual weight increases by 100%, and stock biomass increases by 200%. Since we situated this 25% closed area in the southern North Sea, where the plaice fisheries are concentrated, the catch in numbers is reduced by 60%. The increased survival, however, results in a higher average weight, and catch in weight is only reduced by 40%. The final scenario, closing the whole North Sea for all fishing, obviously results in a recovery of the stock, abundance nearly doubling, and biomass increasing nearly five-fold.

Overall, all scenario's show nearly identical results, except for the last two, based on protected areas of much larger size than the ones currently considered.

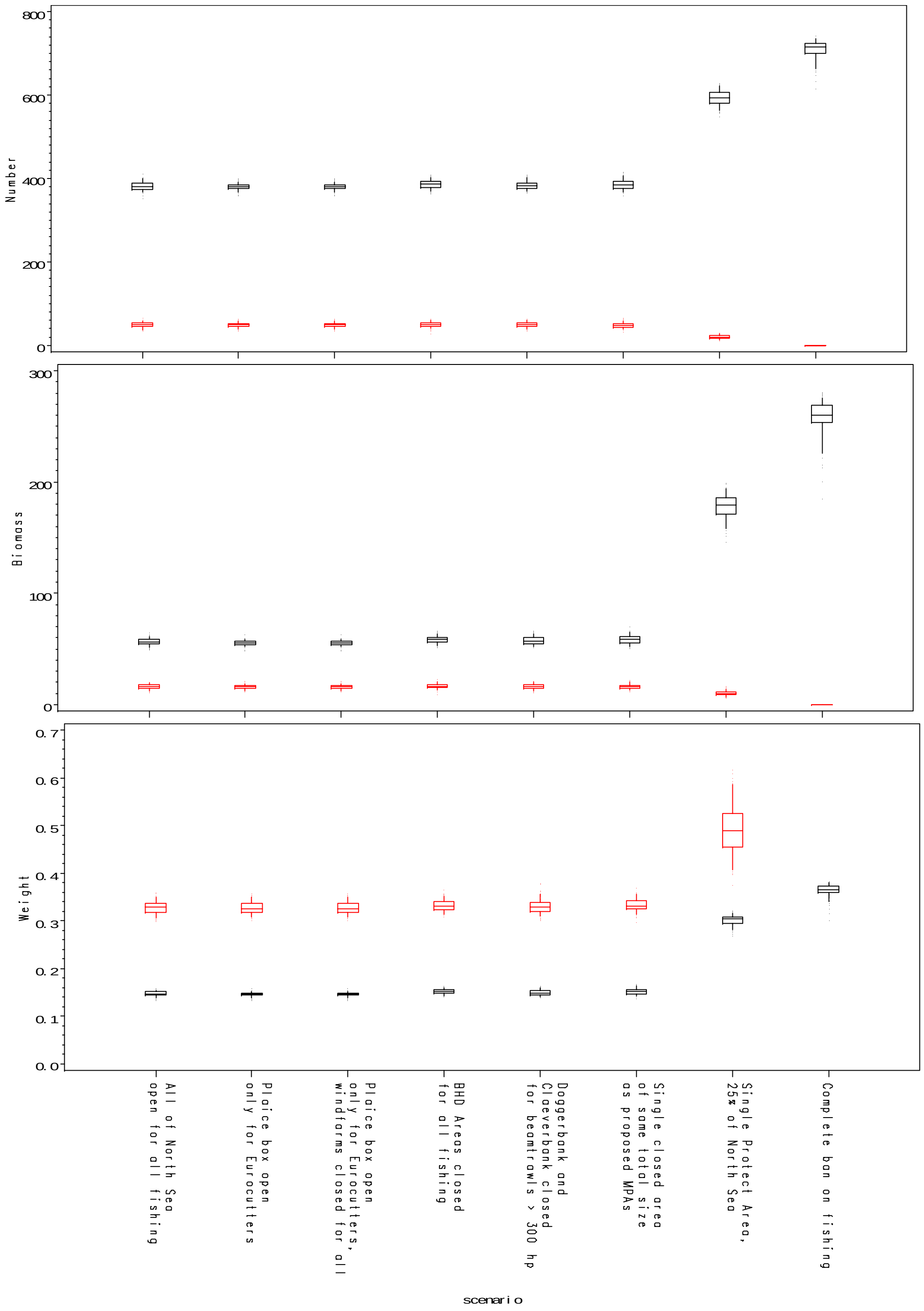


Figure 5.8.1: Comparison of model output for stock (black solid boxes) and catch (red dotted boxes) in terms of numbers (top panel), weights (middle panel) and weight of individual fish (bottom panel) for the 8 scenario's described in the results section. Note that in scenario 8 all fishing is prohibited (catch is zero), and no average weight in the catch can be calculated.

Figure 5.8.2 - Detailed model results

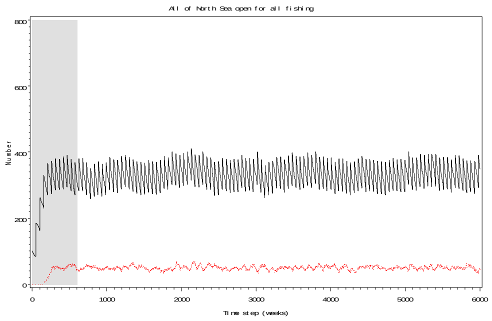


Figure i. 'Time series' of the numbers in the population and the catch, for **scenario 1** (All of North Sea open for all fishing), as derived from the simulation model. Solid black line indicates the population number; dotted red line the number in the catch.

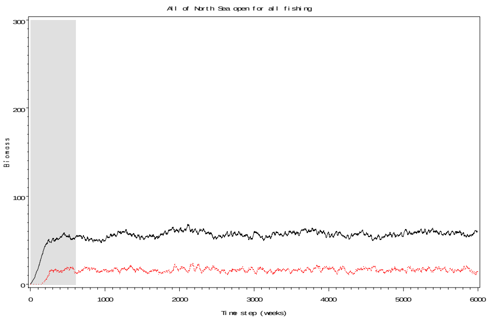


Figure ii. 'Time series' of the biomass in the population and the catch, for **scenario 1** (All of North Sea open for all fishing), as derived from the simulation model. Solid black line indicates the population biomass; dotted red line the biomass of the catch.

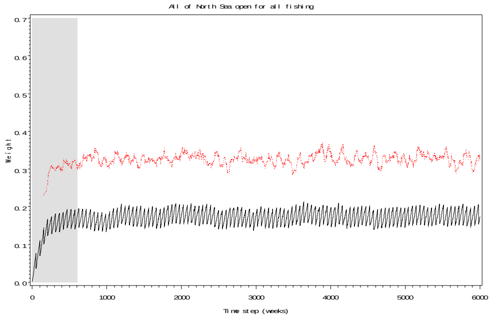


Figure iii. 'Time series' of the average individual weight in the population and the catch and the catch, for **scenario 1** (All of North Sea open for all fishing), as derived from the simulation model. Solid black line indicates the average weight in the sea; dotted red line the average weight in the catch.

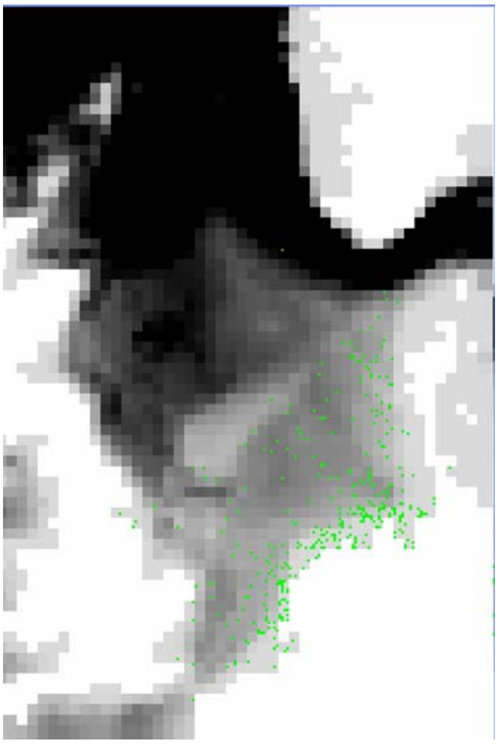


Figure iv. Spatial distribution of the population, for **scenario 1** (All of North Sea open for all fishing), as derived from the simulation model (snapshot of the distribution at t=6000).

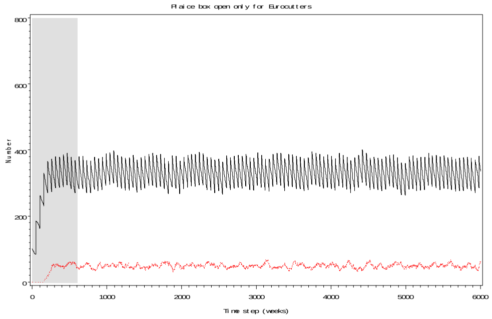


Figure v. 'Time series' of the <u>numbers</u> in the population and the catch, for **<u>scenario 2</u>** (Plaice box open only for Eurocutters), as derived from the simulation model. Solid black line indicates the population number; dotted red line the number in the catch.

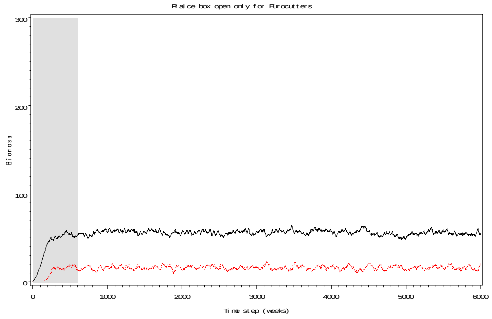


Figure vi. 'Time series' of the <u>biomass</u> in the population and the catch, for **<u>scenario 2</u>** (Plaice box open only for Eurocutters), as derived from the simulation model. Solid black line indicates the population biomass; dotted red line the biomass of the catch.

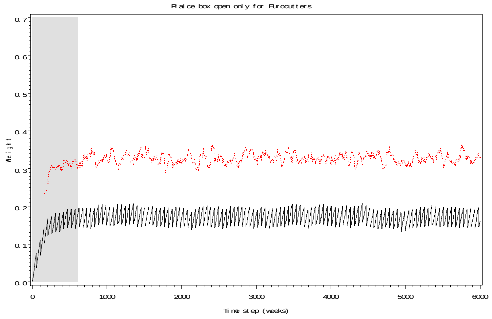


Figure vii. 'Time series' of the average <u>individual weight</u> in the population and the catch, for **<u>scenario 2</u>** (Plaice box open only for Eurocutters), as derived from the simulation model. Solid black line indicates the average weight in the sea; dotted red line the average weight in the catch.

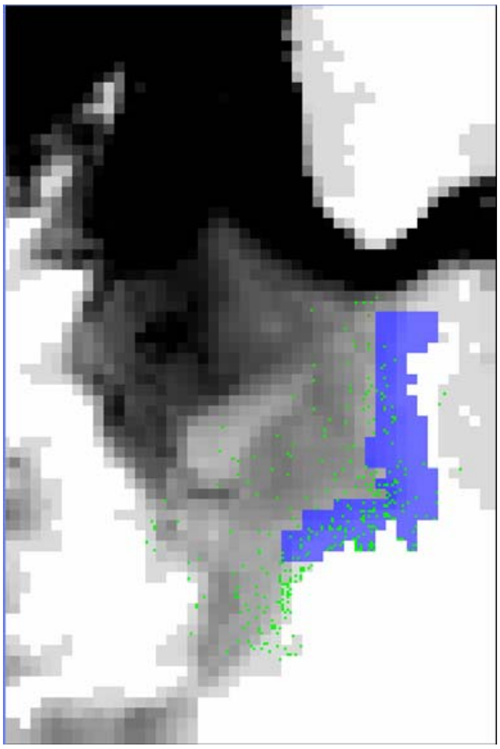


Figure viii <u>Spatial distribution</u> of the population, for **<u>scenario 2</u>** (Plaice box open only for Eurocutters), as derived from the simulation model (snapshot of the distribution at t=6000).

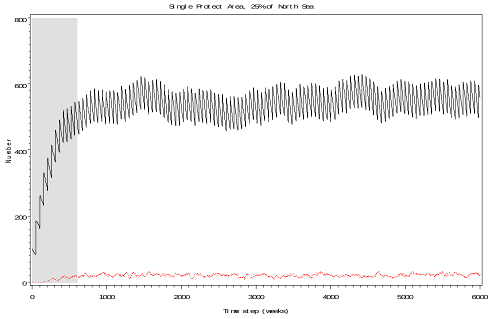


Figure ix. 'Time series' of the numbers in the population and the catch, for **scenario 7** (Single Protect Area, 25% of North Sea), as derived from the simulation model. Solid black line indicates the population number; dotted red line the number in the catch.

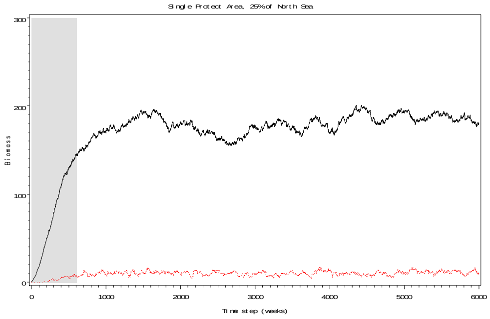


Figure x. 'Time series' of the biomass in the population and the catch, for **scenario 7** (Single Protect Area, 25% of North Sea), as derived from the simulation model. Solid black line indicates the population biomass; dotted red line the biomass of the catch.

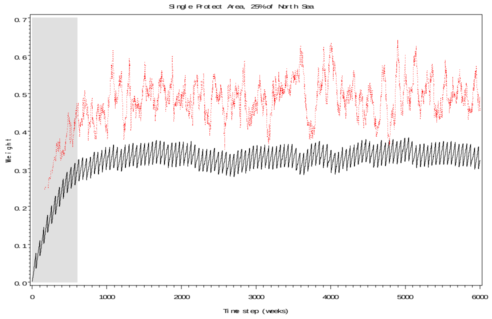


Figure xi. 'Time series' of the average individual weight in the population and the catch, for **scenario 7** (Single Protect Area, 25% of North Sea), as derived from the simulation model. Solid black line indicates the average weight in the sea; dotted red line the average weight in the catch.

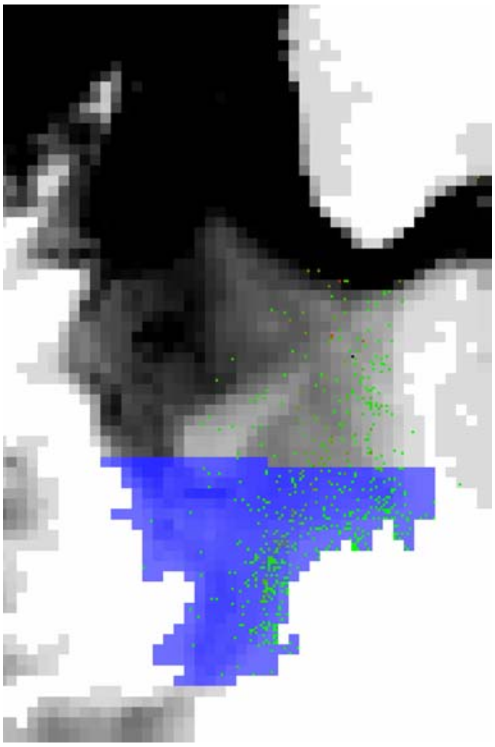


Figure xii. Spatial distribution of the population, for **scenario 7** (Single Protect Area, 25% of North Sea), as derived from the simulation model (snapshot of the distribution at t=6000).

6. Progress in work: Coldwater coral case study

6.1. Bioeconomic models for cold water coral – the production function approach, the HABFISH and RUM models.

Claire W. Armstrong (NCFS) , Naomi Foley (NUIG) and Viktoria Kahui (NCFS)

6.1.1. Introduction

Coral reefs are generally associated with shallow tropical seas, however, recent cold-ocean exploration using advanced acoustics and submersibles have revealed unexpectedly widespread and diverse coral ecosystems in deep water on continental shelves, slopes, seamounts, and ridge systems around the world (Roberts et al, 2006).

Scientific research has observed certain species of commercial importance aggregating around cold water coral (CWC) reefs (Fosså et al, 2002; Husbø et al, 2002). This may imply that CWC is an important habitat for supporting certain species. If cold water coral can be linked empirically to a commercial species then there is evidence that coral depletion can have a harmful effect on the fishing industry.

Although scientific research is progressing on cold water corals, no economic studies have been applied to the resource, as far as we are aware. Cold water corals potentially provide a number of ecological goods and services which can be grouped in terms of direct, indirect and non-use values. In the three models presented here, we study the indirect use values, specifically the role of cold water coral as a habitat for a commercial species.

The three approaches applied study different aspects of the fisheries-habitat connection to cold water coral. The production function approach attempts to determine a statistical connection between cold water coral, and in the case studied here, redfish on the Norwegian coast. The bioeconomic HABFISH model, is an analytical study of how fishing that affects a non-renewable resource such as cold water coral, which again has some biological input into a commercial fish species, affects the economic behaviour in that fishery. The RUM model can highlight the productivity link between cold water coral and for instance orange roughy abundance statistically, by analysing the harvesting decisions of vessels for orange roughy.

6.1.2. The production function approach

Estimating linkages between Lophelia and redfish on the Norwegian coast

The waters surrounding Norway hold some of the best examples of cold water corals, particularly Lophelia pertusa. A number of commercial fish species have been identified in coral areas among which are redfish (*Sebastes marinus*), ling and (*Molva molva*), tusk (*Brosme brosme*) and saithe (*Pollachius virens*) (Husebø et al. 2002).

Redfish became an important commercial species in Norway in the mid-eighties. The fishery for redfish has been an open access fishery until recently. The vessels mainly harvesting these species are trawlers – factory, fresh fish and small trawlers. Though the harvests have been somewhat erratic, catch levels have dropped since the end of the 90’s.

Fosså et al (2002) estimated that Norwegian coral grounds had been destroyed by 30% - 50% from anthropogenic activities. In 1999 the Norwegian Ministry of Fisheries issued regulations for the protection of known coral grounds.

The production function approach provides a useful method for valuing potential linkages between habitat and fisheries. The basic assumption underlying this approach is that, if a coral ground serves as a habitat for a commercial fishery, then this ecological service serves as an additional environmental input to the fishery (Barbier 2000). Applications of this valuation methodology to habitat-fishery linkages can be grouped into 'static' and 'dynamic' approaches (op. cit.). There have been a number of studies recently that employ either the static or dynamic production function approach to estimate the economic benefits of wetland-fishery linkages. The work in PROTECT applies the dynamic approach to the case of the Norwegian redfish fishery.

The model applied is adapted from the work of Barbier and Strand (1998) in which they develop the model to value mangrove-fishery linkages. The model used modifies the standard bio-economic fishery model to account for the effect of a change in CWC habitat on carrying capacity and therefore on production.

The Norwegian redfish fishery was open access until recently. This suggests that fishing effort in the next period will adjust to real profits made in the current period. The analysis of fishery-lophelia linkages is conducted by examining the effects of a change in lophelia area on the long-run open access equilibrium of the Norwegian redfish fishery.

The data consists of landings and price data from the Norwegian Raw Fish Organisation as well as cost and effort data from the Norwegian fisheries directorate's annual profitability investigations for fishing vessels (in Norwegian) for the years 1986 – 2004 Data on the area of Lophelia is not available. It is assumed an initial total coverage of Lophelia. Fosså et al (2002) first estimate of fishery impacts on cold water coral reefs indicate that 30 – 50% of reefs are damaged or impacted. It is these estimates that are used for the approximation of coral decline from 1984 – 2004. It is assumed that damage stops in 1998 with the introduction of the Norwegian legislation prohibiting damage to known coral grounds.

6.1.3. The HABFISH model

Given that there is some connection between habitats, such as deep water coral, and commercial fish species or their prey, as shown in the above mentioned production function approach, bioeconomic models can then tell us something about how these resources optimally should be managed. The HABFISH bioeconomic model is one such model, and is based on Swallow's (1990) seminal work which synthesized renewable and non-renewable resource utilisation. In this work Swallow designs a model that allows for a non-renewable resource that interacts with a renewable resource via the latter's growth function. In our context, the interaction would be the effect that deep water coral has upon the growth potential of commercial fish species, or their prey. The analysis shows that there are circumstances where even under optimal harvesting management conditions, the non-renewable resource utilisation leads harvesting of the renewable resource in excess of the sustainable rate. Furthermore it is shown that if a profitable renewable resource sector survives, the utilisation of the non-renewable resource ceases before all profitable opportunities are exhausted. These two results are in opposition to standard results when there are no interactions between the two types of resources.

This model is well suited to the cold water coral case, as this coral is very slow growing, and can for all modelling purposes be considered a non-renewable resource. In our model we furthermore expand upon the Swallow model by allowing for the non-renewable resource to affect the costs of harvesting, for instance by aggregating stocks for harvest. This is shown to vastly change the analysis itself, though the final results may not be significantly changed.

A central question that must also be addressed is how other management options, such as open access, which has been prevalent in many fisheries affects the results in the Swallow model. This would also give an indication as to how the cold water coral resources have fared in the years where they have been linked to commercial harvesting.

6.1.4. The RUM model

In a previous bioeconomic analysis (Kahui, 2006) of a network of no-take areas around Stewart Island in New Zealand for the shellfish species paua (abalone), a nested logit model is applied to spatially recorded catch and effort data by the Ministry of Fisheries between 1998 and 2003 to capture the two level decision-making process of divers (Smith and Wilen 2003). On any given day, divers decide whether to go diving at all, and if so, which of the 16 statistical areas around Stewart Island to visit. Weather conditions, spatially varying levels of catch per unit of effort and distance are used as explanatory variables to select areas for closure according to the ‘least economic impact’ in terms of loss of diving trips.

An age-structured biological model is developed with parameters specifically applied to paua stocks around Stewart Island (Annala et al. 2005). Virgin paua biomass as of 1974 is estimated on the basis of growth, survival, post-larval recruitment and egg production in the absence of fishing. Historic catch rates are then applied to find overall and area-specific levels of exploitation rates, spawning biomass, egg production, legal biomass and numbers of paua. In a final step, the economic model is linked to the biological model to simulate the imposition of no-take areas when taking account of the initial disproportional shift of harvest to fished areas in the first year, and the increase in overall pressure on legal biomass in the years thereafter.

The findings have implications for the management of the paua fishery at Stewart Island. For a heterogeneously abundant species, such as paua, spatial management in addition to quota limits could be vital in ensuring the long-term sustainability of the fishery given the inherent variability of the marine environment.

A similar model can be applied to the decision making of vessels trawling for orange roughy in Ireland/Norway and/or New Zealand. On any given trip vessels decide whether to trawl in a deep water coral area or not, and then choose amongst a number of patches for their fishing activities. Should the estimation results turn out significant, we can show a direct productivity link between deep water coral and orange roughy abundance deducted from the harvesting decisions of vessels. This could have significant implications for other fish species and provide a valuable indication of the indirect use value of cold water corals. The analysis can also show the response of vessels should spatial closure be implemented.

6.1.5. Discussion

The presented analysis ties together to give information regarding the use of marine protected areas and the management of cold water coral. The production function approach illustrates how the connection between the habitat and the fish species in question can be ascertained. The HABFISH model then illustrates how optimal economic management would determine the closure of deep water coral areas. The RUM model can provide decision makers with some idea of the productivity link between cold water corals and orange roughy, and the redistribution of vessel trips should marine protected areas be implemented.

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6.2 Study on ecological–economic modelling of marine reserves in fisheries

Claire W. Armstrong, NCFS

The economic analysis is shown to be far more pessimistic with regards to the potential of marine reserves as a fisheries management tool, than what one finds in the purely ecological analysis, the reason being the latter's neglect of issues such as discounting and economic incentive behaviour. However, economic analysis, despite some of it being relatively advanced with regards to spatiality, is still simplistic with regards to for instance ecosystem and habitat content. A simple expansion of the existing bioeconomic models with regards to positive habitat effects of area closures is presented and analysed, showing room for improved results from marine reserve implementation as compared to the existing analysis.

The study is presented in it entirety in **Annex 8**.

6.3 Environment Valuation for Deep-sea coral and Sandeel case studies

Premachandra Wattage, CEMARE

6.3.1. Introduction

During the reporting period, environmental valuation techniques have been designed in two case studies, i.e. deep-sea corals and North-sea case studies. The purpose of these techniques is to assign values or preferences for resources and environmental attributes that do not command a market price. The absence of market prices means that some other way of imputing monetary values has to be found, which in practice generally involves establishing people's preferences (reflected in their willingness to pay, WTP) for specified benefits derived from marine environmental assets. An extensive literature on environmental valuation now exists that best illustrates the valuation problem in the context of MPAs.

6.3.2. Main model feature

Contingent valuation method (CVM) is widely used for eliciting respondent's preferences for unpriced benefits associated with marine environmental quality, especially the non-use values. The results provide a monetised measure of environment value. However, the specific nature of the resource to be valued in two case studies of the project makes it difficult or rather impossible to depend on one type of model. Asking the general public to value a resource that has never seen or is difficult to imagine may lead to flawed estimates. The scenario of the sandeel case study is simple for the general public to understand compared to the deep-water coral study. Due to this reason, CVM is only suitable for the analysis of data that will be collected in the sandeel case study. The rationale given for needing familiarity is the assertion that respondents cannot have well-defined preferences in an economic sense for good with which they have no direct experience (Carson et al. 2001). However, the argument for using CVM is that non-use value is based on the benefits from a marine resource from knowledge of its continued existence or availability to future generations

Main model feature of the CVM study is that the estimation of willingness to pay function. To see this more clearly, assume that the willingness to pay function takes the form:

$$WTP = X\beta + \mu$$

where WTP is an $nx1$ vector, X is an nxk matrix containing a constant and possibly other explanatory variables such as income, preference for environment conservation and taste attributes, β is a $kx1$ vector of unknown parameters, and μ is an $nx1$ vector of random terms distributed $N(0,\sigma^2)$.

Choice Experiments (CE) is a form of analysis used to represent individual judgements of multi-attribute stimuli (Wattage et. al. 2005). The aim of this approach is to estimate the structure of an individual's preferences by establishing the relative importance of attributes. To achieve this, sets of alternatives that are prespecified in terms of levels of attributes are incorporated into a questionnaire. The total utility that an individual derives from that alternative is thereby determined by the utility to the individual of each of the attributes. The aim of the CE model is therefore to estimate (a) the relative importance of the individual attributes; (b) the trade-offs or marginal rates of substitution that individuals are willing to make between these attributes; and (c) the total satisfaction or utility scores for different combinations of attributes. The characteristics of the alternatives that the individual must choose from are multi-attribute in nature.

The effect (preference) of introducing a MPA to protect deep-sea corals is studied using discrete choice models. Respondent's choice among attributes listed in the questionnaire **(Annex 9)**, given choice set C_a of available levels, will correspond to the utility. For each level *i*, the utility U_i is the sum of a systematic component V_i and a random component of e_i. The probability of choosing level *i* from choice set C_a is therefore:

$$P(i \mid C_a) = P(U_i > \max(U_j)) = P(V_i + e_i > \max(V_j + e_j)) \forall (j \neq i) \in C_a$$

From this, we estimate the utility:

$$V = a_i + \sum_k (b_{ik} \times x_{ik}) + \sum_{j \neq i} z_j (d_{ij} + \sum_l (g_{ijl} \times x_{jl}))$$

Where:

ai = intercept for level i
bik = effect of attribute k for level i, where k = 1,...,K.
xik =amount of attribute k for level i.
Dij = availability cross-effect of level j on level i.
zj = availability code, 1 if jCa and 0 otherwise.
gijl = cross-effect of attribute l for level j on level I, where l = 1,...L.
xjl = amount of attribute l for level j.

6.3.3. Survey questionnaires

Questionnaire design and field data collection has been planned during the reporting reported. Two questionnaires for CVM and CE are presented in the **Annex 9**. In designing the CE questionnaire, orthogonality has been assumed. Orthogonality usually implies that the coefficients will have minimum variance. The *factors* of experimental design are variables that have two or more fixed values or *levels.* Experiments are performed to study the effects of the factor *levels* on the *dependent* variable. In a *discrete choice* study, the factors are the *attributes* of the hypothetical products or services. The *response* from people or subjects is the *preference* or *choice*.

Several workshops and discussions were held before making the final decision on the questionnaire format. Questionnaires will be tested in the field (pilot) before their despatching to the general public by surface mail.

6.3.4. Experimental design of the choice study

Design of experiments has been used to identify the preference choice bundle presented in the questionnaire. The *factors* of experimental design are variables that have two or more fixed values or *levels.* Experiments were performed to study the effects of the factor *levels* on the *dependent* variable. In a *discrete choice* study, the factors are the *attributes* of the hypothetical products or services. The *response* from people or subjects is the *preference* or *choice*.

The aim of the approach is to estimate the structure of an individual's preferences by establishing the relative importance of attributes. To achieve this, a set of alternatives that are prespecified in terms of levels of attributes are incorporated into a questionnaire. The total utility that an individual derives from that alternative is thereby determined by the utility to the individual of each of the attributes. The aims of the CE based technique are therefore to estimate (a) the relative importance of the individual attributes; (b) the trade-offs or marginal rates of substitution that individuals are willing to make between these attributes; and (c) the total satisfaction or utility scores for different combinations of attributes.

Choice-experiments consider the number of alternatives while either holding the attribute levels associated with each alternative constant, or by varying them, thereby producing choice sets. It also uses similar principles of experimental design, with respondents' expressing opinion by making a choice between the different combinations presented. Fixed choice set designs are particularly widely used

In the implementation of a CE study, there are several steps and considerations that have to be completed, and therefore form the basic framework of evaluation (Green and Srinivasan, 1978 and 1990). Firstly, a set of attributes (p=1,…,t) are chosen and the alternatives defined. This involves 3 key elements: understanding the decision problem and environment, identifying determinant attributes, and establishing attribute positioning measures.

The methodology disaggregates the management process into key attributes with different potential levels. Attribute positioning measures, such as the level of accomplishment of attributes, are developed that satisfy the research objectives and are meaningful to the individuals targeted for survey.

Two types of methods are generally implemented for data collection: two-factor-at-a-time and full-profile. The two-factor method is simple to apply, reduces information overload and is more appropriate for a postal survey, but there are several limitations to the approach (see Green and Srinivasan 1978). However, CE approaches depend on personal interviews and the full-profile approach is considered to give a more realistic view of alternatives. Hence, the greater the number of attributes examined, the greater the number of potential comparisons that need to be assessed, so limits need to be placed on the number of attributes that can be realistically examined.

For example, in the deep-water corals case study considered, a full factorial design of the four main attributes identified yields a total of 128 possible combinations or profiles, where two attributes have four levels and two attributes have three levels. [4] In the north-sea case study a full factorial design of the three main attributes identified yields a total of 81 possible combinations or profiles, where three attributes have four levels.[5] Due to the need for a respondent to asynchronously consider all profiles, only a fraction of the possible factorial combinations can be used to estimate the main effects and selected interaction effects. This is achieved by selecting an orthogonal, fractional factorial design. Most stated preference research employs main effects plan that preclude the analysis of interaction effects between attributes. The plan uses statistical techniques to select a subset of all possible factorial combinations, which will have proper representation of the full set. Orthogonality of the design ensures that individual estimates of the respective attributes and levels are independent of each other.

6.3.5. Methods of analysis

A premise of the CE approach is that choices can be modelled as a function of the attributes of the alternatives relevant to a given choice problem using random utility theory. There are several methods that have been proposed for the estimation of the parameters, including MONANOVA (monotone analysis of variance) for full factorial designs and OLS and LOGIT-based approaches for fractional factorial designs also.

In this study, we will use a multinomial logit (MNL) model.[6] The MNL model is derived from the assumption that the error terms of the utility functions are independent and identically Gumbel distributed.[7]

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[4] Thus there are $(2^4).(2^3)$=128 possible alternatives.

[5] Thus there are (3^4)=81 possible alternatives.

[6] Also termed conditional logit model.

The independence from irrelevant alternatives (IIA) assumption is a key property. IIA infers that the ratio of the probabilities of any two alternatives is independent from the choice set, i.e. that the ratio of the probabilities of choosing any two options will be unaffected by the attributes or availability of other options. As noted by Train (2003), Luce (1959) considered IIA to be a property of appropriately specified choice probabilities. However, in general application it is as a resulting property of the logit model that IIA is evaluated (Train, 2003), by fitting the model that contains cross-alternative effects and examining the significance of these effects. As a result, it may not be an appropriate test for all choice situations (Train, 2003).

Contingent valuation data will be analysed using the MNL procedures. Depending on the dichotomous type of answers that will be collected in the questionnaire for the WTP values, this procedure will be more suited for data analysis.

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[7] The Gumbel distribution is used to find the minimum (or the maximum) of a number of samples of various probability distributions.

Annexes to PROTECT Periodic Activity Report - Section 2

Annex 1:	H.-H. Hinrichsen, R. Voss, K. Wieland, F. Köster, K. H. Andersen, P. Margonski Spatial and temporal heterogeneity of the cod spawning environment in the Bornholm Basin, Baltic Sea. (Accepted in *Marine Ecology Progress Series).*
Annex 2:	Eske Teschner and Gerd Kraus: Spatio-temporal stock trends of Baltic sub-components derived by disaggregated MSVPA.
Annex 3:	Per J. Sparre: Assessing the effect of Marine Protected Areas by the TEMAS model
Annex 4:	Jesper L. Andersen: Description of the bioeconomic model BEMCOM.
Annex 5:	Christensen,A., Daewel,U., Jensen,H., Mosegaard,M., St. John,M. and Schrum, C.: Hydrodynamic backtracking of fish larvae by individual-based modelling (submitted to *Marine Ecology Progress Series*).
Annex 6	Greenstreet, S., E. Armstrong, H. Mosegaard, H. Jensen, I. Gibb, H. Fraser, B. Scott, G. Holland and J. Sharples (2006). Variation in the abundance of sandeels Ammodyes marinus off southeast Scotland: an evaluation of area-closure fisheries management and stock abundance assessment methods. ICES J. Mar. Sci., Ices J Mar Sci 63, 1530-1550.
Annex 7.	Frederiksen, M. & Wanless, S. 2006. Draft report, PROTECT Work Package 5, Case Study 2: Assessment of the effects of the Firth of Forth sandeel fishery closure on breeding Seabirds.
Annex 8:	Armstrong, C.W. A note on the ecological–economic modelling of marine reserves in fisheries. *Ecological Economics* (in Press).
Annex 9:	Prem Wattage: Draft Environmental valuation questionnaires: (A) Cold Water Corals, (B) Seabirds, fish and MPAs.

Annex 1 Spatial and temporal heterogeneity of the cod spawning environment in the Bornholm Basin, Baltic Sea

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Running head: Heterogeneity of the Baltic cod spawning environment

Abstract

This study quantifies the spatial heterogeneity of the environmental conditions associated with successful spawning by cod in the Bornholm Basin. Quarterly means of the thickness of reproductive volume enabling egg survival in 1989-2003 indicate that most favourable spawning conditions were located in the central area inside the 80-m isobath. On average, spatial patterns were similar in the 2nd and 3rd quarters, but with overall lower thicknesses of reproductive volume and less horizontal extension of conditions suitable for egg survival in the 3. quarter. The observed basin wide variation in thickness of reproductive volume and oxygen content inside this volume can result in marked horizontal differences in oxygen related egg mortality, especially during stagnation years. The spawning habitat selected by adult Baltic cod in the Bornholm Basin was characterized by comparing data on egg abundance with environmental variables measured concurrently to the egg collection. A clear preference for spawning at locations in the deep basin is evident after both inflow events as well as for stagnation periods. In consequence, the closed area for fishing implemented in the Bornholm Basin during main spawning perods 1995-2003, although located in the northern center of the basin, did not necessarily ensure undisturbed spawning in stagnation years.

Key words: Baltic cod, environmental conditions, reproductive volume, egg survival probability

Introduction

The continuous fresh water runoff from a number of large rivers along the Baltic coast, accompanied by irregular events of saline water inflows, cause a permanent stratification of the Baltic Sea water body as far north as the Aaland Sea (Elmgren 1984). The effective separation of surface- and deep-water caused by differences in density hampers mixing and thus transport of oxygen from the upper layers down to the deep areas. Oxygen depletion is thus a common phenomenon over large areas in the deep basins, which affects the reproduction of cod through limiting the survival of eggs (Nissling et al. 1994; Wieland et al. 1994). Neutral buoyancy and peak abundance of cod eggs occurs in the region of the halocline, with some quantities occurring in the more saline deep layer (salinity > 11 psu; Kändler 1949; Müller and Pommeranz 1984; Wieland 1995). These conditions are met exclusively at locations with water depth deeper than 60 m (Wieland et al. 1994). Cod eggs are regularely exposed to water being below the critical oxygen limit for successful egg development (oxygen > 2ml/l, Nissling and Westin 1991, Bagge et al. 1994).

A recently developed oxygen-related survival factor (Köster et al. 2005) considers not only the volume potentially sustaining egg development, but gives a measure of the oxygen conditions within the salinity range over which cod eggs are neutrally buoyant.
Temperature controlled laboratory experiments have been performed to describe the influence of oxygen concentration on the proportion of viable hatch in relation to the surviving fraction at norm-oxic conditions (Rohlf 1999). These experiments confirmed that oxygen concentrations above the threshold level of 2 ml/l utilized in the definition of the Baltic cod reproductive volume (RV) have a strong positive impact on egg survival. A sigmoid oxygen-egg survival relationship (Köster et al. 2005) can be applied to estimate the fraction of the egg production probably surviving during the spawning season to the larval stage.

Compared with other stocks in the North Atlantic, the eastern Baltic cod stock has a lengthy spawning period (ICES 2005). This behavioral pattern has been explained as an adaption to the variable and sometimes critical environmental conditions in the Baltic Sea (e.g.MacKenzie et al. 1996). The time of peak spawning identified on basis of egg abundance estimates in the Bornholm Basin occurred between the end of April and mid June in the 1970s and 1980s (Wieland et al. 2000a). From the early 1990s onwards the time of highest egg production has gradually changed to later summer months. This observed shift in the main spawning time may have been caused by i) removal of early spawners by the fishery mainly directed to pre-spawning and early spawning concentrations (ICES 1999/ACFM:15), ii) a coupling of maturation processes at ambient temperature as reported by Wieland et al. (2000a), and iii) and by a decline of the calanoid copepod *Pseudocalanus acuspes* on which especially early spawned cod larvae relied upon as food (Hinrichsen et al., 2002a), so that only late spawners had a chance of successful reproduction.

Historically, there have been three main spawning areas for the central Baltic cod stock: the Bornholm Basin, the Gdansk Deep, and the Gotland Basin (Fig. 1). Recent analyses of spatial and temporal heterogeneity of RV have recognized that conditions for successful cod-egg development are most likely to be found in the Bornholm Basin (MacKenzie et al. 2000). However, exceptions to this broad pattern do occur. During the longest recorded period without a major Baltic inflow of North Sea water (1977 – 1993) RV in the Bornholm Basin became very low. On the other hand, oxygen renewal events (e.g. inflows) can rapidly and significantly improve conditions in all basins (Matthäus and Lass 1995). However, the most recent years showed that the direct effect of a single water inflow is limited since i) unfavourable hydrographic conditions may already return in the following year, i.e. the improvement for cod is restricted to one spawning season only, or ii) inflowing oxygenated water replaces only lower parts of the bottom water resulting in an intermediate water layer with sufficient salinity to keep cod eggs floating, but not sustaining their development due to low oxygen concentration. The hydrography in the Bornholm Basin during stagnation periods can be characterized by a high variability in oxygen concentration in bottom and halocline waters caused by regular smaller to medium sized (non-major) inflow events. Inflows mainly occurr during winter months, thus inflow situations are in general causing low bottom water temperatures as the inflowed water originates from surface areas in the western Baltic during winter (Hinrichsen et al. (2002b).

The present study describes the spatial environmental heterogeneity of the Baltic cod spawning habitat in the Bornholm Basin. We have calculated station-based averaged environmental variables describing the spawning habitat quality in order to identify longer-term spatial differences. We will present horizontal property fields associated with stagnation periods contrasted to more favourable spawning conditions strongly affected by inflow events. Our approaches are empirically based, and involve besides environmental variables also egg abundance data. Finally, the approach attempts to characterize spatial and temporal variability in an eastern Baltic cod spawning areas and its habitat related environmental parameters from 1989 to 2003 in the light of the applicability of closed areas for fisheries to ensure undisturbed spawning as enforced until 2003 (ICES, 2004/ACFM:17).

Material and Methods

Horizontal distribution patterns

The hydrographic and ichthyoplankton data set consists of measurements from 51 cruises carried out in the Bornholm Basin between May 1989 and November 2003. The station grid represents the Bornholm Basin enclosed by the 60-m isobath (Fig. 1). This station grid encompasses the historical distribution of peak egg and larval abundance of Baltic cod in the Bornholm Basin (Wieland 1995). The earliest cruises covered only 21 to 36 stations, while since 1994 45 standard stations were covered with a horizontal resolution between 8 and 10 nautical miles. The hydrographic survey data were used to calculate the Baltic cod spawning conditions. Physical parameters (conductivity, temperature, and oxygen) of the water column were usually measured with CTD/O_2 systems, while the horizontal distribution of cod eggs was based on Bongo hauls (oblique hauls, surface to 5 m from bottom; 300 and 500 µm mesh size; 60 cm diameter) which was deemed suitable for sampling eggs and larvae up to 10 mm length.

Based on the hydrographic standard station grids we calculated quarterly means of cod spawning conditions in the Bornholm Basin in terms of spawning layer thickness. Horizontal maps were constructed interpolating observed data onto a regular grid by objective analysis (Bretherton et al. 1976). An Gaussian isotropic covariance function

$$f(r) = \sigma^2 \exp(-r^2/R^2) \qquad (1)$$

was chosen with R the autocorrelation scale and σ the variance and r the distance between data points. A spatial autocorrelation function of the fields was derived from the isotropic covariance function with the parameter R determined from the fit of this function to the raw covariances of the field observations. Objective analysis has the advantage that it can make use of statistical results and assumptions concerning measurement noise and small-scale errors inferred from the observed data. Thus, at every single data point an estimate can be given that depends linearly on the total number of measurements, i.e. a weighted sum of all observations. It was assumed that the error due to measurement errors and small scale noise amounts to 15%

of the total variance of the fields. Only areas where the expected r.m.s (root mean square) error in the interpolation was < 50% of the standard deviations of the fields were considered. As most of the surveys were designed to produce quasi-synoptic maps of physical and biological properties, a unit array configuration with a horizontal resolution of dx = dy = 5 km was provided based on the standard station grid, whereby each of the grid points is representative of the analysed properties centered around it. Objective analysis provides a smoothed version of the original measurements, with a tendency to underestimate the true field because of specific assumptions involved in our treatment of measurement noise and small-scale signals unresolved by the observation array. Error estimates depend only on the statistics of the field, the noise level, and on the locations of the observation points, and not the measurements themselves.

Estimates of RV (salinity > 11 psu, oxygen > 2 ml/l, temperature > 1.5 °C) by subsequent horizontal integration are measures for the volume of water fulfilling minimum requirements for successful cod egg development (Nissling and Westin 1991; Plikshs et al. 1993). However, they do not directly consider the environmental conditions inside the water volume principally sustaining egg development. Therefore, oxygen-related cod egg survival (Köster et al.2005) has been considered in our analyses as an additional habitat related environmental parameter to characterize spatial variability of eastern Baltic cod spawning areas.

Selection of spawning habitat and habitat related environmental parameters

In order to assess and identify the preferred ranges of Baltic cod spawning habitat as well as the environmental variables within the spawning habitat, quotient curves derived from egg abundance data and individual environmental variables have been constructed. Each environmental variable was assigned a number of environmental categories, and the ratio of their percentage frequencies of occurrence of each categorie and the percentage of total number of eggs within each categories c was calculated

$$Q_c = \%\text{eggs}_c / \%\text{environmental variable}_c \qquad (2)$$

This analysis is based on the minimum number of hydrographic and ichthyoplankton stations carried out during the earliest cruises at the end of the1980s and the beginning of the 1990s. Between 18 and 28 categories of each environmental variable were chosen in order to ensure that the maximum occurrence per category did not exceed 15 to 20% of the measurements. Quotient values greater than 1 are considered as positively chosen categories and those less than 1 indicate avoidance of those environmental categories for spawning (van der Lingen et al. 2001). This approach was used to examine relationships between recently spawned cod eggs (stage Ia) and environmental variables representing the size, location as well as the quality of the cod spawning habitat. The locations of the earliest egg stage may vary from the distribution of the adults that are spawning, but the duration of the earliest egg stage is short (ca.2-3 days; Wieland et al.1994) and drift velocities at the level of cod egg occurrence are only low (a few kilometer per day; Krauss and Brügge, 1991). Thus, the horizontal distribution of newly spawned eggs provides a measure for the habitat selection of adult cod. For statistical evaluation, the adapted R library “Shachar” (developed by M. Bernal, Instituto Espapañol de Oceanografia, Centro Oceanográfico de Malaga, Apdo 285, Puerto Pesquero s/n, 29640 Fuengirola, Spain) was applied to include a randomisation test, which estimates the 95% confidence limits for rejecting the null hypothesis, i.e. that the observed quotient within a particular bin of an environmental variable is obtained by pure chance alone.

Results

Horizontal distribution patterns of spawning conditions and egg survival probability

Quarterly means of the thickness of reproductive volume enabling egg survival in 1989-2003 indicate that most favourable spawning conditions were located in the central deep area of the Bornholm Basin, i.e. inside the 80-m isobath. On average, spatial patterns were similar in the 2nd and 3rd quarters (Fig. 2), but with overall lower thicknesses of reproductive volume and less horizontal extension of conditions suitable for egg survival in the 3. quarter. As depicted in Fig. 3, highest variation in thickness of RV also occurred in the center of the Bornholm Basin, with a tendency of higher variability in the northern part. The spatial autocorrelation of mean and standard deviation of the RV varied for the x-direction between 51 and 63 km and between 33 and 40 km in y-direction.

The spatial distribution of early cod egg in the Bornholm Basin during peak spawning show highest abundances in the area enclosed by the 80m-isobath. Considerably lower abundance in intermediate depths (60 – 80 m) and in general an absence of eggs in areas < 60 m (Fig. 4). This egg distribution is not necessarily matching the best environmental conditions. Especially in stagnation years (1989, 1995, 2001 and 2002) highest abunadances of cod eggs were found outside the best spawning conditions. Pronounced reproductive volumes outside the central basin are not utilized for intensive spawning. The spatial autocorrelation scales are interannually highly variable and compared to the mean distribution of RV much lower (13 to 47 in x- and 13 to 29 km in y-direction) in size.

Cod egg survival probability in the RV in terms of the oxygen concentrations experienced by the cod eggs are presented in Fig. 5. On average, the highest cod egg survival probability is located in the north-western area between the island of Bornholm and the Swedish coast. Furthermore, high egg survival probability occurred in the northern area of the central basin. The southern central part of the basin as well as areas with water depths shallower than 80 m showed significantly lower survival probability. Spatial autocorrelation is similar to the thickness of RV with 46 km in x- and 29 km in y-direction. The observed basin wide variation in thickness of RV and oxygen content inside the RV can result in marked horizontal differences in oxygen related mortality, especially during stagnation years. The mean distribution of youngest egg stage (Ia) during main spawning periods 1989-2003 suggests main spawning effort in the centre and to a minor extent to the southeast of the Bornholm Basin leading to a mis-match of average spawning activity and optimal spawning conditions in terms of oxygen content in the RV.

RV characteristics during stagnation and post inflow periods

During stagnation years the thickness of RV was lower compared to mean conditions, while the horizontal variability in the thickness of RV was more pronounced (Fig. 5). On average, the RV had a thickness of < 15 m throughout most of the central basin. During inflow and post inflow years (1993, 1994, 2003) the size in the vertical extension was higher (> 20 m), lower thicknesses of the spawning volume were only recorded at the outer edge of the central basin. However, in 1994 the favourable oxygen conditions did not persist throughout the year, with a strong decrease of the RV from spring 1994 to summer 1995.

Selection of spawning habitat and habitat related environmental parameters

During both stagnation periods as well as after inflow events the stage Ia cod egg distribution indicates that spawning mainly occurred within the deep basin of the Bornholm Basin at stations with water depths deeper than 80 m (Fig. 6). During stagnation years there is a significant preference for stations with vertical extents of RV at the 25 and 29 m levels, whereas during inflow years there are two less pronounced significantly positive categories – one mid and one high category. Note that during inflow years the mid level is equal to the highest level during stagnation periods and was mainly found at stations with water depths deeper than 80 m. Generally, high concentrations of newly spawned eggs were found in the central basin after saline and oxygen-rich inflows and lower abundances occurred at the edges of the basin, accompanied by decreasing values of RV thickness. During stagnation periods, the highest concentrations of the youngest egg stage were also found in the centre of the basin, but the distribution was more widely spread and the overall egg abundance was lower than in post inflow years. No clear significant preference for spawning in well-oxygenated water masses is detectable during both stagnation and inflow years.

Seasonal development of reproductive volumes and egg survival probability
Hydrographic data collected in the months following the most recent major Baltic inflow in early 2003 illustrate the role that oxygen consumption can have on the seasonal development of the RV. From the data it is obvious that the oxygen concentrations near and below the halocline quickly decreased from March to August. (Fig. 7). Due to biological processes, the oxygen content at the bottom decreased from almost 6 to 3 ml/l. Oxygen depletion in the upper halocline (50 to 70 m) as well as above the halocline was totally different. Lateral advection of water masses (upper halocline) as well as temperature increase and mixing processes in the upper water layer are mainly responsible for the change in oxygen content. Although oxygen consumption in the volume of water suitable for cod egg development was significant, the RV decreased only from 217 km^3 in March to 200 km^3 in August 2003. However, the average oxygen related egg survival probability decreased from on average 78% in March to 23% in August.

Discussion

This present study quantifies the spatial heterogeneity of the environmental conditions associated with successful spawning by cod in the Bornholm Basin. On average, spatial patterns were similar in the 2nd and 3rd quarters, but with overall lower thicknesses of reproductive volume and less horizontal extension of conditions suitable for egg survival in the 3. quarter. The observed basin wide variation in thickness of reproductive volume and oxygen content inside this volume can result in marked horizontal differences in oxygen related egg mortality, especially during stagnation years. A clear preference for spawning at locations in the deep basin is evident after both inflow events as well as for stagnation periods.

Recruitment of Baltic cod critically depends on egg survival (e.g. Köster et al. 2001). Oxygen concentration at dwelling depths is an important factor affecting egg mortality rates. It has been recognized earlier that hydrographic conditions in the central and eastern Baltic are critical for successful reproduction of cod and that the inflow of saline and oxygenated water from the North Sea is a prerequisite for the formation of strong year classes (e.g. Kosior and Netzel 1989; Bagge et al. 1994). However, in order to evaluate this hypothesis statistically, it is necessary to quantify the water volume suitable for successful development of eggs, which represents a measure of suitable habitat size (Plikshs et al. 1993; MacKenzie et al. 2000). It has been demonstrated that the RV declines in general from spring to summer (MacKenzie et al. 1996), and since the timing of spawning has changed in the past decade from spring to summer (Wieland et al. 2000a), the RV represents the spawning volume correctly only if it is estimated for those times of the year which are relevant for the development of eggs. On the other hand, it has been recognized that the thickness of RV is not a good measure of oxygen-related egg survival in the Bornholm Basin (Köster et al. 2001), especially not, if measured only in the deep part of the basin as it has been done before the 1990s.

Although the RV has been successfully integrated into stock-recruitment relationships (e.g. Sparholt 1996; Jarre-Teichmann et al. 2000), the amount of variance explained in recruitment is limited and especially the major Baltic inflow in 1993 did not increase the reproductive success as expected (Köster et al. 2001). The RV may not be the best proxy for egg survival, as it does not i) consider the impact of sub-lethal oxygen deficiency on egg survival, ii) neglects the possibility that eggs float outside the RV in water layers not sustaining their development beyond a certain egg stage and iii) ignores a potential horizontal mis-match in spawning activity and the presence of the volume. Hinrichsen et al. (2002b) analysed the impact of physical forcing processes on variations in the volume of deep oxygenated water suitable for reproductive success of central Baltic cod. These processes suggested to revise the previous estimates of the RV by including variations in timing and volume of terrestrial runoff, variability of the solubility of oxygen due to variations of the sea surface temperature in the western Baltic as well as the influence of variations in wind stress.

The basic hydrographic processes affecting the environmental conditions for cod egg survival are understood (e.g. Matthäus and Schincke, 1999; Hinrichsen et al. 2002b), but reliable predictions of where and when egg survival, generating subsequent high recruitment as well, are yet not possible (ICES, 2004/ACFM:17). This is not only caused by uncertainties with respect to atmospheric forcing conditions and its hydrodynamic response, but mainly due to the fact that biological processes as distribution of spawning effort, egg mortality due to other effects than hydrography as well as larval and early juvenile mortality are complicating the recruitment process. Hydrographic features influence the spatial distribution of the spawning stock (Tomkiewicz et al. 1998) and the vertical distribution of the eggs (Wieland and Jarre-Teichmann 1997). Potential egg survival depends on the vertical overlap between the eggs and favourable hydrographic conditions (Andersen and Möllmann 2004). It would therefore be beneficial to take the vertical distributions of eggs into account when the egg survival is calculated. However, in the past those measurements were seldom performed as they are technically not so easy to conduct (e.g. Wieland et al. 2000b). A preliminary attempt was conducted by Köster et al. (2001), but even if the approach seems to hold for the Bornholm Basin it does not for the Gdansk Deep and the Gotland Basin, as estimated egg survival rates are too low compared to direct estimates of egg mortality and observed larval abundance (Köster et al. 2005). This can be explained either by underestimating buoyancy in these lower salinity eastern spawning areas in general or by ignoring spatial variability in hydrographic conditions enabling egg survival in specific parts of the basins, even if average conditions are unfavourable (Köster et al. 2005).

Direct determination of spawning habitats is often limited by low numbers of observations. However, in our example, the applied method of simple quotient rule analysis is based on a large number of surveys which have provided a considerable number of spatial egg distributions with correspondingly measured environmental data. Hence, our approach enables the characterisation of Baltic cod spawning habitat in terms of different environmental variables. The method provides information whether the selected environmental variables are randomly related to the spawning locations of Baltic cod or if they present statistically significant habitat choices. Such characterization of spawning habitat of Baltic cod could allow the identification of processes which are likely to alter the size and/or qualitity of the reproductive volume of Baltic cod. Furthermore, spawning habitat size and properties may also be predicted in the absence of biological data (egg or fish abundance) through physical field observation, i.e., the method might be useful to allow regular monitoring of space-time variability of size and location of the spawning habitat.

As a direct application of this approach, our study can be seen as an attempt to characterize spatial and temporal variability of eastern Baltic cod spawning habitats in the light of implemented closed areas to ensure undisturbed spawning. Fig. 8 represents the distribution of the RV and the abundance of the youngest egg stage in August 1991 and 1999. In 1991 a high horizontal overlap between RV and egg abundance was observed. In 1999 the highest RV was encountered in the northwestern part of the basin near the island Bornholm, while the main egg abundance was recorded in less favourable spawning environment at the eastern edge of the basin. The rectangle superimposed into Fig. 8 represents the area of fishing closure applied from 1995 to 2003 during main spawning season. If such a closed area had been used in August 1991 it would have protected most of the spawning fish from the fishery. On the other hand, in August 1999 both the horizontal distribution of the layer where spawning has been identified to be successful as well as the horizontal distribution of cod eggs did not match to the closed area. Setting-up of temporally and spatially closed areas is considered to be an efficient management measure to protect vulnerable fish stocks and key habitats. Protected areas are set up in zones and at times corresponding to specific features and/or events, such as sensitive habitats, large concentration of a species to be protected, easy identification or easy controllability. Although their theoretical benefits are often well identified, there is only little empirical evidence to demonstrate their real effectiveness, because proper monitoring and assessment requires detailed information that in many cases is not available or scarce. Similarly, the inter-annual variability in the distribution pattern of the spawners and surviving egg production of eastern Baltic cod makes it difficult to define an appropriate closed area within the Bornholm Basin as the recently most important Baltic cod spawning area. It is clear, however, that closures should target the deepest part of the basins, which are presently of importance for successful development and mortality of Baltic cod. The presently enforced closed area is enlarged and covers a larger part of the central Bornholm Basin, but still leaves the northwestern and the southern part of the central part of the central basin with water depths >80 m unprotected.

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Figure legends

Fig 1: Map of the Baltic Sea with standard station grid in the Bornholm Basin (45 stations)

Fig. 2. Mean horizontal distribution of the thickness of Baltic cod RV [m] in the Bornholm Basin: a) 2nd quarter and b) 3rd quarter

Fig. 3. Variation in mean thickness of Baltic cod RV in the Bornholm Basin (standard deviation; m)

Fig. 4. Thickness of Baltic cod RV [m] and egg stage Ia distribution in the Bornholm Basin (1989 –2003)

Fig. 5. Mean survival probability (color scale) and mean Baltic cod egg stage Ia distribution in the Bornholm Basin (1989 – 2003; dots; n/m²)

Fig. 6. Quotient rule analysis: (a) stagnation years, and (b) inflow years 1993, 1994 and 2003-bars quotient curves, dots –number of observations, dotted line- 95% confidence interval, upper panel: thickness of reproductive volume, middle panel: o_2-oxygen content in the Baltic cod reproductive volume, lower panel: bottom depth), full line represents quotient values of 1 on log-scale

Fig. 7. Sequence of horizontally averaged oxygen profiles in the deep central Bornholm Basin between March and August 2003

Fig. 8. Thickness of of Baltic cod RV (color scale; m) and Baltic cod egg stage Ia distribution (dots, n/m²) in the Bornholm Basin; (a) August 1991 and (b) August 1999. Rectangle represents closed area for all fisheries in 1995-2003 during main spawning season.

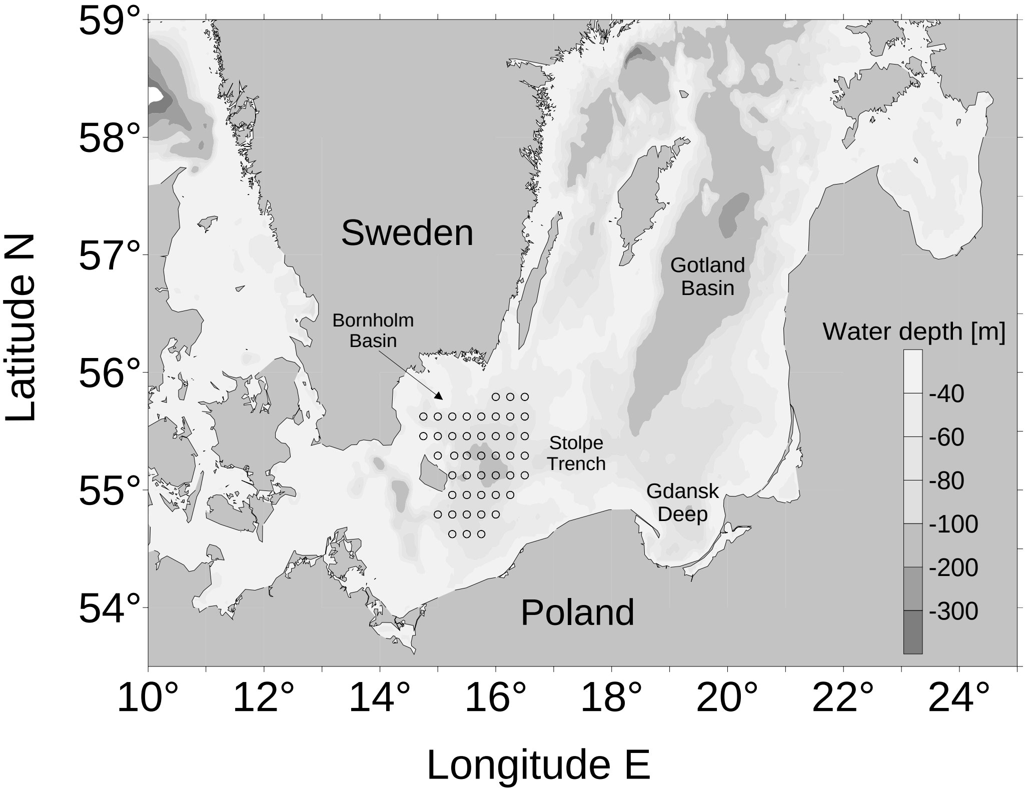

59°
58°
57°
56°
55°
54°
Latitude N
10°
12°
14°
16°
18°
20°
22°
24°
Longitude E
Sweden
Poland
Bornholm Basin
Stolpe Trench
Gotland Basin
Gdansk Deep
Water depth [m]
-40
-60
-80
-100
-200
-300

Fig 1

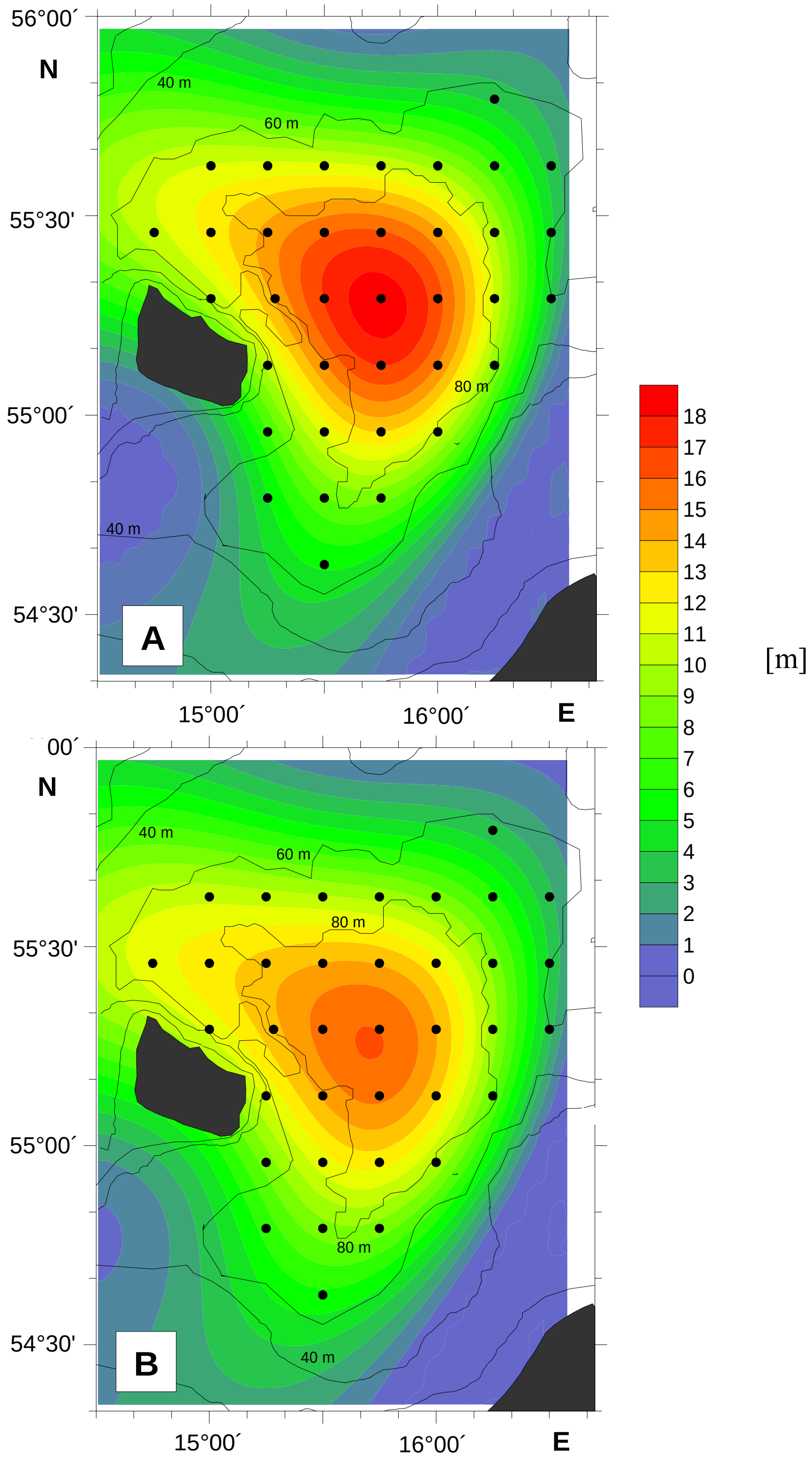
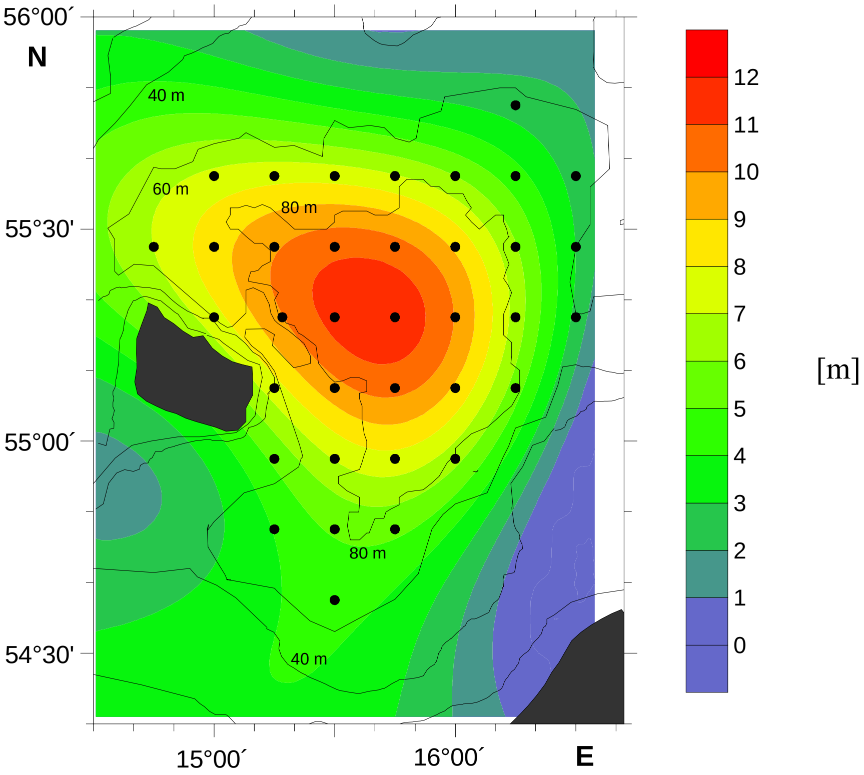

56°00´
N
40 m
60 m
55°30'
80 m
55°00´
40 m
54°30'
A
15°00´
16°00´
E
00´
N
40 m
60 m
80 m
55°30'
55°00´
80 m
54°30'
40 m
B
15°00´
16°00´
E
18
17
16
15
14
13
12
11
10
9
8
7
6
5
4
3
2
1
0
[m]

Fig. 2.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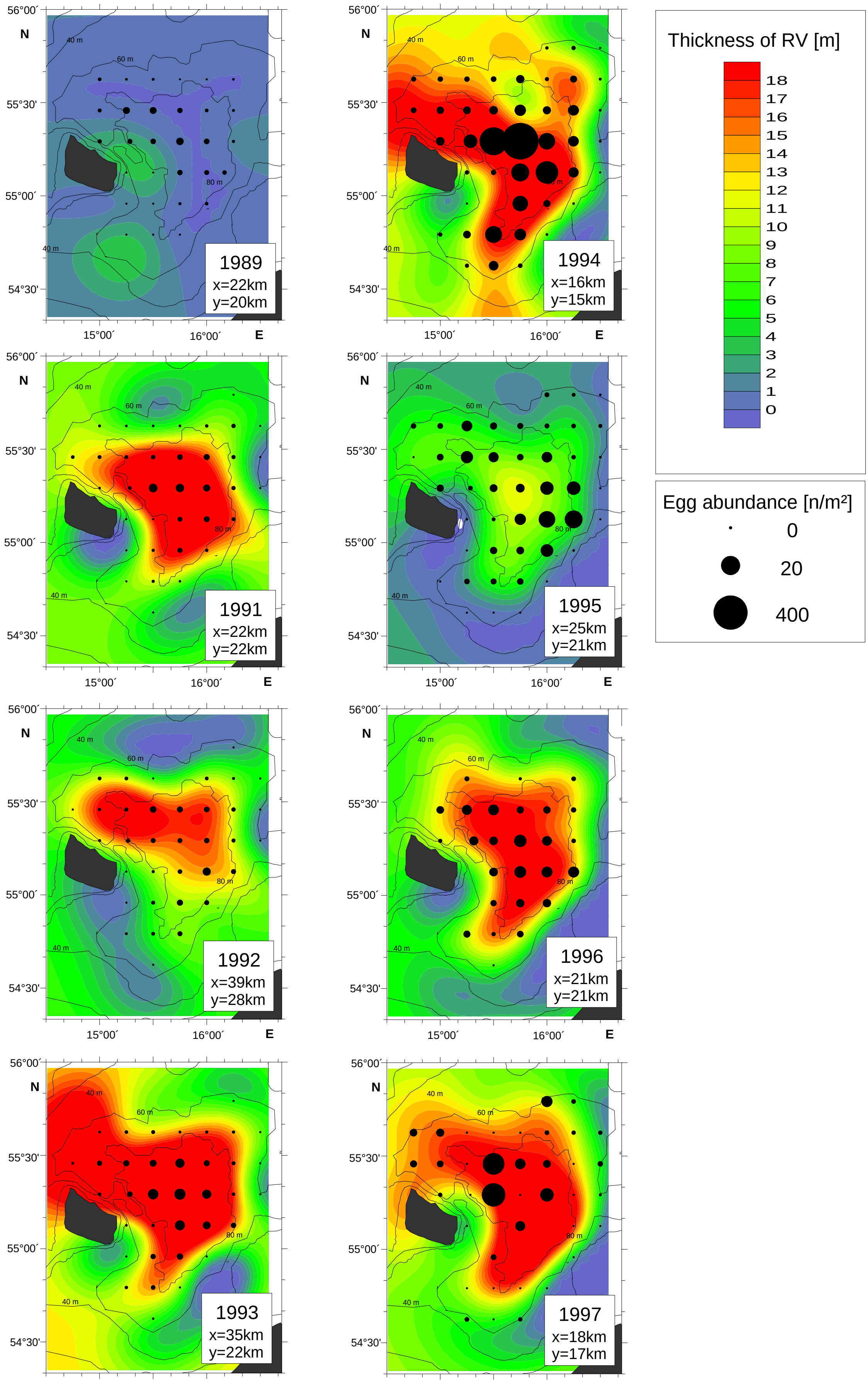
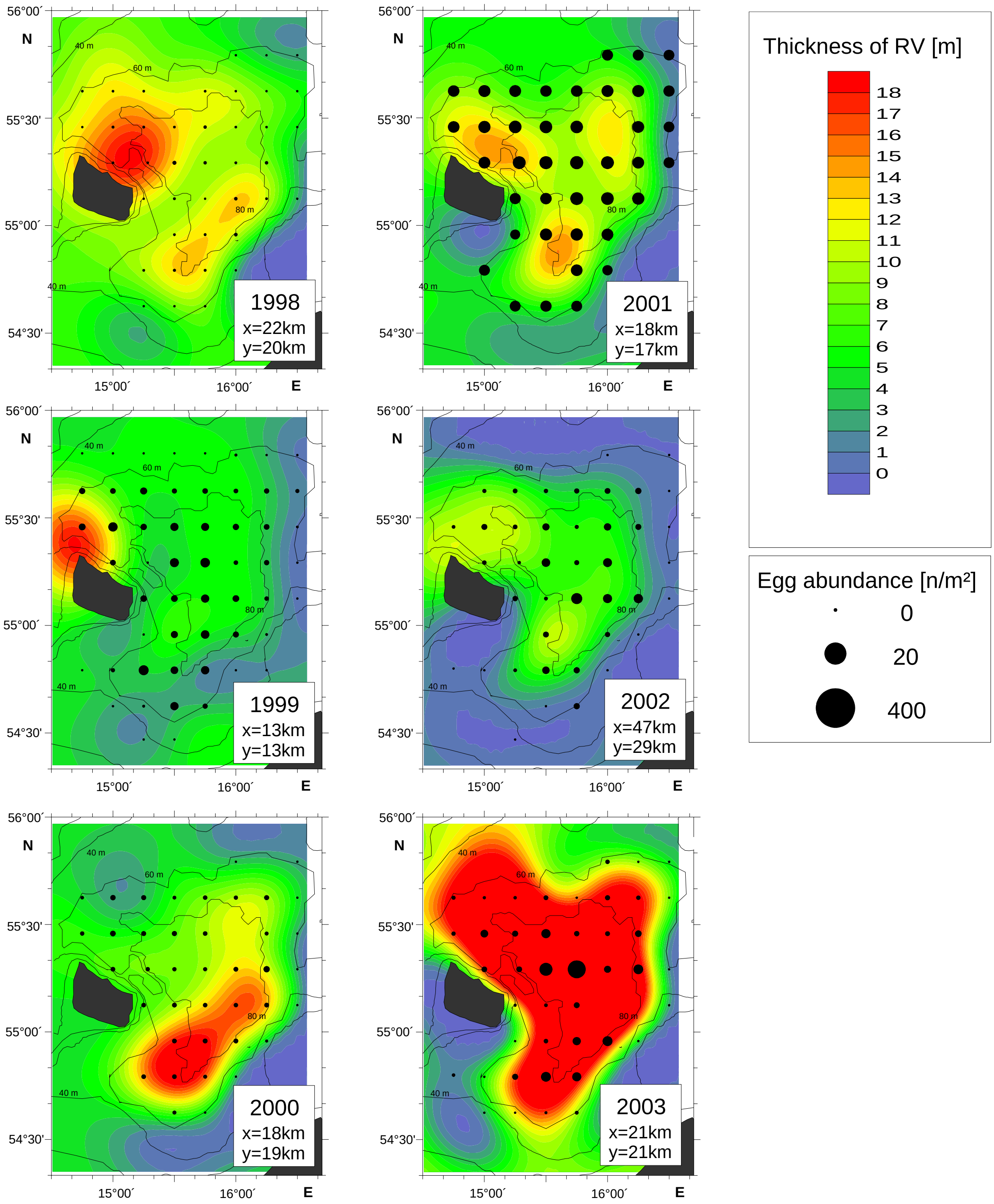
N
55°30'
55°00´
54°30'
15°00´
16°00´
E
40 m
60 m
80 m
80 m
40 m
12
11
10
9
8
7
6
5
4
3
2
1
0
[m]

Fig. 3.


56°00´
55°30'
55°00´
54°30'
15°00´
16°00´
N
E
40 m
60 m
80 m
1989
x=22km
y=20km
1994
x=16km
y=15km
1991
x=22km
y=22km
1995
x=25km
y=21km
1992
x=39km
y=28km
1996
x=21km
y=21km
1993
x=35km
y=22km
1997
x=18km
y=17km
Thickness of RV [m]
18
17
16
15
14
13
12
11
10
9
8
7
6
5
4
3
2
1
0
Egg abundance [n/m²]
0
20
400


56°00´
55°30'
55°00´
54°30'
N
E
15°00´
16°00´
40 m
60 m
80 m
1998
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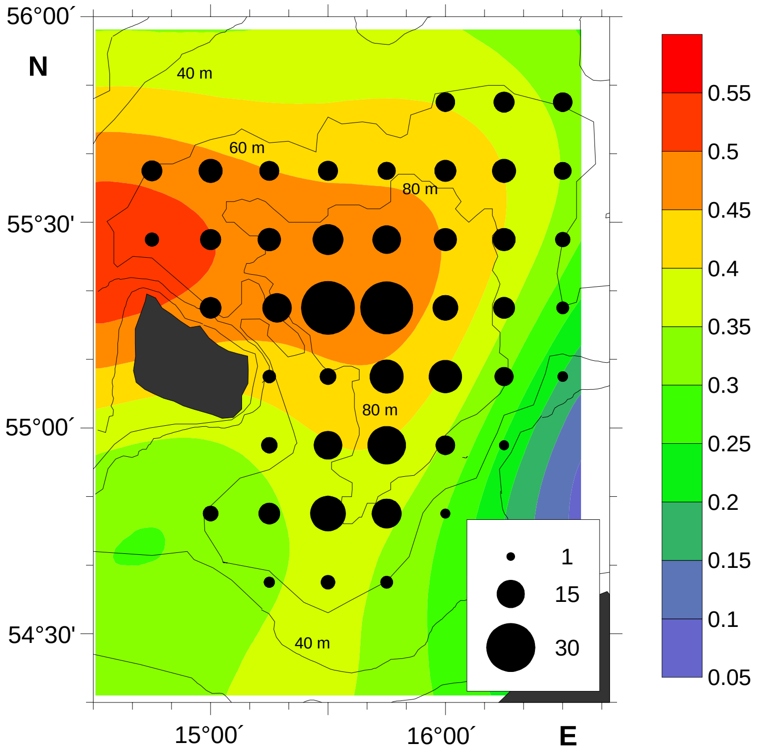
y=20km
2001
x=18km
y=17km
1999
x=13km
y=13km
2002
x=47km
y=29km
2000
x=18km
y=19km
2003
x=21km
y=21km
Thickness of RV [m]
18
17
16
15
14
13
12
11
10
9
8
7
6
5
4
3
2
1
0
Egg abundance [n/m²]
0
20
400

Fig.4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N
55°30'
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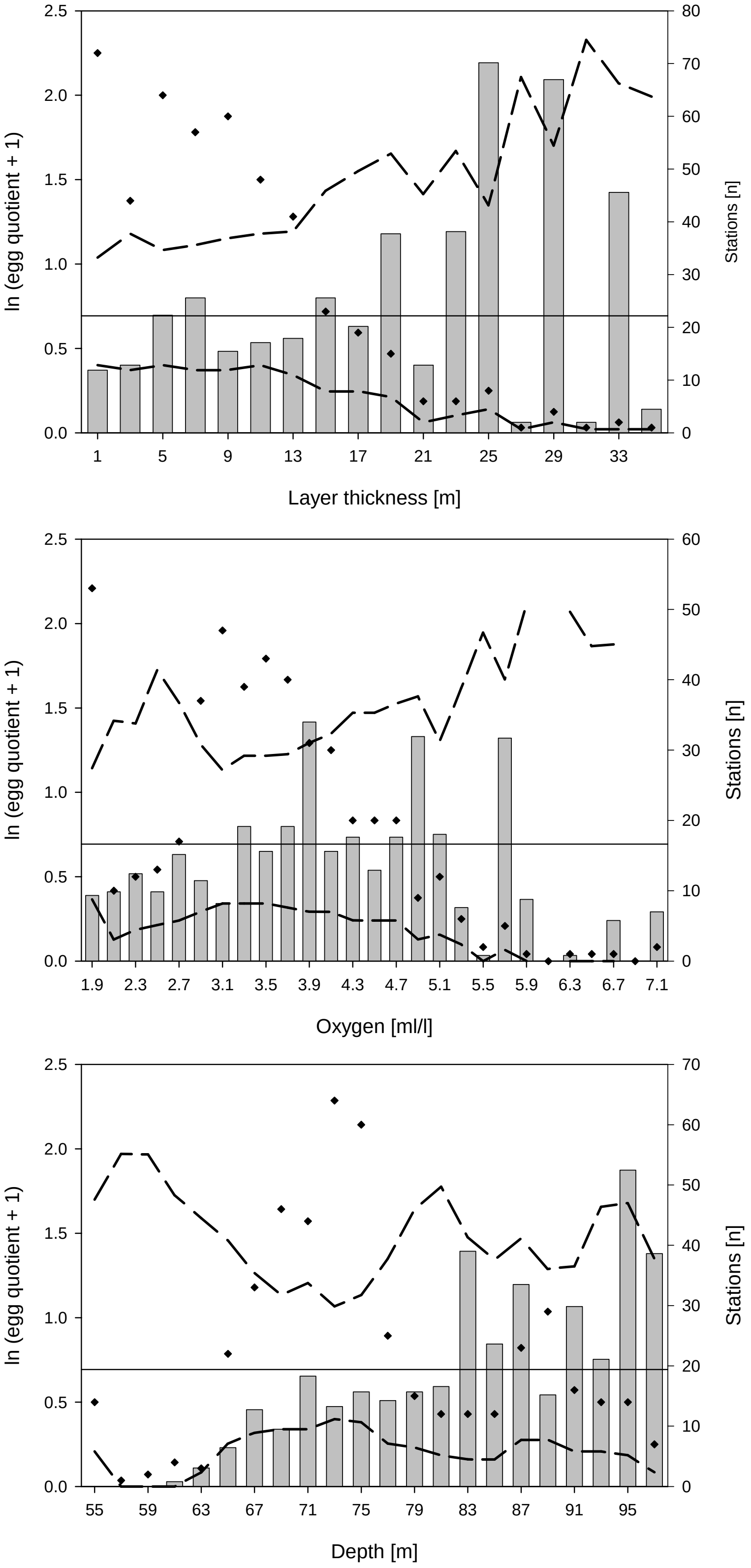
16°00´
E
40 m
60 m
80 m
80 m
40 m
1
15
30
0.55
0.5
0.45
0.4
0.35
0.3
0.25
0.2
0.15
0.1
0.05

Fig. 5

(a) Stagnation years



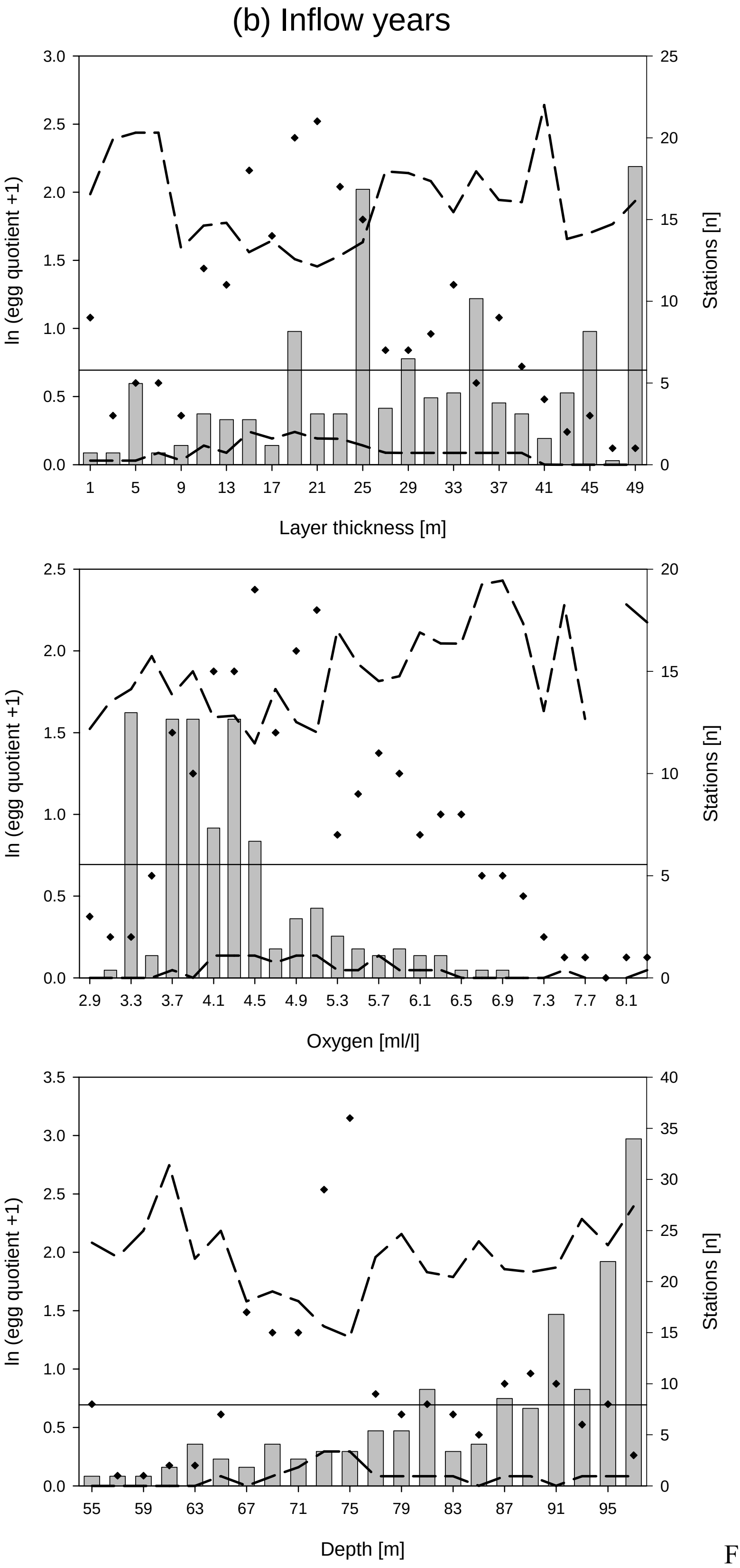

(b) Inflow years
ln (egg quotient +1)
Stations [n]
0.0
0.5
1.0
1.5
2.0
2.5
3.0
0
5
10
15
20
25
1
5
9
13
17
21
25
29
33
37
41
45
49
Layer thickness [m]
ln (egg quotient +1)
Stations [n]
0.0
0.5
1.0
1.5
2.0
2.5
0
5
10
15
20
2.9
3.3
3.7
4.1
4.5
4.9
5.3
5.7
6.1
6.5
6.9
7.3
7.7
8.1
Oxygen [ml/l]
ln (egg quotient +1)
Stations [n]
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
0
5
10
15
20
25
30
35
40
55
59
63
67
71
75
79
83
87
91
95
Depth [m]

Fig. 6.

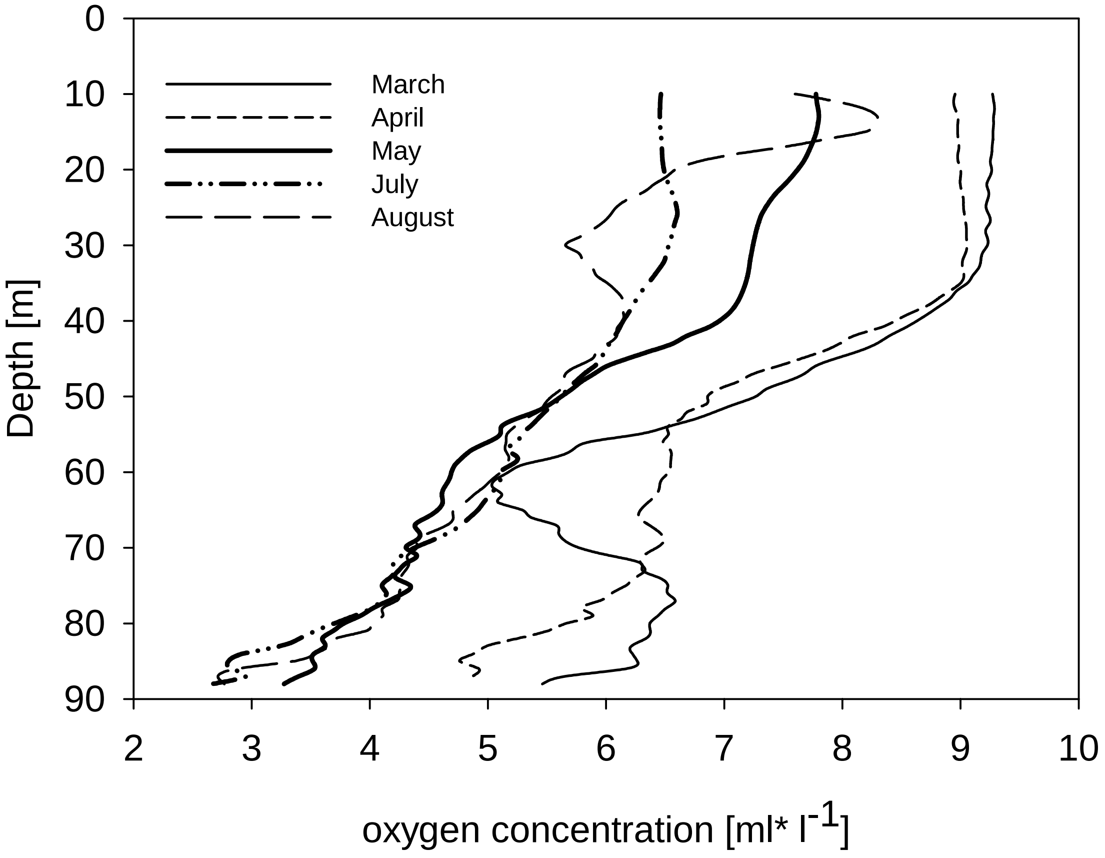

0
10
20
30
40
50
60
70
80
90
Depth [m]
2
3
4
5
6
7
8
9
10
oxygen concentration [ml* l^{-1}]
March
April
May
July
August

Fig. 7.

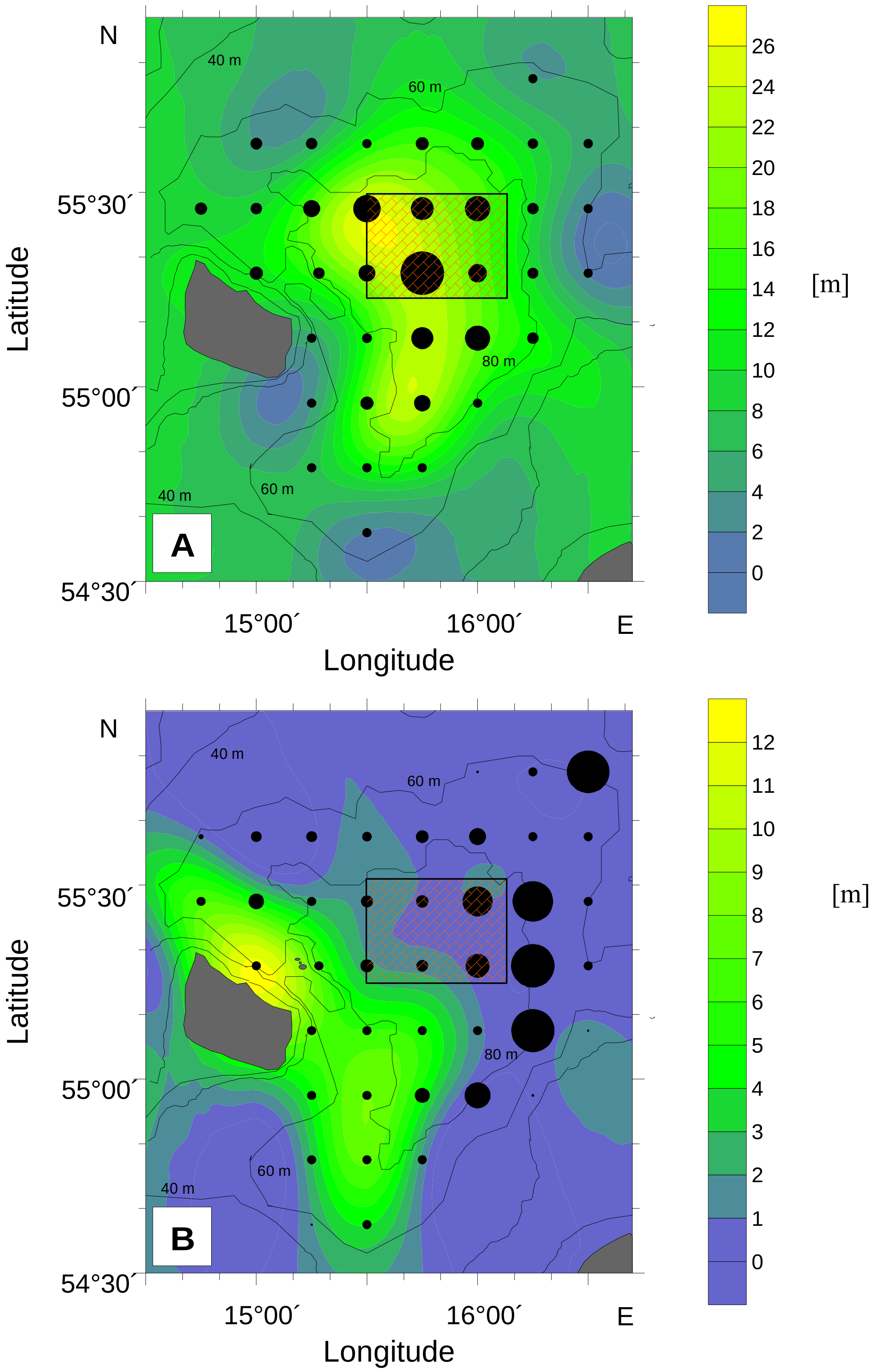

N
40 m
60 m
80 m
40 m
60 m
A
55°30´
55°00´
54°30´
Latitude
15°00´
16°00´
E
Longitude
26
24
22
20
18
16
14
12
10
8
6
4
2
0
[m]
N
40 m
60 m
80 m
60 m
40 m
B
55°30´
55°00´
54°30´
Latitude
15°00´
16°00´
E
Longitude
12
11
10
9
8
7
6
5
4
3
2
1
0
[m]

Fig. 8

Annex 2: Spatio-temporal stock trends of Baltic sub-components derived by disaggregated MSVPA

Eske Teschner and Gerd Kraus, IfM-GEOMAR

1. Introduction

In the Baltic Sea the spatial and temporal suitability of the spawning habitats of cod (*Gadus morhua*) vary dramatically with the oxygen conditions at the depth of incubation of the eggs (e.g., Wieland *et al.* 1994). As a consequence, different stock components of cod exhibit distinct trends in different areas of the Central Baltic (Sparholt and Tomkiewicz 2000), with a corresponding variation in predation pressure on its major prey species, sprat (*Sprattus sprattus*) and herring (*Clupea harengus*) (Sparholt 1994). In turn the population development of these planktivores determines the predation intensity on early life stages of cod (Köster and Möllmann 2000). Hence, a prerequisite for the implementation of effective MPAs is to resolve the spatial and temporal dynamics of cod and sprat as a basis for defining spatial and temporal windows for possible MPAs. This requires analyses on distribution and abundance patterns of adult fish relative to key environmental conditions and habitats. In the Baltic Sea, the MSVPA represents a suitable tool for calculating adult stock sizes of cod, sprat and herring taking into account species interactions. At present MSVPAs are run for two areas in the Baltic, a Western and Central Baltic component to match the stock units used in the regular stock assessments, with the Central Baltic component dominating in terms of biomass and abundance (ICES 1998/ACFM:16). Within these two regions, the abundance and biological characteristics of the three species are heterogeneous both spatially (between Sub-divisions) and temporally (inter and intra annually). For example, population sizes of Central Baltic cod, as resolved by international bottom trawl (Sparholt and Tomkiewicz 2000) and ichthyoplankton surveys (Köster *et al.* 2001a), have revealed distinct distributional trends. The abundance and characteristics of herring and sprat have also been observed to vary spatially and temporally in the different Sub-divisions of the Central Baltic (e.g., Ojaveer 1989). The herring stock in the Central Baltic is comprised of a number of different spawning components exhibiting variations in spawning period and growth rates as well as meristic, morphometric and otolith characteristics (e.g., Parmanne *et al.* 1994). For sprat the existence of distinct populations is controversial as deviations in growth rates observed between sub-areas have been explained by immigration from the western Baltic and by migration between different basins (Parmanne *et al.* 1994). However, other authors state that sprat in the eastern Central Baltic form local populations (Ojaveer 1989), which can be separated, primarily by otolith characteristics (Aps 1981).

Consequently, the Baltic Sea area would particularly be suitable for establishment of a *spatially resolved* multispecies model to quantify the stock dynamics of cod, herring and sprat in the different Subdivisions, displaying distinct environmental conditions. These requirements are met by the area-disaggregated MSVPA, which has a spatial resolution of single Sub-divisions (SD), i.e. SD 25 (Bornholm Basin), SD 26 (Gdansk Deep) and SD 28 (Gotland Basin). Thereby the 3 historic major spawning areas, each with characteristic hydrographic conditions and stock trends, are separated.

2. Material and Methods

The disaggregated MSVPA Setup

2.1 Stock units

Disaggregated MSVPAs were conducted in Subdivisions 25, 26 and 28; here all species were assumed to be unit stock components:

Cod composed of age-groups 0-8 (oldest age-group as plus group), preying on herring and sprat. Herring and sprat were defined as prey in age-groups 0-8 (oldest age-group as plus group) for herring and age-groups 0-7 for sprat. Exhibiting cannibalistic behaviour, cod was also considered as prey in the MSVPA of the Baltic.

Input Data for the disaggregated MSVPA

2.2 Weights and numbers

Quarterly catch-at-age in numbers and weight-at-age in the catch according to Subdivisions were revised and updated for years 1976–2003 following the compilation scheme presented in ICES (1997b). Input for 2004 and 2005 was based on national data reported to WGBFAS (ICES 2004 and 2005). Missing data on weight-in-the-catch of cod for age-classes 0 and 1 were substituted in the 3rd quarter by a value of 0.028 in Subdivision 25 and 0.005 in Subdivision 26 and 28. In the 4th quarter a value of 0.028 was used for all Subdivisions.

Any other missing values on weight-in-the-catch were substituted by a mean of neighbouring years for herring and sprat and by a weighted mean of the sub-divisions for cod. Weight-at-age in the catch was assumed to be equal to weight-at-age in the sea, exceptions being weight-at-age for cod age-groups 0-2.

Here, due to size selection by commercial gear, mean values for two time periods (1977-1989 and 1990-2005) were used. For the years from 1998 up to 2005 the values of the second time period were used (Tab. 1).

Tab. 1: Values for cod weight-at-age in the stock used in all SDs

All SDs	Time period 1977 – 1989			Time period 1990 – 2005		
	Age 0	**Age 1**	**Age 2**	**Age 0**	**Age 1**	**Age 2**
1st Quarter		0.065	0.206		0.052	0.226
2nd Quarter		0.073	0.242		0.090	0.339
3rd Quarter	0.005	0.089	0.310	0.005	0.138	0.425
4th Quarter	0.028	0.125	0.460	0.024	0.195	0.520

2.3 Maturity ogives

Maturity ogives for cod in different Subdivisions represent averages over 5 years periods available from 1980 (applied also before 1980) onwards for combined sexes as presented in ICES (1998/ACFM:16), updated with data for 1998 and 1999 presented in ICES (1999b) and ICES (2000/ACFM:14) and 2000 to 2003 as presented in the last SGMAB-Report (ICES, 2005). For the years 2004 and 2005 the values of the last year were used. According to ICES (1998) the maturity ogives for herring and sprat stocks were assumed to be constant over time.

2.4 Stomach content data

Quarterly cod stomach content data according to Subdivision as revised in ICES (1997/J:2) were utilized as input. Intra-cohort cannibalism in cod was excluded by changing prey age to predator age minus 1 and omitting 0-group cod in 0-group cod stomachs.

2.5. Quarterly food intake by cod

Quarterly, age-specific consumption rates of cod were estimated as described in Temming (1996) and ICES (1997a) for each sub-division (see 2.2). Also alternative (consumption rates with effect of oxygen on evacuation) quarterly, age-specific consumption rates of Baltic cod were calculated (see 2.3).

2.6. Other input data

The residual natural mortality (M1) was assumed to be 0.2 year^{-1} equally distributed over quarters corresponding to standard MSVPA runs in the Baltic (Sparholt, 1991). The Suitability coefficients were estimated according to standard suitability submodel implemented in the Baltic MSVPA (ICES, 1997b). The constant biomass of "other food" assumed to be 1 million tons, similar to ICES (1996/Assess: 2).

Tuning

The tuning of the MSVPA was performed for each Subdivision utilizing the procedure developed by Vinther (2001), iteratively running MSVPAs and XSAs with automatic recursive data exchange. Abundance indices utilized for tuning originated from the Baltic International trawl survey (BITS) directed to cod and the International hydroacoustic survey (BIAS) directed to sprat and herring.

The XSA settings were as follows:

Cod:

Including age-groups 2–8 abundance indices from international bottom trawl surveys 1994–2005, catchability was set to be dependent of stock size for ages < 3 and independent of age > 5, shrinkage of the terminal population towards a mean F over last 5 years and 3 oldest ages was applied with a standard error of 0.5–0.8, otherwise default settings of the Lowestoft assessment programme package were used.

Sprat:

Using international hydroacoustic survey results as tuning fleets; depending on the performance covering 1987 or 1992 to 2005 with year 1993 excluded, as insufficient area coverage and problems in the intercalibration of the equipment occurred (ICES 1997/Assess:12), catchability was set to be dependent of stock size for ages < 3 and independent of age > 4, shrinkage of the terminal population towards a mean F over last 3–5 years and 3–5 oldest ages was applied with a standard error of 0.5–0.8, otherwise default settings of the Lowestoft assessment programme package were used.

Herring:

Using international hydroacoustic survey results as tuning fleets; depending on the performance covering 1982 or 1986 to 2005 with 1992/1993 excluded in Subdivision 25, 1993 in Subdivision 26, 1993 and 1997 in Subdivision 28 as insufficient area coverage and problems in the intercalibration of the equipment occurred (ICES 1997/Assess:2; ICES 2000/ACFM:14), catchability was set to be dependent of stock size for ages < 3 and independent of age > 5, shrinkage of the terminal population towards a mean F over the last 5–6 years and 6–7 oldest ages was applied with a standard error of 0.8–1.0, otherwise default settings of the Lowestoft assessment programme package were used.

3 Results and Discussion

Stock numbers

The time trend in stock numbers for cod, herring and sprat in the different Sub-divisions (SD) as determined by the three MSVPA runs are displayed in Fig. 3.1

Generally, an opposite trend in the stock performance of cod and sprat was obvious (Fig. 1; upper and lower panel), with a decreasing cod stock and an increasing sprat stock in the recent years. For cod the highest stock numbers could be recorded in the early 1980s with values of about 1 billion in SD 25 and 26 and of about 500 million in SD 28 (Fig. 1; upper panel). Cod abundance drastically declined in all areas since the beginning of the 1980s and since the mid 1990s cod is almost absent from SD 28. Since 1981 cod abundance was always highest in SD 25. Here, in the Bornholm Basin, stock fluctuations have been most pronounced during the recent low stock phase (since the beginning of 1990s). At low stock sizes, abundance in SD 26 showed only little variability, however with a slowly declining stock trend since the mid 1990s.

Sprat showed a slight decrease after peak population sizes in the mid 1990s in all Subdivisions. However, in SD 28 sprat abundance increased sharply from 2002 to 2004 (Fig. 1; lower panel), which is mainly caused by the strong 2003-year-class of sprat and secondary caused by unequal distribution of fishing effort between the three Subdivisions (high effort in SD 28 results in high catch numbers and following in high calculated stock numbers).

Estimated stock numbers for herring show slight decreases in population numbers in SD 25 and SD26 from the beginning of the 1980s to this day with present values of about 6.5 billions in SD 25 and 4.8 billions in SD 26. In SD 28 a slightly different time trend could be observed: After a general increase up to the year 1996 with maximum value of about 17.4 billions, the stock numbers decreased again to minimal levels of about 3.4 billions in 2005.

Presently, highest population numbers were calculated for cod and herring in SD 25 and for sprat in SD 28. Therefore the potential impact of sprat predation on cod egg mortality is highest in SD 28 due to the high stock numbers of sprat.

Population biomass

Estimates of population biomass of cod as well as herring and sprat are presented in Fig. 2. For cod a substantial decrease in the biomass is obvious in all three areas from 1983 onwards (Fig. 2; upper panel). In SD 28 the stock biomass declined from maximal values in the early years with no subsequent sign of recovery. In SD 25 and 26 stock biomass was in general higher. After a slightly enhanced reproductive success and a reduction in fishing mortality in the early 1990s, biomass increased again in both areas to an intermediate maximum in 1994/95, followed by a continuous decline until now.

The corresponding development of the herring biomass estimates is presented in Fig. 2, middle panel. For SD 25 a more or less continuous decline from the beginning of the time series to 1999 is indicated, with a slight increase afterwards. In SD 26 the herring biomass shows an obvious decrease from early years to 1988. Afterwards biomasses were significantly lower, with a slight declining trend. Contrary to the other SDs, the herring biomass in SD 28 showed a slight positive development up to the year 1993. Afterwards a continuous decrease in biomass was apparent.

The sprat biomass in SD 25 decreased until 1980 (Fig. 2; below). Afterwards a strong increase in biomass is obvious to maximum values in 1995, followed by a substantial decrease. The biomass of sprat in SD 26 and 28 showed a rather similar time trend, but not in the same order of magnitude. After a slight decrease, the biomass in both SDs increased in the beginning of the 1980s. In the early 1990s a further increase was obviously, followed by a decreased in biomass until now. Unlike SD 26, a strong increase in biomass was obvious in SD 28 from 2003 onwards.

Predation mortality rates

Estimates of predation mortality rates of juvenile cod and sprat are presented in Figs. 3-4. For cod in the period 1978-1985, i.e. when comparatively high stock levels existed, a substantial difference in the cannibalism level between the areas is apparent (Fig. 3), with lowest predation mortalities in SD 25. However, even when being comparatively low, the cannibalism rates in SD 25 are estimated to be still in the order of 0.49-0.89 for 0-group fish (second half of the year) and 0.22-0.40 for 1-group cod (yearly values), compared to corresponding values of 0.51-1.72 (0-group) and 0.32-0.79 (1-group) in both other areas. On the contrary, predation mortalities of 2-group cod were in general low, i.e. less than 0.15 per year, independent of the area. Since the mid 1980s predation mortalities were rather similar in SD 25 and 26 (0.11-0.38 and 0.07-0.16 for age-group 0 and 1, respectively). The area-specific deviation in cannibalism rates is coupled to a lower suitability of juvenile cod as prey of adult cod in SD 25. In SD 28, the predation mortality declined from high levels in 1983 to be virtually absent in the 1990s. This is related to the steady decline in the predatory adult cod stock throughout the last decade (see above).

Determined predation mortalities of sprat were rather similar in all three areas up to 1987 (Fig. 4). Maximum predation pressure on 0-group occurred in the first year of the analysis (1976) and afterwards in 1983 and on older age-groups in 1980. Independent of the prey age-group, a general decrease in predation until 1992 is obvious in all areas staying on a constant level afterwards. Predation mortalities were ranging in the beginning of the time series (since 1977) in the order of 0.18-0.31 for 0-group sprat (second half of year) as well as 0.72-1.14 and 0.24-0.65 for 1- and 2-5-group sprat respectively. For juvenile fish this is in the order of magnitude of the predation rates estimated for herring, however, considerably higher for adult fish. Lowest predation mortalities in 1991 in SD 25 and 26 were ranging from 0.04-0.06 for 0-groups and 0.07-0.12 for older fish. Corresponding values in SD 28 were close to zero throughout the 1990s.

Fishing mortality rates
Determined fishing mortality rates (for simplicity summed over quarters) of cod and herring (average over age-groups 3-6) as well as sprat (average over age-group 3-5) are displayed in Fig. 5. Fishing mortalities of cod in SD 25 were in general higher than in SDs 26 and 28. They were fluctuating between 0.4-1.66 without any clear time trend (Fig. 5, upper panel), with the exception that since 1997 the F-values were always at approx. 1.2, i.e. above the long-term average. For both other SDs, fishing mortality was most of the time lower but again no clear time-trend was obvious. Within the 1990s, even at low biomass values, no substantial decrease in mortality rates is obvious.

For herring, an increase in fishing mortalities from the beginning of the time series until the mid 1980s is estimated for all Sub-divisions (Fig. 5, middle panel). In SD 25 and 26, this level (approx. 0.2) was kept throughout the remaining time period covered, while for SD 28 this was followed by an increase to relatively high levels in most recent years.

The fishing mortalities determined for sprat were much more variable, than those determined for both other species (Fig. 5, lower panel), with maximum changes from 0.94 to 0.11 in successive years. Although the variability makes it difficult to detect any consistent time trends, an increase in fishing mortality since 1992 from 0.04-0.13 in 1993 to 0.34-0.58 in 1997 is obvious for all areas.

In conclusion, a potential MPA for cod seems currently to be most suitable in the SD 25, i.e. the Bornholm Basin due to (a) highest stock sizes of cod in this area, (b) the lowest impact of cannibalism on recruitment and (c) a lower stock size of sprat acting as potential predators on cod eggs (compared to SD 28).

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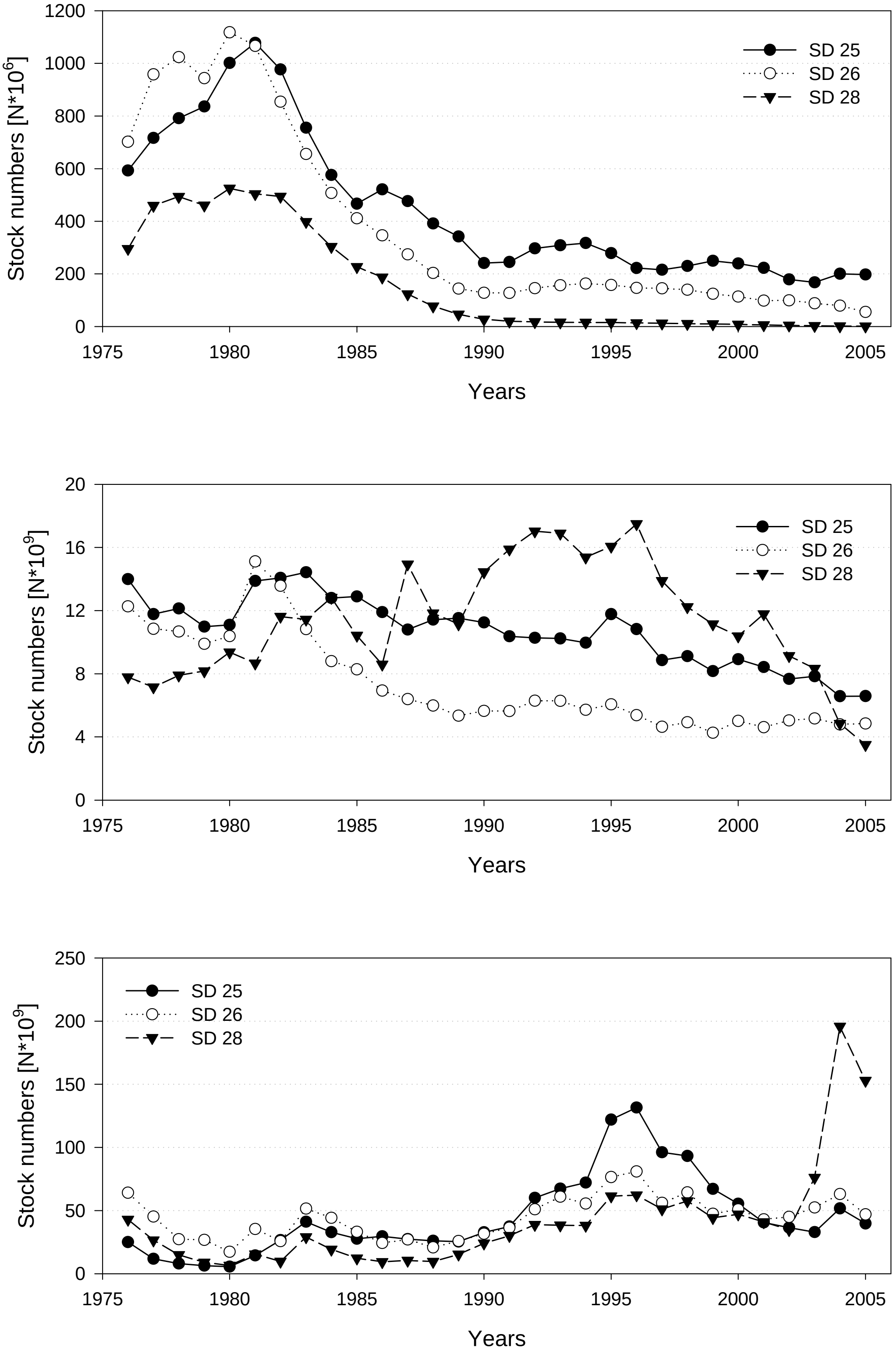

Fig. 1. Stock numbers for cod (above), herring (middle) and sprat (below) in the 1st quarter of each year summarized over age-group 1-8 for cod and herring and age-group 1-7 for sprat.

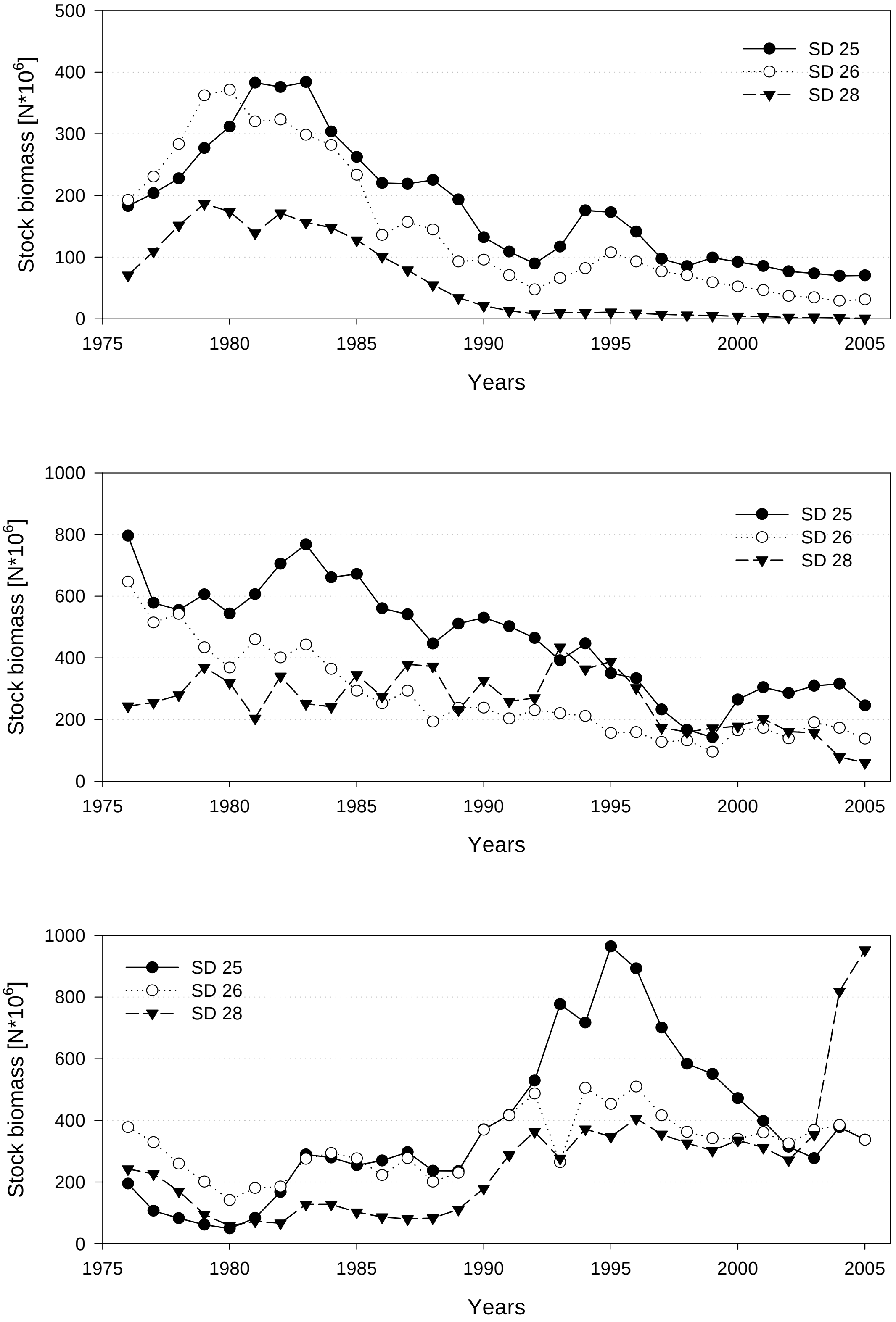

Fig. 2. Stock biomass for cod (above), herring (middle) and sprat (below) in the 1st quarter of each year summarized over age-group 1-8 for cod and herring and age-group 1-7 for sprat.

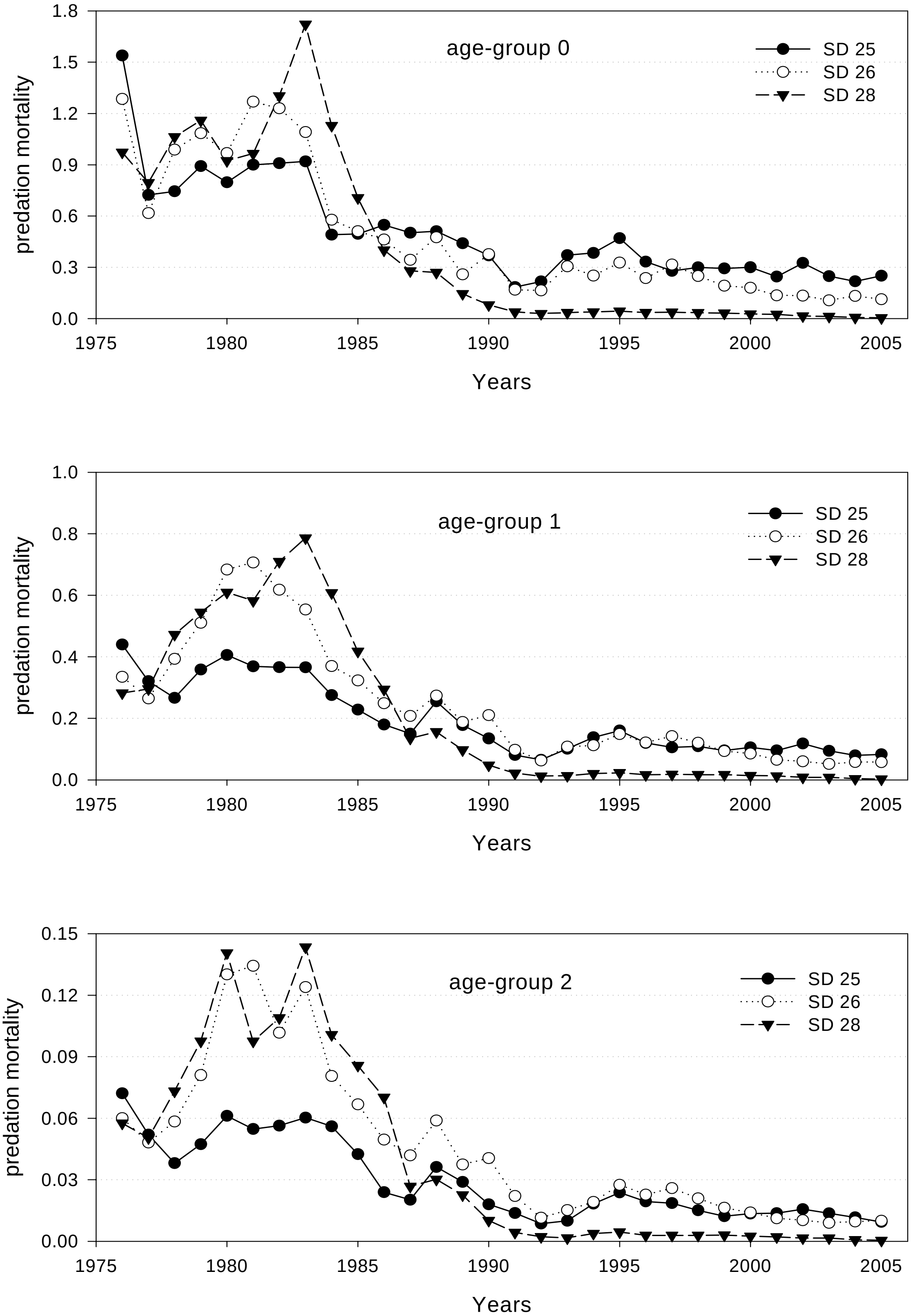

Fig. 3. Predation mortality rates of cod age-group 0 (3rd and 4th quarter; above), age group 1 (per year; middle) and age group 2 (per year; below) in different Subdivisions (SD) of the Central Baltic

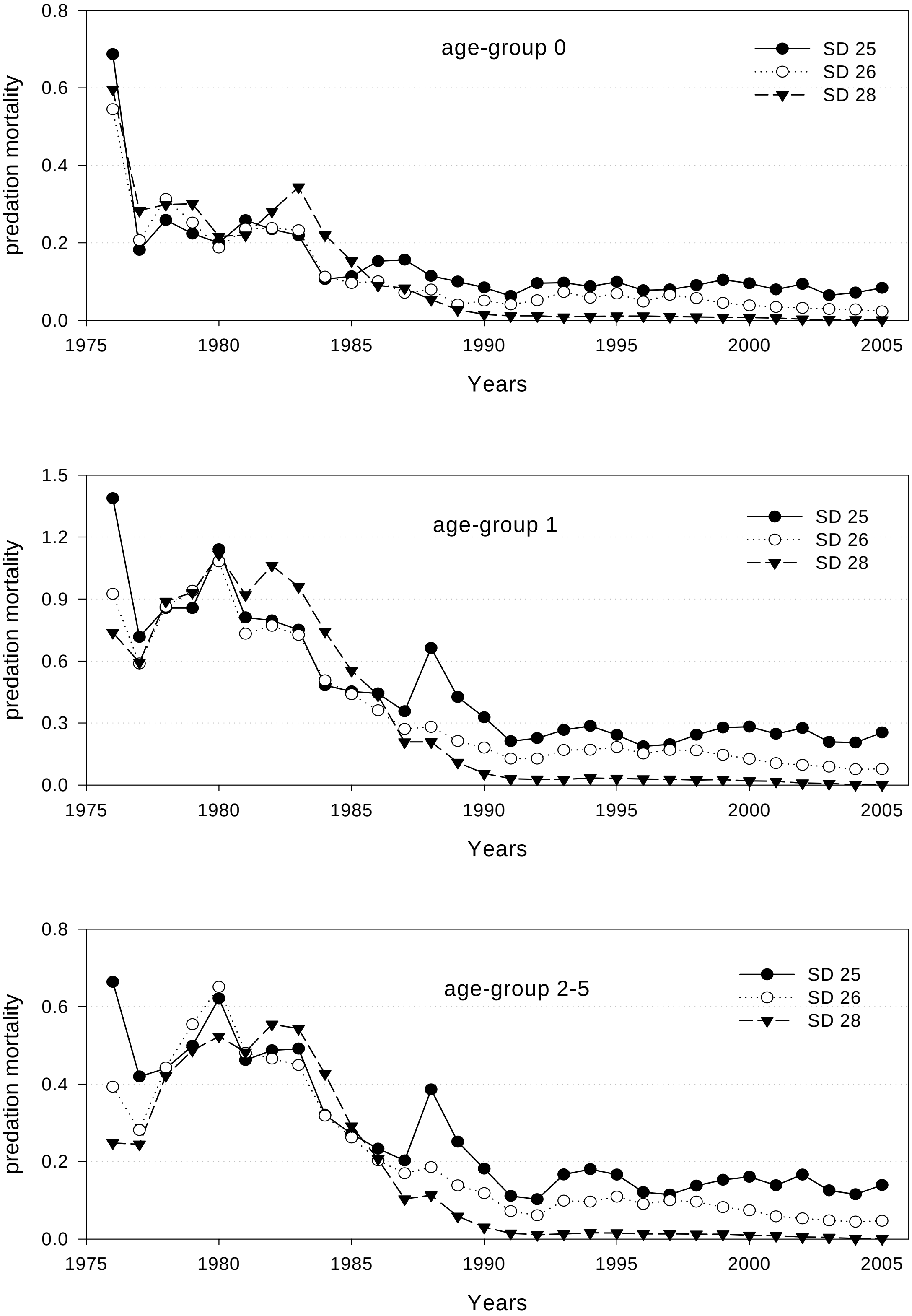

Fig. 4. Predation mortality rates of sprat age-group 0 (3rd and 4th quarter; above), age group 1 (per year; middle) and age group 2-5 (as arithmetic averages per year; below) in different Subdivisions (SD) of the Central Baltic

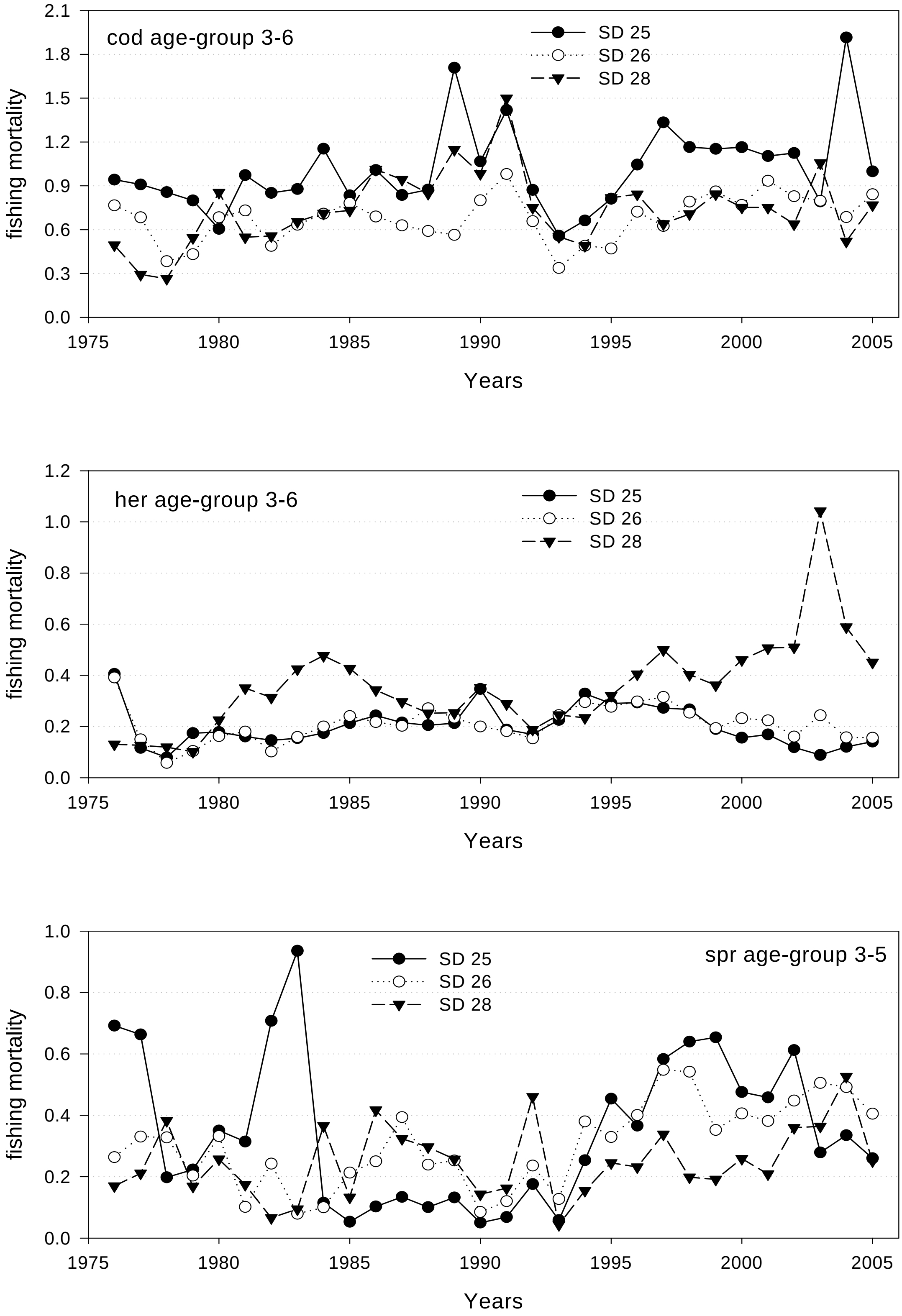

Fig. 5. Fishing mortality rates per year of cod age-groups 3-6 (as arithmetic averages; above), of herring age-groups 3-6 (as arithmetic averages; middle) and of sprat age-groups 3-5 (as arithmetic averages; below) in different Subdivisions (SD) of the Central Baltic Sea

Annex 3: Assessing the effect of Marine Protected Areas by the TEMAS model

Per J. Sparre, DIFRES

1 INTRODUCTION

To make a complete assessment of the effect of marine protected areas, which has the purpose of improving the production of Baltic cod recruits, it is required to model a long suite underlying relationships, such as.

8) The relationship between spawning stock biomass and recruitment
9) The relationship between environment and recruitment, including the impact of the environment on egg and larvae survival.
10) The temporal and spatial distribution of spawners (distribution of egg production) and juveniles, including spawning migration and migration of juvenile.
11) The relationship between fishery and recruitment, including gear selection and spatial /temporal distribution of fisheries.
12) The spatial/temporal reallocation of fishing effort after closure of an MPA, including, e.g. the impact of economy on the behaviour of fishers.
13) Predation on cod larvae and juveniles, including cannibalism.
14) Food availability for cod larvae and juveniles.

More fundamental mechanisms could be listed, but even these 7 items makes one almost give up making a complete model for the effect of MPAs. Some theories and some parameters estimation exists for all the items listed, but none are believed to be fully understood or fully documented with observations and estimations of model parameters.

The word "assessment" is used here conceptually as used by ICES working group. An assessment is composed of two parts (1) Estimation of parameters from historical data (2) Prediction based on the parameters estimated under (1). The main thing to predict is the recruitment, and needless to say to any worker with more than one year of experience in fisheries science, this is "next to impossible". What may possibly be concluded from any model on recruitment are statements like "It is believed that the regulation (e.g. an MPA) is likely to improve the future recruitment". Only the novice in fisheries science can hope to make quantitative prediction of recruitment. This is needless to say to the experienced fish stock assessment worker.

So when a model for the recruitment of Baltic cod, which can be used to assess the effect of MPAs, is presented below, there is no expectation from the side of the author, that it can ever be used for quantitative predictions. To underline this fact (which applies to any other recruitment model for any fish stock in the world), the model is formulated as a stochastic model, giving output in the form of probability distributions, rather than single figures.

The model presented here will deal with only items 1 to 3 in the list above, although the TEMAS model can handle 4 and 5, whereas TEMAS does not cover items 6 and 7, as it assumes constant natural mortality and growth rates of larvae, juveniles and adults. The handling of 4 and 5 is a multispecies, multi fleet model of fisheries including a module for fishers behaviour (in particular reaction to regulations) based on random utility model combined with a microeconomic model. However, that technical/economic module is not covered here.

The model presented below is an extract from the report on the complete model. Many aspects of the biological model are ignored. The interested reader can obtain the full report from the author.

2. THE BIOLOGICAL FRAME OF TEMAS

The biological model behind TEMAS, is the traditional model by Thompson and Bell (1934), which has been discussed in many textbooks on dynamics of fish stocks The major part of the biological model behind TEMAS is the traditional model, or generalizations of the traditional model. TEMAS extends the traditional models with a spatial model, accounting for, e.g. migration using the approach of Quinn et al, (1990). All these models originally were thought of as "fish stock assessment model", where parameters were estimated by (e.g.) VPA or "Cohort analysis" (Virtual Population Analysis, Derzhavin, 1922). In general, TEMAS has inherited all the unsolved problems of traditional fish stock assessment as implemented by ICES.

It is generally accepted that there are two separate Baltic cod stocks, the Western Stock in ICES Areas 22-24, and the Eastern stock in ICES Areas 25-32. The definition by the ICES areas, however, is rather problematic, and there is no doubt that mixing of the two stocks takes place. Needless to say the cod do not respect the borders defined by the ICES areas. The cod do not respect the sub-divisions of the Baltic as defined by ICES, which are not defined relative to the cod distribution.

With a few rare examples, the identification of the relationship between parent stock (SSB, spawning stock biomass) and subsequent recruitment (R) has remained elusive for marine fishes (Gilbert, 1997, Hilborn, 1997, Myers, 1997). The precautionary approach dictates that unless it is scientifically demonstrated that there is no relationship between the parent stock and subsequent recruitment, such a relationship should be assumed to exist, even if the data are ambiguous. Observations of stock and recruitment show large variation around any SSB/R curve, so scientists are not in a position to predict future recruitment with any accuracy. They are only able to tell the probability distribution of the future recruitment, and only then, if a long time series of SSB/R observations is available.

There is a suite of special theories on the factors that determines the recruitment of East Baltic cod. The spawning success is linked to the spatial and temporal distribution of the cod spawning. There is an extensive literature on the spawning of Baltic cod (Kuster *et al*, 2001,2004, 2006, Andersen & Mollmann, 2004). Section 7 of. the 1999 Report of the Baltic Fisheries Assessment Working Group. (ICES, 1999) summarises the knowledge basis. A more comprehensive contribution from ICES is the Report of the Study Group on Closed Spawning Areas of Eastern Baltic Cod (ICES,2004a and b). The following text is extracted from these reports. The success of recruitment is considered the key to the recovery of the Baltic cod, and the MPA's are designed to improve the success of recruitment. Therefore, special attention is given to this aspect of the cod biology.

The Bornholm Basin, the Gdansk Deep and the Gotland Basin cod are the principal spawning areas of the eastern Baltic cod stock (Figure 2.1). The salinity and oxygen conditions mainly define the spawning habitat of this stock as well as the water volume suited for egg and larval development. Salinity levels above 11 PSU are necessary to enable cod eggs to reach neutral buoyancy and an oxygen content above 2 ml/l in the water volume in which the eggs float is further required for successful egg development. These conditions define the so-called "reproductive volume", (RV), which has been shown to be positively related to the recruitment of Central Baltic cod.

The processes affecting the RV are:
i) The magnitude of inflows of saline oxygenated water from the western Baltic,
ii) Temperature regimes in the western Baltic during winter affecting the oxygen solubility prior to advection (which normally takes place during winter months),
iii) River runoff and
iv) Oxygen consumption by biological processes.

The Baltic Sea is characterised by a series of deep basins separated by shallow sills, and an inflow will usually fill up the first basin (the Bornholm Deep) only, with little or no transport in an eastern direction. Only if the inflow is very large or more likely if the advected water is replaced by an even denser water mass in a subsequent inflow or a subsequent inflow of less dense water glides over the earlier inflow water, the eastern Baltic basins will benefit from the water exchange. Thus, hydrographic monitoring and the unique topography make predictions of RV in a given year possible when conducted after the inflow period in January to March. The largest problem in the prediction is whether the inflow will turn south into the Gdansk Deep or north into the Gotland Deep, a process depending on local forcing conditions.

As a secondary effect of large inflows into the Bornholm Deep is that there is an increased likelihood of a potential inflow the following year will reach the eastern spawning areas.

The conditions for reproduction are potentially met in the Bornholm Basin deeper than 60 m, in the Gdansk Deep deeper than 80 m and in the Gotland Basin deeper than 90 m, where cod spawning takes place. However, the oxygen conditions in the eastern spawning areas are unfavourable for egg survival and development during stagnation periods. The conditions for successful egg development have been very limited in the Gotland Basin and Gdansk Deep since 1986.

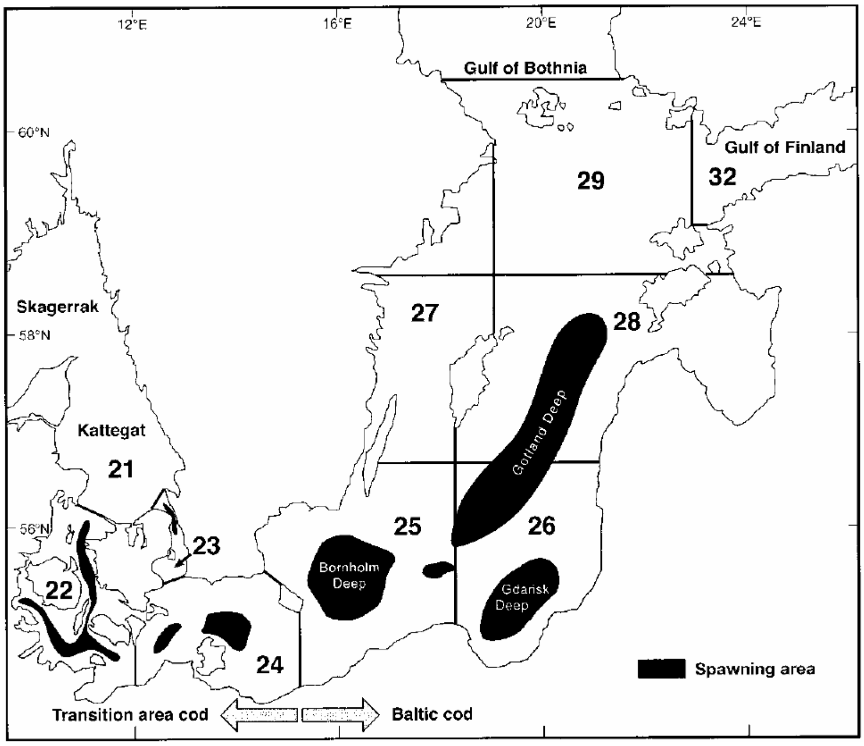

Figure 2.1. Historical spawning areas for cod in the Baltic Sea (from Bagge et al. 1994), modified by Aro (2000).

The size and distribution of the spawning stock component and thus the potential egg production in the various areas has also changed over time. The change in spawning stock distribution is evidenced by abundance indices from Baltic International Trawl Survey as well as from SSB estimates based on a spatially dis-aggregated multispecies VPA. Both show a very low spawning stock in Subdivision 28 (central Gotland Basin) at present, while the adult population components in Subdivision 25 (the Bornholm Basin) and 26 (the southern Gotland Basin and the Gdansk Deep) have remained at similar levels. A seasonal shift in the spawning stock distribution between areas seems also to occur. The proportion of the spawning stock increased in the Bornholm Basin during the spawning period while it decreased in the eastern spawning area. This pattern indicates spawning migration into the Bornholm Basin. Cod spawning migrations have previously been described from tagging experiments and from analyses of commercial catch rates with the migration intensity depending on the oxygen conditions in eastern spawning areas.

The hydrographic conditions may not only affect the horizontal distribution of cod spawning aggregation, but also the vertical distribution. Thus, lack of oxygen at the bottom can result in pelagic aggregations of spawning cod in the mid water layer just below the halocline. During the recent stagnation period pelagic aggregations of spawning cod have been abundant in all spawning areas.

The combination of decreasing egg production and low egg survival explains the low abundance of egg and larval in the Gdansk Deep and especially the Gotland Basin throughout the 1990s as well as in most recent years. As a result, the Bornholm Basin is at present the main spawning area of the eastern Baltic cod stock.

The spawning time of the eastern Baltic cod stock is very extended, i.e., from March to August – in some years extended into September. The main spawning season lasts approximately 3 months. Peak egg abundance were observed in May / early June in the 1970–80s, while a successive shift to later month was observed in the 1990s with highest egg abundance encountered from late June to late July. The timing of spawning seems to be relatively similar in the three main spawning areas. The females generally started spawning in April and continued at least into August with the majority being in spawning condition in June-July. Males reach generally spawning condition earlier and aggregate also earlier in the spawning areas than females, which means a high fishing intensity on pre-spawning aggregations of cod will result in increased male fishing mortality rates.

A special version of TEMAS applied to the Baltic cod has been developed. This Baltic cod version attempts to account for some of the basic features of the theory for Baltic cod recruitment presented above, but a full account has not been attempted. TEMAS is primarily a model that describes fisheries, it is not the hydrographical model, that would be required to match the full theory outlined above.

3. THE SPATIAL FRAME OF TEMAS

TEMAS offers the opportunity to account for spatial aspects, in the sense that fish and fleets can be allocated to a number of areas in a given time period. TEMAS uses a simple “box-model” to handle spatial aspects (Quinn II et al., 1990). This will require a number of additional input parameter, for example “migration coefficients”, the concept of which will be explained below. TEMAS is not suited for handling of a large number of areas. It is not anticipated that TEMAS applications will use more than, say, 10 divisions of the total area. TEMAS is not constructed to deal with a division of the area in small squares (say, 30 by 30 Nm, or smaller). A division of the sea area in TEMAS is relevant only when each division differs conspicuously in terms of distributions of resources and fleets. Furthermore, some knowledge (or at least some opinions) on the distributions and movements of fleets between the selected areas and stocks must be present.

For a theoretical discussion of migration in connection with age based fish stock assessment the reader is referred to Quinn II et al. (1990). These authors also discuss the estimation of migration parameters. In principle their model is the approach planned for this version of TEMAS.

The migration is modelled in a time discrete manner:

a) Migration takes place at the end of each time period and the process of migration takes zero time.
b) During a time period the fish/shrimps are assumed to be homogeneously distributed within the area.

The "Migration Coefficient", (MC), from area A to area B is defined as the fraction of the animals in area A which moves to area B. In this definition, the "movements" include the "move" from area A to area A, i.e., the event that the animal does not move. The migration coefficient depends on (or has the indices): FAr: Starting area TAr: Destination area
Note that the sum of migration coefficients over destination areas always becomes 1.0, as the starting area is also considered a destination area: $1.0 = \sum_{TAr} MC(FAr, TAr, q, a)$
where a = age group and q = time period (division of year).

The spatial set-up for the the Baltic cod used in TEMAS, will include the spawning areas of cod (for example 1: Bornholm deep, 2: Gotland deep and 3:Gdansk deep, see Figure 3.1). TEMAS will be used to simulate the migration of spawners into the MPA, as well as the migration out by juveniles and adults after spawning. For that purpose we will need 4-5 areas. Furthermore the cod resource will be divided into a western stock and an eastern stock, and mixing of the stocks will be simulated. The MPAs may be considered one area (to make calculations simpler) or it may be considered 2 or 3 separate areas. The areas shown on Figure are composed of ICES rectangles (Figure 2.1) and ICES areas, 22-32 (Figure 3.1). ICES statistical rectangles are used here because the basic data (logbook data) are by statistical rectangle.

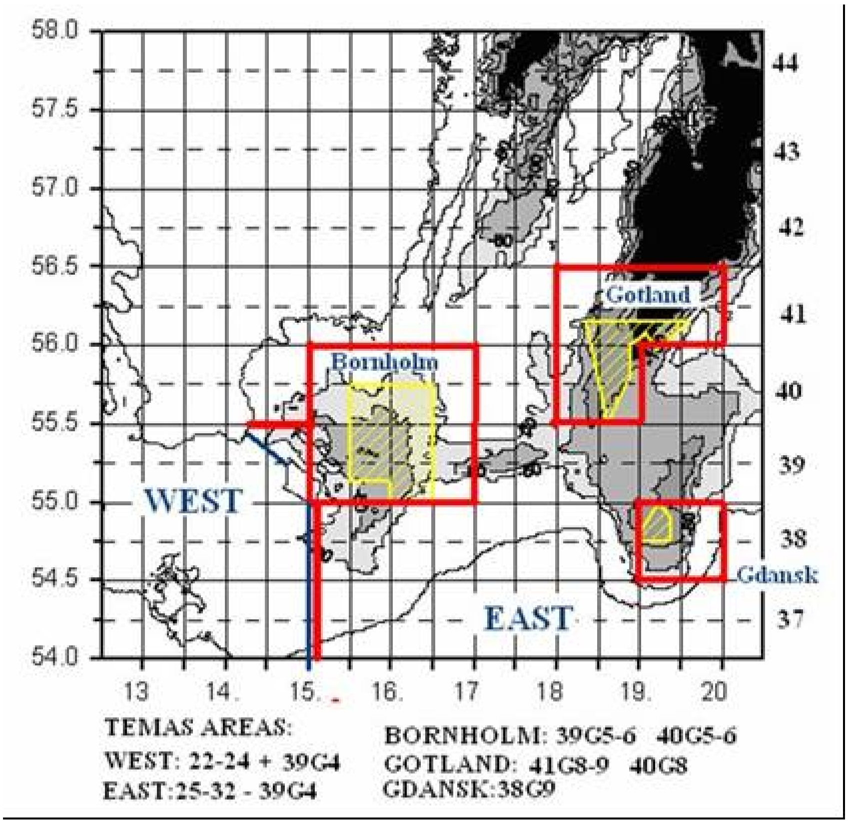

Figure 3.1. Tentative definition of Areas of the TEMAS simulation for the Baltic cod. MPA currently in force are the hatched areas.

The selection of areas is always a compromise between conflicting objectives or conditions. As mentioned above, the availability of data (by statistical rectangles) is one condition. The importance of an area in terms of landings, is another example. Figure 3.2 shows the landings of cod 1993-2003 by areas (composed of ICES rectangles) of Figure 3.1. The Gdansk area turns out to be inferior in terms of cod landings, and it should be considered it is worthwhile to include it in the simulation of the Baltic cod. Figure 3.2 shows landings in the period 1995-2003 only. Had the time series gone back to the eighties the picture would be different. In those day when the cod stock was a lot bigger than in 2003, the cod would have a wider distribution, extending into the northern areas. It is believed that currently it is only the Bornholm deep that contributes to the spawning, whereas the Gotland and Gdansk deeps also contributed substantially in the eighties.

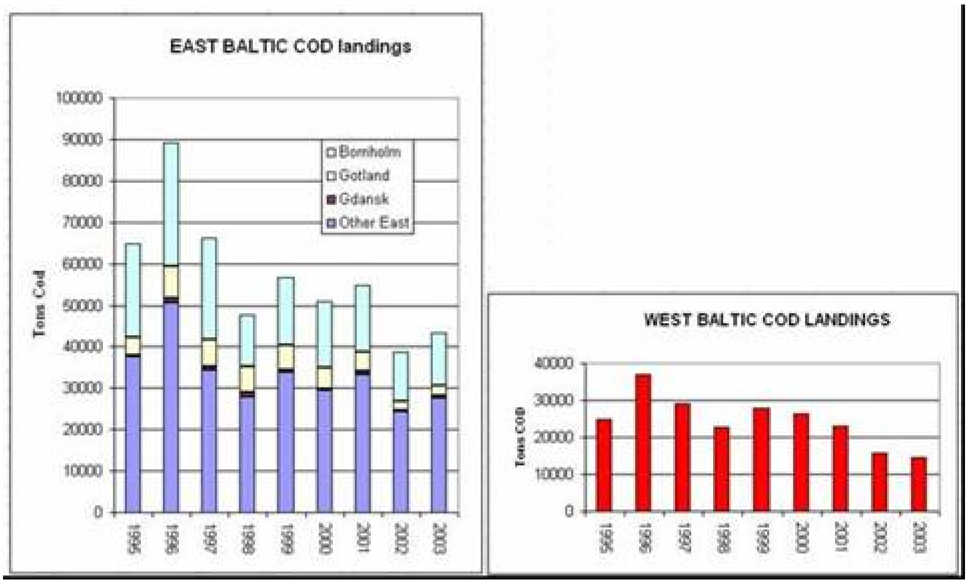

Figure .3.2. Landings of cod by areas (composed of ICES rectangles) of Figure 3.1, by Denmark, Germany, Latvia, Poland and Sweden.

4 THE MATHEMATICAL MODEL TO ASSESS THE EFFECT OF MPAs

4.1. NOTATION OF TEMAS

.The style of notation is similar to the traditional one for mathematics. Note that dot "•" instead of an index means summation over the index in question. Thus $X(i,\bullet,j) = \sum_u X(i,u,j)$. Indices in alphabetical order: The symbols used in the TEMAS biological sub-model for indices are:

Index	Explanation	Range
A	Age group	$a = 0,1,2,\ldots,a_{max}(St)$
Ar	Area	$Ar = 1,2,\ldots,Ar_{max}$
Q	Time period (as time)	$q = 1,..,q_{max}$
qa	Time period (as age)	$qa = 1,..,q_{max}$,
Y	Year	$y = y_{firSt,\ yfirst}+1,\ldots,y_{last}$
St	Stock	$St = 1,\ldots,St_{max}$

The indices "q" or "qa" stand for divisions of the year, such as "month", "quarter", "half year" etc. The time period concept may be used to indicate absolute time, and time relative to the birth of a cohort, that is the age of the cohort. The age of the cohort, however, is given in years and time periods only for the first two years of life, as the from age two and onwards, it is assumed that the difference between (year, period)-cohorts is so small that it can be ignored. This somewhat complicated age-concept is introduced to enable the model to make a fair approximation for length at age of juvenile fish. This is necessary for the analysis of gear selection aspects and recruitment, which are most important for juvenile specimens.

Time Variables in alphabetical order:

Symbol	Explanation
Dt	Basic time step (fraction of year). $dt < 1.0$. $dt = 1/q_{Max}$
y_{first} ,y_{last}	First year, Last year

4.2. GROWTH, MATURITY AND NATURAL MORTALITY

Growth of individuals are most often given in ICES WG by a year specific weight at age arrays. To simplify the model, that is, to reduce the number of parameters, TEMAS uses the von Bertalanffy growth equation, which has only three parameters. If we furthermore assume that some or all growth parameters remain constant from year to year, a considerable reduction in number of parameters has been achieved, relative to the ICES model.

Mean Body length of stock “St”, in the middle of time period q of year “y” of age group “a”, LGT(St,a,y) is given by the Von Bertalanffy equation (1934):

$$\mathrm{Lgt(St,\ y,a,q)} = \mathrm{L}_\infty(St)*\left(\ 1-\exp\left[\ -K\ \ (St)*(Age(a,q,qa)-T_0(St))\right]\ \right) \qquad (4.2.1)$$

The age of the fish (or cohort) in units of years is defined (illustrated by Table A.2.1):

$$Age(a,q,qa)=\begin{cases} a+(q-qa+0.5)*dt & \mathit{if}\ a<2 \\ a-da_{Mean}(St)+(q-0.5)*dt & \mathit{if}\ \ a\geq 2 \end{cases} \qquad (4.2.2)$$

,da_{Mean}(st) is the mean time at recruitment, which is the basis of age allocated to fish at age 2 and older. Thus, after age group 1, the influence of the birth period is assumed to be negligible. This elaborate definition of the age concept is made to accommodate the need to describe the relationship between age and length for juveniles.
Table 4.2.1 illustrates the age concept of TEMAS by showing the number of survivors by age group. In this case the year is divided into 12 months, and recruitment can occur each month. In the present example recruitment occurs only in months 3-7. In the first two years of life, each month-cohort is accounted for, but after age 2, the month-cohort are pooled into a year-cohort. For the year-cohorts, number of survivors are given for each month, as the model in this runs with a time step of one month. In the case TEMAS is used to evaluate the effect of closed seasons and MPAs, one will often want a short time step, like a month. In other uses of TEMAS the time step may be 2 month, or quarter of the year.

The body length at age can be made a stochastic variable in TEMAS, by introduction of the stochastic factor.
Body length is assumed to be the same for stock, landings and discards. This is a simplification of the model relative to ICES, which usually operates with separate weight at age keys for landings, stock and discards.
Mean Body weight is derived from the body length

$$Wgt\ \ (St,y,a,q)=QF\ \ (St,q)*Lgt(St,y,a,q)^{QE\ \ (St)} \qquad (4.2.3)$$

The condition factors, QF(St,q), is assumed to depend on the time of the year, q. That means that the user has the option to let the condition factor vary over seasons of the year. The condition exponent, QE(St), is assumed to remain constant during the year.
Also the length/weight relationship can be made stochastic in TEMAS through the stochastic factor,

Table 4.2.1. Illustration of the age concept in TEMAS. In this case the time step is one month.

Age/Time	2006 P1	2006 P2	2006 P3	2006 P4	2006 P5	2006 P6	2006 P7	2006 P8	2006 P9	2006 P10	2006 P11	2006 P12	2007 P1	2007 P2	2007 P3
Age 0 Per. 1	**0**	**0**	**115**	**216**	**348**	**235**	**47**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**665**
Age 0 Per. 2	**0**	**0**	**0**	**113**	**212**	**342**	**232**	**46**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
Age 0 Per. 3	**0**	**0**	**0**	**0**	**111**	**209**	**336**	**228**	**45**	**0**	**0**	**0**	**0**	**0**	**0**
Age 0 Per. 4	**0**	**0**	**0**	**0**	**0**	**110**	**205**	**331**	**224**	**45**	**0**	**0**	**0**	**0**	**0**
Age 0 Per. 5	**0**	**0**	**0**	**0**	**0**	**0**	**108**	**202**	**325**	**220**	**44**	**0**	**0**	**0**	**0**
Age 0 Per. 6	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**106**	**198**	**320**	**216**	**43**	**0**	**0**	**0**
Age 0 Per. 7	**35**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**104**	**195**	**314**	**212**	**42**	**0**	**0**
Age 0 Per. 8	**166**	**34**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**102**	**191**	**308**	**209**	**42**	**0**
Age 0 Per. 9	**190**	**162**	**33**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**100**	**188**	**303**	**205**	**41**
Age 0 Per. 10	**122**	**187**	**159**	**33**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**98**	**184**	**297**	**201**
Age 0 Per. 11	**58**	**119**	**183**	**156**	**32**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**96**	**180**	**291**
Age 0 Per. 12	**0**	**57**	**117**	**179**	**153**	**31**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**94**	**177**
Age 1 Per. 1	**0**	**0**	**56**	**114**	**175**	**150**	**31**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**92**
Age 1 Per. 2	**0**	**0**	**0**	**55**	**112**	**172**	**146**	**30**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
Age 1 Per. 3	**0**	**0**	**0**	**0**	**53**	**109**	**168**	**143**	**29**	**0**	**0**	**0**	**0**	**0**	**0**
Age 1 Per. 4	**0**	**0**	**0**	**0**	**0**	**52**	**107**	**164**	**140**	**29**	**0**	**0**	**0**	**0**	**0**
Age 1 Per. 5	**0**	**0**	**0**	**0**	**0**	**0**	**51**	**104**	**160**	**136**	**28**	**0**	**0**	**0**	**0**
Age 1 Per. 6	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**50**	**102**	**156**	**133**	**27**	**0**	**0**	**0**
Age 1 Per. 7	**21**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**48**	**99**	**152**	**130**	**27**	**0**	**0**
Age 1 Per. 8	**100**	**20**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**47**	**97**	**148**	**126**	**26**	**0**
Age 1 Per. 9	**115**	**97**	**20**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**46**	**94**	**144**	**123**	**25**
Age 1 Per. 10	**74**	**112**	**95**	**19**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**45**	**92**	**141**	**120**
Age 1 Per. 11	**35**	**72**	**109**	**93**	**19**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**44**	**89**	**137**
Age 1 Per. 12	**0**	**34**	**70**	**106**	**90**	**18**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**43**	**87**
Age 2 Per. 1	**293**	....	....	....	....	....	....	....	....	....	....	....	**311**	....	....

Age 2 Per. 2	……	**285**	……	……	……	……	……	……	……	……	……	……	……	**303**	……
Age 2 Per. 3	……	……	**278**	……	……	……	……	……	……	……	……	……	……	……	**295**
Age 2 Per. 4	……	……	……	**273**	……	……	……	……	……	……	……	……	……	……	……
Age 2 Per. 5	……	……	……	……	**266**	……	……	……	……	……	……	……	……	……	……
Age 2 Per. 6	……	……	……	……	……	**259**	……	……	……	……	……	……	……	……	……
Age 2 Per. 7	……	……	……	……	……	……	**253**	……	……	……	……	……	……	……	……
Age 2 Per. 8	……	……	……	……	……	……	……	**246**	……	……	……	……	……	……	……
Age 2 Per. 9	……	……	……	……	……	……	……	……	**240**	……	……	……	……	……	……
Age 2 Per. 10	……	……	……	……	……	……	……	……	……	**233**	……	……	……	……	……
Age 2 Per. 11	……	……	……	……	……	……	……	……	……	……	**227**	……	……	……	……
Age 2 Per. 12	……	……	……	……	……	……	……	……	……	……	……	**221**	……	……	……
Age 3 Per. 1	**179**	……	……	……	……	……	……	……	……	……	……	……	**216**	……	……
Age 3 Per. 2	……	**174**	……	……	……	……	……	……	……	……	……	……	……	**210**	……
Age 3 Per. 3	……	……	**169**	……	……	……	……	……	……	……	……	……	……	……	**205**
…………………………………………………………															
Age 9 Per. 10	……	……	……	……	……	……	……	……	……	**8**	……	……	……	……	……
Age 9 Per. 11	……	……	……	……	……	……	……	……	……	……	**7**	……	……	……	……
Age 9 Per. 12	……	……	……	……	……	……	……	……	……	……	……	**6**	……	……	……

The relationship between age and maturity, is modelled by the logistic curve. The maturity is usually linked to the length of the fish, so that fast growing fish will mature at a younger age than slow-growing specimens. Maturity ogive, that is the fraction of mature fish as a function of body length is

$$\mathrm{Mat(St, y, a, q)} = \frac{1}{1 + \exp(\mathrm{Mat1(St)} - \mathrm{Mat2(St)} * \mathrm{Lgt}(St, a, y, q))} \qquad (4.2.4)$$

where

Mat1(St) = ln(3)* $\mathrm{LGT}_{50\%\mathrm{Mat}}$(St)/($\mathrm{LGT}_{75\%\mathrm{Mat}}$(St) - $\mathrm{LGT}_{50\%\mathrm{Mat}}$(St)),
Mat2(St) = ln(3)/($\mathrm{LGT}_{75\%\mathrm{Mat}}$(St) - $\mathrm{LGT}_{50\%\mathrm{Mat}}$(St)) and
$\mathrm{LGT}_{\mathrm{X\%DMat}}$(St) = Length at which X % are mature.

The natural mortality is not assumed to remain constant from year to year, and depend only on stock and age group, M(St, a, y, q) = Natural mortality.

5. STOCK NUMBERS, MIGRATION AND STOCK BIOMASS.

5.1. STOCK NUMBERS AT BEGINNING OF TIME PERIOD

Stock number of stock “St”, at the beginning of time period “q” of year “y” in area “Ar” is derived as follows:

If y = y_{first} and q > 1 and a > 0	then	N(St, y_{first}, a, q,Ar) is computed by Eq. (5.3.1)
If y = y_{first} and q = 1 and a $\geq$ 0	then	N(St, $y_{firSt,}$, a, 1,Ar) is an input parameters to TEMAS
If y > y_{first} and a > 0	then	N(St, y, a, q,Ar) is computed by Eqs. (5.3.1)
If y > y_{first} and a = 0 and q = 1	then	N(St, y, a=0, q=1, Ar) = Rec(St, y, 1, Ar)
If y > y_{first} and a = 0 and q > 1	then	N(St, y, a=0, q,Ar) = Rec(St, y, q,Ar) + N(St,y,a=0,q-1,Ar) where N(St,y, a=0, q-1, Ar) is computed by Eq.(5.3.1)

The recruitment, Rec(St, y, q, Ar) of stock “St” in Area “Ar” in quarter “q” of year “y is defined by the stock/recruitment model.

5.2. Stock number at end of time period, (before migration)

Number of survivors of stock “St”, at the end of quarter “q” of year “y” in area “Ar” (before migration):

$$N1\ (St,y,a,q,Ar) = N(St,y,a,q,Ar) * \exp(-Z(St,y,a,q,Ar) * dt)$$

(5.2.1)

Where

Z(St, y, a, q, Ar) = Area specific “Total mortality” of stock “St” in area “Ar” in year “y” during quarter “q” of age group “a”.

N(St, y, a, q, Ar) = Stock number of stock “St”, at the beginning of quarter “q” of year “y” in area “Ar”

Note that the indices of N and N1 remain unchanged when considering the death process during a time period of the year. The transition between time periods is in the model dealt with “just before migration” and “just after migration”.

5.3. stock number just after migration

Number of stock “St”, at the beginning of time period “q” of year “y” in area “Ar” (just after migration).

If q = < q_{Max} then $N(St,y,a,q+1,Tar) = \sum_{Far=1}^{Ar_{Max}} MC(St,a,q,FAr,TAr) * N1(St,y,a,q,Ar)$

if q = q_{Max} (and a < a_{max}(St)) then (5.3.1)

$$N(St,y+1,a,1,TAr) = \sum_{FAr=1}^{Ar_{Max}} MC(St,a,q_{Max},FAr,TAr) * N1(St,y,a,q_{Max},Ar)$$

where

MC(St, a, q, FAr, TAr) = Migration coefficient for age group “a” of stock “St” moving from area “FAr” to area “TAr” in time period “q” .

N1(St, y, a, q, Ar) = Stock number of stock “St”, at the end of quarter “q” of year “y” in area “Ar” (before migration).

5.4. STOCK BIOMASS AND SPAWNING STOCK BIOMASS

Mean number of survivors during time period, q, is:

$$N_{Mean}(St,y,a,q,Ar) = N(St,y,a,q,Ar)\frac{1-\exp(-Z(St,y,a,q,Ar)*dt)}{Z(St,y,a,q,Ar)*dt}$$

(5.4.1)

Mean stock biomass in period q is defined as

$$B(St,y,q,Ar) = \sum_{a=0}^{a_{Max}(St)} N_{Mean}(St,y,a,q,Ar)*Wgt(St,y,a,q)$$

(5.4.2)

The conventional mean stock biomass, is the sum over areas:

$$B(St,y,q,\bullet) = \sum_{Ar=1}^{Ar_{Max}} \sum_{a=0}^{a_{Max}(St)} N_{Mean}(St,y,a,q,Ar)*Wgt(St,y,a,q)$$

The conventional spawning stock biomass in area “Ar” is:

$$SSB_{Total}(St,y,q,Ar) = \sum_{a=0}^{a_{Max}(St)} N_{Mean}(St,y,a,q,Ar)*Wgt(St,y,a,q)*Mat(St,a,q)$$

(5.4.3)

The conventional spawning stock biomass in time period q is the sum over areas $SSB(St,y,q,\bullet)$

$$SSB_{Total}(St,y,q,\bullet) = \sum_{Ar=1}^{Ar_{Max}} \sum_{a=0}^{a_{Max}(St)} N_{Mean}(St,y,a,q,Ar)*Wgt(St,y,a,q)*Mat(St,a,q)$$

(5.4.4)

The SSB concept used as input to the stock/recruitment model (to be introduced in Chapter 9) is related to the overage annual spawning stock, $SSB_{Total}(St,y,\bullet,\bullet)$, defined as:

$$SSB_{Total}(St,y,\bullet,\bullet) = \frac{1}{q_{Max}} \sum_{q=1}^{q_{Max}} \sum_{Ar=1}^{Ar_{Max}} \sum_{a=0}^{a_{Max}(St)} N_{Mean}(St,y,a,q,Ar)*Wgt(St,y,a,q)*Mat(St,a,q)$$

(5.4.5)

However, the spawning of cod is confined to a spawning season. Naturally the SSB should be the average biomass of spawners during the spawning season.

Let $RDist_{Period}(St,q)$, be the relative temporal distribution of spawning on time periods.

$$RDist_{Period}(St,q) = \frac{Egg\ production\ in\ period\ q}{Total\ Annual\ egg\ production}$$

(5.4.6)

The Western cod stock has its peak spawning in February. The eastern cod has the peak spawning in April and a longer spawning season. This leads to the definition of the “effective SSB”

$$SSB_{Effective}(St,y,\bullet,\bullet) =$$

$$\sum_{q=1}^{q_{Max}} \sum_{Ar=1}^{Ar_{Max}} \sum_{a=0}^{a_{Max}(St)} N_{Mean}(St,y,a,q,Ar)*Wgt(St,y,a,q)*Mat(St,a,q)*RDist_{Period}(St,q)$$

(5.4.7)

5.5. SPAWNING STOCK BIOMASS OF THE REPRODUCTIVE VOLUME.

The “spawning success”, is defined as a proxy for the survival of eggs into juveniles. In the Baltic, the spawning success is area specific, as it (largely) depends on the depth, which in turns determines temperature, salinity and oxygen concentration of the water that defines the “reproductive volume”.

The average spawning success factor, r_{NotMPA}, thus becomes a function of area and year. We select the area of highest spawning success, and here we name it "MPA", assuming that the marine protected area (MPA) is chosen as the one with the best spawning success. The MPA may be one or more areas, but to make things simple we assume that there is only one MPA (the Bornholm deep in the case of Baltic cod). We introduce the "Recruitment Success Factor" as

$$RSF_{MPA}(St,Ar)(= \begin{cases} rsf_{NotMPA}(St,Ar) \text{ if } Ar \neq MPA \\ 1 \text{ if } Ar = MPA \end{cases}$$

(5.5.1)

where $0 \leq rsf_{NotMPA}(St,Ar,) \leq 1$. "$rsf_{NotMPA}$" is the "reduction factor recruitment success outside the MPA". We shall come back to this concept and add some stochastic features to its definition, related to the stochastic nature of the reproductive volume.

To account for the reproductive volume and spawning success in the stock/recruitment models we introduce the concept of SSB_{RV}, the "spawning stock biomass of the reproductive volume"

$$SSB_{RV}(St,y,\bullet,\bullet) = \sum_{q=1}^{q_{Max}} \sum_{Ar=1}^{Ar_{Max}} \sum_{a=0}^{a_{Max}(St)} N_{Mean}(St,y,a,q,Ar) *$$

$$Wgt(St,y,a,q) * Mat(St,a,q) * RDist_{Period}(St,q) * RSF_{MPA}(St,Ar)$$

(5.5.2)

This concept makes the SSB_{RV} depend on the timing of recruitment, $RDist_{Period}$ and the migration of spawning cod, MC(St,a,q,Far,Tar). If the cod migrate to the MPA during the spawning season the SSB_{RV}, gets bigger than if they remained outside the MPA. Ignoring the special case of $q = q_{Max}$, the expression for SSB_{RV} is

$$SSB_{RV}(St,y,\bullet,\bullet) = \sum_{q=1}^{q_{Max}} \sum_{Ar=1}^{Ar_{Max}} \sum_{a=0}^{a_{Max}(St)} \sum_{FAr=1}^{Ar_{Max}} MC(St,a,q-1,FAr,TAr) * N1(St,y,a,q-1,Ar) *$$

$$\frac{1-\exp(-Z(St,y,a,q-1,Ar)*dt)}{Z(St,y,a,q-1,Ar)*dt} * Wgt(St,y,a,q) * Mat(St,a,q) * RDist_{Period}(St,q) * RSF_{MPA}(St,Ar)$$

(5.5.3)

Tables 5.5.1.a and b show two hypothetical applications of Eq 5.5.3. To make this hypothetical example simple, Z is assumed to be zero, so the number of survivors remain constant in all time periods, q=1,2,3,4. The factor $\frac{1-\exp(-Z(St,y,a,q-1,Ar)*dt)}{Z(St,y,a,q-1,Ar)*dt}$ in Eq. 5.5.3 is given the value 1.0. Furthermore only three age groups are considered, and only three areas (Bornholm Basin, BB, Gotland Basin, GB and the remaining part of Eastern Baltic, EB). In table a, the migration coefficients are chosen so that all spawners move to Bornholm Basin, (the area of highest spawning success) during the entire spawning season (q=1,2). The calculated values of SSB_{RV} are shown in right hand side of the last column. Table b uses the same parameters, except for the migration coefficients.

In table b, not all spawners go to the Bornholm Basin, and since the spawning success outside the Bornholm Basin is lower, the resulting value of SSB_{RV} becomes lower.

If furthermore, Z had been assigned a positive value, with a smaller value in the Bornholm Basin due to area closure (smaller fishing mortality), the effect of spawning migration would become even larger than that of Table a.

To explain Tables 5.5.1.a+b, consider the calculations of numbers in age group 2 in area BB in first and second quarter of year 2 in Table b. These number are 109.6 and 114.7 (underlined)

To achieve these results, the calculations are:

$$MC(a=1,q=1,EB,BB) * N(a=1,y=1,q=4,EB) + MC(a=1,q=1,BB,BB) * N(a=1,y=1,q=4,BB) + MC(a=1,q=1,GB,BB) * N(a=1,y=1,q=4,GB) = N(a=2,y=2,q=1,BB)$$

$$0.6*146 + 1.0*22 + 0.0*2 = 109.6$$

$$MC(a=2,q=1,EB,BB) * N(a=2,y=2,q=4,EB) + MC(a=2,q=1,BB,BB) * N(a=2,y=2,q=4,BB) + MC(a=2,q=1,GB,BB) * N(a=2,y=2,q=4,GB) = N(a=2,y=2,q=2,BB)$$

$$0.55*29.2+0.9*109.6+0*36.2=114.7$$

Table 5.5.1.a. Hypothetical example illustrating the impact of migration on SSB_{RV} (Spawning stock biomass of the reproductive volume). In this case all spawners go to the Bornholm Basin (BB) during spawning. Note the high values of SSB_{RV} (q1)=76.4 and of SSB_{RV} (q2)=114.6. compared to the values in Table b.

From East Baltic (EB)

	q=1	MC(a,1,1,TAr)		q=2	MC(a,2,1,TAr)		q=3	MC(a,3,1,TAr)		q=4	MC(a,4,1,TAr)	
to	EB	BB	GB	EB	BB	GB	EB	BB	GB	EB	BB	GB
Age 2	0.00	1.00	0.00	0.00	1.00	0.00	1.00	0.00	0.00	1.00	0.00	0.00
Age 3	0.00	1.00	0.00	0.00	1.00	0.00	1.00	0.00	0.00	1.00	0.00	0.00
Age 4	0.00	1.00	0.00	0.00	1.00	0.00	1.00	0.00	0.00	1.00	0.00	0.00

From Bornholm Basin (BB)

	q=1	MC(a,1,2,TAr)		q=2	MC(a,2,2,TAr)		q=3	MC(a,3,2,TAr)		q=4	MC(a,4,2,TAr)	
to	EB	BB	GB	EB	BB	GB	EB	BB	GB	EB	BB	GB
Age 2	0.00	1.00	0.00	0.00	1.00	0.00	0.75	0.25	0.00	0.75	0.25	0.00
Age 3	0.00	1.00	0.00	0.00	1.00	0.00	0.75	0.25	0.00	0.75	0.25	0.00
Age 4	0.00	1.00	0.00	0.00	1.00	0.00	0.75	0.25	0.00	0.75	0.25	0.00

From Gotland Basin (GB)

	q=1	MC(a,1,3,TAr)		q=2	MC(a,2,3,TAr)		q=3	MC(a,3,3,TAr)		q=4	MC(a,4,3,TAr)	
to	EB	BB	GB	EB	BB	GB	EB	BB	GB	EB	BB	GB
Age 2	0.00	1.00	0.00	0.00	1.00	0.00	0.25	0.00	0.75	0.25	0.00	0.75
Age 3	0.00	1.00	0.00	0.00	1.00	0.00	0.25	0.00	0.75	0.25	0.00	0.75
Age 4	0.00	1.00	0.00	0.00	1.00	0.00	0.25	0.00	0.75	0.25	0.00	0.75

					Stock numbers, N(y,q)				
Wgt	Mat	RSF	Area		Y=1, q=4 a-1	Y=2, q=1 a	Y=2, q=2 a	Y=2, q=3 a	Y=2, q=4 a
1.00	0.20	0.10	EB	Age 2	146.0	0.0	0.0	131.3	164.1
1.50	0.65	0.10		Age 3	95.0	0.0	0.0	84.8	105.9
2.00	1.00	0.10		Age 4	60.0	0.0	0.0	53.3	66.6
			EB	Tot	301.	0.0	0.0	269.	336.

	Rdist.$_{Period}$(q)			
	0.40	0.60	0.00	0.00
Wgt*Mat*RSF	Y=2, q=1	Y=2, q=2	Y=2, q=3	Y=2, q=4
	Wgt*Mat*RSF*N			
0.02	0.0	0.0	0.0	0.0
0.098	0.0	0.0	0.0	0.0
0.2	0.0	0.0	0.0	0.0
Total	0.0	0.0	0.0	0.0

			al		0			3	6
1.00	0.50	0.50	BB	Age 2	22.0	175.0	175.0	43.8	10.9
1.50	0.90	0.50		Age 3	15.0	113.0	113.0	28.3	7.1
2.00	1.00	0.50		Age 4	9.0	71.0	71.0	17.8	4.4
			BB	Total	46.0	359.0	359.0	89.8	22.4
1.00	0.50	0.10	GB	Age 2	7.0	0.0	0.0	0.0	0.0
1.50	0.90	0.10		Age 3	3.0	0.0	0.0	0.0	0.0
2.00	1.00	0.10		Age 4	2.0	0.0	0.0	0.0	0.0
			GB	Total	12.0	0.0	0.0	0.0	0.0
			Grand Total		359.0	359.0	359.0	359.0	359.0

0.25	17.5	26.3	0.0	0.0
0.675	30.5	45.8	0.0	0.0
1	28.4	42.6	0.0	0.0
Total	76.4	114.6	0.0	0.0
0.05	0.0	0.0	0.0	0.0
0.135	0.0	0.0	0.0	0.0
0.2	0.0	0.0	0.0	0.0
Total	0.0	0.0	0.0	0.0
SSB_{RV}	76.4	114.6	0.0	0.0

The numbers needed in the calculations are indicated by italic font. The remaining calculation is a simple multiplication of five numbers, for example:

$N_{Mean}(y=2,a=2,q=2,Ar=BB)*Wgt(y,a,q)*RSF_{MPA}(BB)*Mat(a,q)*RDist_{Period}(q)$
$114.7 * 1.0 * 0.5 * 0.5 * 0.6 = 17.2$

Hereby, a model is designed that (in theory) can show a relationship between an area closure and the $SSB_{RV.}$ This model, however, does not account for the reaction of fishers, in terms of reallocation of fishing effort, after the introduction of an MPA. TEMAS contains a sub-model, the so-called RUM (Random utility model), that accounts for the behaviour of fishers.

Table 5.5.1.b. Hypothetical example illustrating the impact of migration on SSB_{RV} (Spawning stock biomass of the reproductive volume). In this case not all spawners go to the Bornholm Basin (BB) during spawning. Note the low values of SSB_{RV} (q1)=55.4 and of SSB_{RV} (q2)=83.6. compared to the values in Table a.

From East Baltic (EB)

	q=1	MC(a,1,1,TAr)		q=2	MC(a,2,1,TAr)		q=3	MC(a,3,1,TAr)		q=4	MC(a,4,1,TAr)	
To	EB	BB	GB	EB	BB	GB	EB	BB	GB	EB	BB	GB
Age 2	0.20	*0.60*	0.20	0.30	*0.55*	0.15	1.00	0.00	0.00	1.00	0.00	0.00
Age 3	0.15	0.65	0.20	0.20	0.60	0.20	1.00	0.00	0.00	1.00	0.00	0.00
Age 4	0.10	0.65	0.25	0.15	0.65	0.20	1.00	0.00	0.00	1.00	0.00	0.00

From Bornholm Basin (BB)

	q=1	MC(a,1,2,TAr)		q=2	MC(a,2,2,TAr)		q=3	MC(a,3,2,TAr)		q=4	MC(a,4,2,TAr)	
To	EB	BB	GB	EB	BB	GB	EB	BB	GB	EB	BB	GB
Age 2	0.00	*1.00*	0.00	0.10	*0.90*	0.00	0.50	0.50	0.00	0.60	0.40	0.00
Age 3	0.00	1.00	0.00	0.10	0.90	0.00	0.50	0.50	0.00	0.60	0.40	0.00
Age 4	0.00	1.00	0.00	0.10	0.90	0.00	0.50	0.50	0.00	0.60	0.40	0.00

From Gotland Basin (GB)

	q=1	MC(a,1,3,TAr)		q=2	MC(a,2,3,TAr)		q=3	MC(a,3,3,TAr)		q=4	MC(a,4,3,TAr)	
To	EB	BB	GB	EB	BB	GB	EB	BB	GB	EB	BB	GB
Age 2	0.00	*0.00*	1.00	0.10	*0.00*	0.90	0.50	0.00	0.50	0.70	0.00	0.30
Age 3	0.00	0.00	1.00	0.10	0.00	0.90	0.55	0.00	0.45	0.75	0.00	0.25
Age 4	0.00	0.00	1.00	0.10	0.00	0.90	0.60	0.00	0.40	0.80	0.00	0.20

Wgt	Mat	RSF	Area		Stock numbers, N(y,q)				
					Y=1, q=4, ' a-1	Y=2, q=1 a	Y=2, q=2 a	Y=2, q=3 a	Y=2, q=4 a
1.00	0.20	0.10	EB	Age 2	*146.0*	*29.2*	23.3	99.2	146.5
1.50	0.65	0.10		Age 3	95.0	14.3	12.7	64.0	94.9
2.0	1.0	0.1		Age	60.0	6.0	7.4	40.9	60.3

	$\text{Rdist.}_{Period}(q)$			
	0.40	*0.60*	0.00	0.00
Wgt* Mat* RSF	Y=2, q=1	Y=2, q=2	Y=2, q=3	Y=2, q=4
	Wgt*Mat*RSF*N			
0.02	0.2	0.3	0.0	0.0
0.098	0.6	0.7	0.0	0.0
0.2	0.5	0.9	0.0	0.0

0	0	0		4					
			EB	Total	301.0	49.5	43.5	204.0	301.7
1.00	*0.50*	*0.50*	BB	Age 2	*22.0*	*109.6*	*114.7*	57.4	22.9
1.50	0.90	0.50		Age 3	15.0	76.8	77.6	38.8	15.5
2.00	1.00	0.50		Age 4	9.0	48.0	47.1	23.6	9.4
			BB	Total	46.0	234.4	239.4	119.7	47.9
1.00	0.50	0.10	GB	Age 2	*7.0*	*36.2*	37.0	18.5	5.5
1.50	0.90	0.10		Age 3	3.0	22.0	22.7	10.2	2.5
2.00	1.00	0.10		Age 4	2.0	17.0	16.5	6.6	1.3
			GB	Total	12.0	75.2	76.1	35.3	9.4
			Grand Total		359.0	359.0	359.0	359.0	359.0

Total	1.3	1.9	0.0	0.0
0.25	11.0	17.2	0.0	0.0
0.675	20.7	31.4	0.0	0.0
1	19.2	28.3	0.0	0.0
Total	50.9	76.9	0.0	0.0
0.05	0.7	1.1	0.0	0.0
0.135	1.2	1.8	0.0	0.0
0.2	1.4	2.0	0.0	0.0
Total	3.3	4.9	0.0	0.0
SSB_{RV}	55.4	83.7	0.0	0.0

6. STOCK AND RECRUITMENT MODELS

6.1. FOUR STOCK RECRUITMENT MODELS

There are four options for stock and recruitment model in TEMAS: (1) Beverton and Holt model (Beverton & Holt, 1957) (2) "Hockey stick" model (Barrowman & Meyers, 1999), (3) Ricker Model (Ricker, 1954) (4) the general Deriso-Schnute Model (Deriso 1980, Schnute, 1985). Without going into details, we show the characteristic shapes of the four models in Figure 6.1.1. The deterministic recruitment model in TEMAS is a function of spawning stock, SSB_{RV}, only. Dependence of environmental factors, can be accounted for by "stochastic factors" that determines the reproductive volume. The four standard S/R-models are extended to account for spatial and temporal variation in recruitment for Baltic cod.

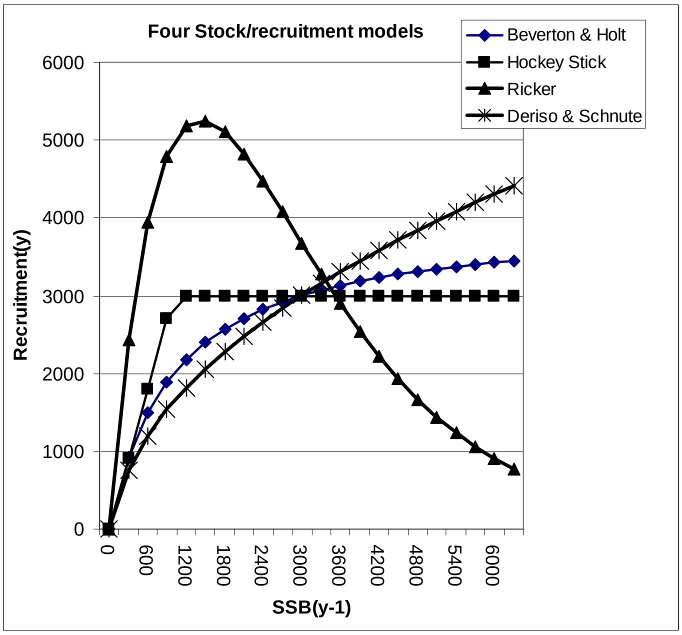

Figure 6.1.1. Examples of the four alternative Stock-Recruitment models available in TEMAS, SR_1: Beverton & Holt, SR_2: Hockey-stick, SR_3: Ricker, SR_4:Deriso-Schnute.

6.1. DETERMINISTIC STOCK RECRUITMENT MODEL

The deterministic recruitment is in TEMAS derived from the “average annual stock SSB of the reproductive volume” of last year. The general stock/and recruitment model used in TEMAS for predicting recruitment becomes

$$Rec(St,y,\bullet,\bullet) = STR_X(SSB_{RV}(St,y-1,\bullet,\bullet))$$

(6.1.1)

where suffix “x” can take the values 1,2,3,4 according the the choice of S/R model. (1) Beverton & Holt (2) “Hockey stick” (3) Ricker (4) Deriso-Schnute (Figure 6.0.1)

6.2. TEMPORAL AND SPATIAL RECRUITMENT MODEL

After the total stock recruitment is derived, it is subsequently distributed on areas and time periods by the input parameters, $RecDist_{Area}(St,Ar)$ and $RecDist_{Period}(St,q)$, the relative distribution of recruitment on areas and time periods as will be discussed below.

$$\mathrm{Re}c(St,y,q,Ar) = \mathrm{Re}cDist_{Area}(St,Ar) * \mathrm{Re}cDist_{Period}(St,q) * STR_X(SSB_{RV}(St,y-1,\bullet,\bullet))$$

(6.2.1)

$$\mathrm{Re}cDist_{Area}(St,Ar) = \frac{\mathrm{Re}cruitment\ \ number\ in\ \ area\ \ "ar"}{Total\ \ \mathrm{Re}cruitment\ \ Number} = \frac{N(St,y,0,q,Ar)}{\sum_{i=1}^{Ar_{Max}} N(St,y,0,q,i)}$$

(6.2.2)

Thus, $RecDist_{Area}(St, Ar)$ is assumed to be independent of time period, “q”. The distribution on time periods is defined the same way, $RecDist_{Period}(St, q)$ is assumed to be independent of area, “Ar”. A hypothetical example of area and period distributions is shown in Table 6.2.1. The recruitment is distributed on all areas and periods in Table a, whereas Table b concentrates all spawning in area 3 in period 2.

Table 6.2.1a. Hypothetical example of $RecDist_{Area}$, $RecDist_{Perd}$ *and* $RecDist_{Area}$*$RecDist_{Period}$

		RecDist$_{Period}$				
		Period 1	Period 2	Period 3	Period 4	Total
	RecDist$_{Area}$	0.333	0.556	0.111	0.000	1.000
Area 1	0.136	0.045	0.076	0.015	0.000	0.136
Area 2	0.682	0.227	0.379	0.076	0.000	0.682
Area 3	0.136	0.045	0.076	0.015	0.000	0.136
Area 4	0.045	0.015	0.025	0.005	0.000	0.045
Total	1.000	0.333	0.556	0.111	0.000	1.000

Table 6.2.1b. Hypothetical example of $RecDist_{Area}$, $RecDist_{Period}$ *and* $RecDist_{Area}$*$RecDist_{Period}$

		RecDist$_{Period}$				
		Period 1	Period 2	Period 3	Period 4	Total
	RecDist$_{Area}$	0.000	1.000	0.000	0.000	1.000
Area 1	0.000	0.000	0.000	0.000	0.000	0.000
Area 2	0.000	0.000	0.000	0.000	0.000	0.000
Area 3	1.000	0.000	1.000	0.000	0.000	1.000
Area 4	0.000	0.000	0.000	0.000	0.000	0.000
Total	1.000	0.000	1.000	0.000	0.000	1.000

The recruitment model now reads

$$\mathrm{Re}c(St,y,q,Ar)=\mathrm{Re}cDist_{Area}(St,ar)*\mathrm{Re}cDist_{Period}(St,q)*STR_X(SSB_{RV}(St,y-1,\bullet,\bullet))$$

(6.2.3)

6.3. STOCHASTIC MODEL OF RECRUITMENT

TEMAS has the option to let recruitment becomes a stochastic variable, through the stochastic factor $\varepsilon_{SR}(St)$,

$$Rec(St,y,\bullet,\bullet)=STR_X(SSB_{RV}(St,y-1,\bullet,\bullet))*\varepsilon_{SR}(St)$$

(6.3.1)

Where $\varepsilon_{SR}(St) = \varepsilon_{SR1}(St) * R_{RepVol}(St)$ (6.3.2)

is the product of two stochastic factors of stock/recruitment relationship, of stock "St". The factor $\varepsilon_{SR1}(St)$ is a stock dependent log-normally distributed stochastic variable with mean value 1.0 and standard deviation σ_{SR} .

The factor $R_{RepVol}(St)$, the "reproductive volume factor", is specially designed to accommodate the dynamics of Baltic cod, where the recruitment is believed to be enhanced by large reproductive volumes. The reproductive volume becomes big, when the inflow of salty water from the North Sea is big. This happens only in certain years, and $\varepsilon_{SR2}(St)$ is a uniformly distributed stochastic variable controlling a reproductive volume factor, $R_{RepVol}(St)$

$$R_{\mathrm{Re}\,pVol}(St) = \begin{cases} R_{RV}(St) \;\; if \;\; \varepsilon_{SR2}(st) \le 1/N_{\mathrm{Re}\,pVol}(St) \\ 1 \;\; if \;\; \varepsilon_{SR2}(St) > 1/N_{\mathrm{Re}\,pVol}(St) \end{cases} \qquad (6.3.3)$$

Where $N_{RepVol}(St)$ is the average number of years between occurrences of large reproductive volumes. $R_{RV}(St)$ is the average relative magnitude of recruitment in years of high reproductive volume.

Estimation of the average number of years between occurrences of large reproductive volumes, $N_{\mathrm{Re}\,pVol}(St)$ for Baltic cod, is illustrated in Table 6.3.1 and Figure 6.3.1. The data the data used to produce Figure 6.0.1. The definition of “normal years” and “outstanding years” is subjective, and is based on visual splitting of the recruitment frequencies into two lognormal distributions. The result is that every sixth year is outstanding (is an “inflow year”) for western Baltic cod, whereas every seventh year is outstanding for eastern Baltic cod, whereas. This is indeed a rather crude way of estimating $N_{\mathrm{Re}\,pVol}(St)$, but is probably the best we can do for the time being.

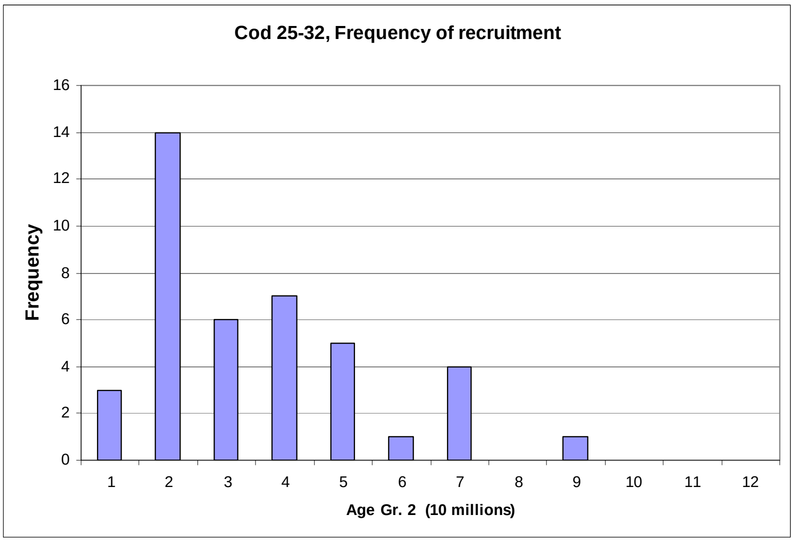

Figure 6.3.1. Recruitment frequency (age group 2) of Eastern Baltic cod. There are 40 observations covering the period from 1966 to 2005 (Source ICES, WGBFAR, 2006)

Lower*)	Upper*)	Index	Frequency
Normal 0	100	1	3
Normal 100	200	2	14
Normal 200	300	3	6
Normal 300	400	4	7
Normal 400	500	5	5
		Total	35
Out St. 500	600	6	1
Out St. 600	700	7	4
Out St. 700	800	8	0
Out St. 800	900	9	1
		Total	6
		Grand total	41

$$N_{\mathrm{Re}\,pVol}(2) = 41/6 = 6.8$$

$$N_{\mathrm{Re}\,pVol}(1) = 36/6 = 6$$

Table 6.3.1. Estimation of $N_{\mathrm{Re}\,pVol}(St)$ *(average number of years between occurrences of large reproductive volumes) for Eastern Baltic cod based on the data in Figures 6.3.1, (Source: ICES WGBFAR, 2006).*

TEMAS can draw numbers from a random number generator. TEMAS makes a long suite of simulations, where parameter values are drawn from the random number generator each time. Figure

6.3.2 Shows a hypothetical example of simulated recruitments with the model described above. Making 1000 or more simulations of the entire system over, say, 10 years, provides enough results to produce frequency diagrams of simulation results. That makes it possible to present results not as single figures (e.g.. the SSB 20 years from now), but the probability distribution of it.

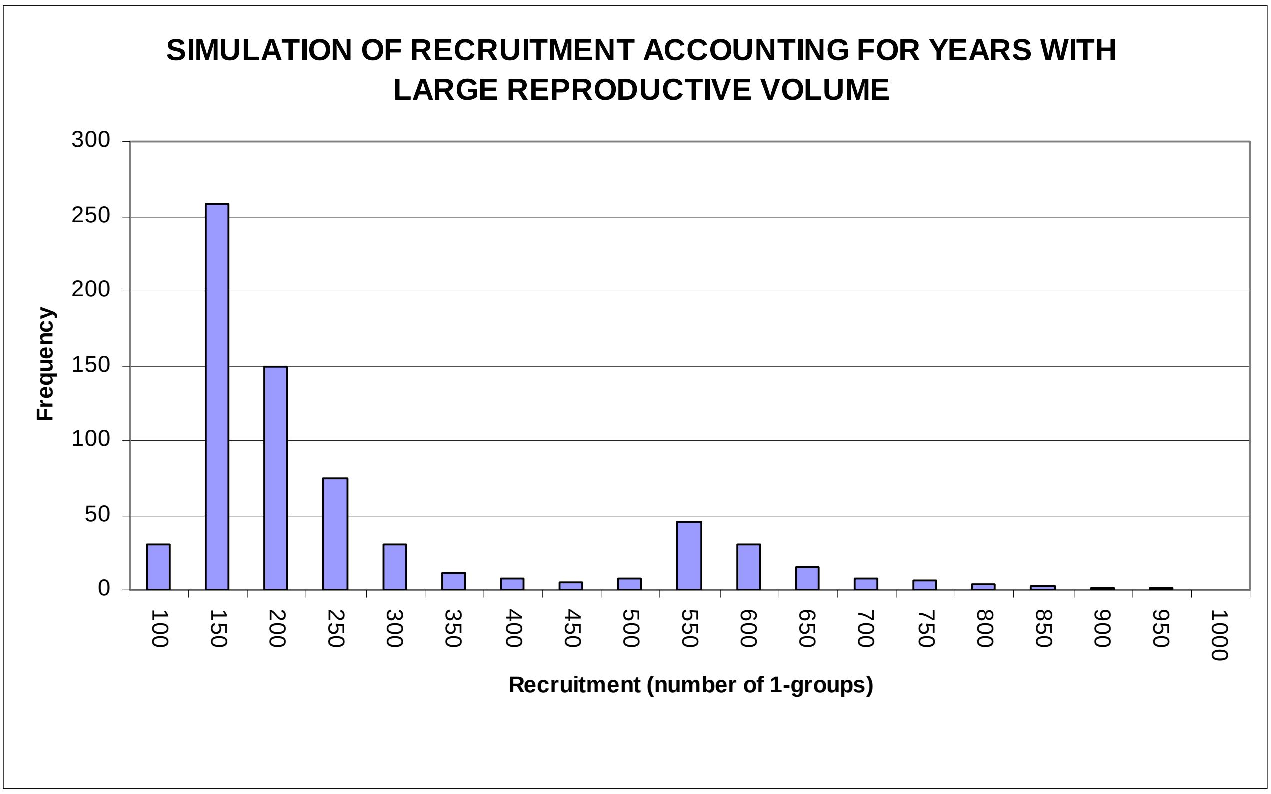

Figure 6.3.2. Hypothetical example of simulated recruitments with the model used for Baltic cod.

When the reproduction volume is high, the size of the spawning grounds becomes larger, or in other words, the spawning success increases in all areas (Kuster *et al*, 2001,2004, 2006). Therefore the spawning success becomes a function of, $\varepsilon_{SR2}(St)$, the uniformly distributed stochastic variable, that determines the years of outstandingly high reproductive volume. The spawning success factor, r_{NotMPA}, becomes a function of $\varepsilon_{SR2}(St)$.

$$RSF_{MPA}(St, Ar, \varepsilon_{SR2}(St))(= \begin{cases} rsf_{NotMPA}(St, Ar, \varepsilon_{SR2}(St)) \;\; if \;\;\; Ar \neq MPA \\ 1 \;\;\; if \;\;\; Ar = MPA \end{cases}$$

where $0 \leq rsf_{NotMPA}(St, Ar, \varepsilon_{SR2}(St)) \leq 1$

The spawning success factor outside the MPA (or the "outside MPA reduction factor"), rsf_{NotMPA} is defined similarly to $R_{\mathrm{Re}\,pVol}(St)$, (reproductive volume factor)

$$rsf_{NotMPA}(St, Ar, \varepsilon_{SR2}(St)) = \begin{cases} rsf^{High}_{NotMPA}(St, Ar) \;\; if \;\; \varepsilon_{SR2}(St) \leq 1/N_{\mathrm{Re}\,pVol}(St) \\ rsf^{Low}_{NotMPA}(St, Ar) \;\; if \;\; \varepsilon_{SR2}(St) > 1/N_{\mathrm{Re}\,pVol}(St) \end{cases}$$

(6.3.4)

where $0 \;\leq\; rsf^{Low}_{NotMPA}(St, Ar, \varepsilon_{SR2}(St)) \;\leq\; rsf^{High}_{NotMPA}(St, Ar, \varepsilon_{SR2}(St)) \leq \; 1$

The distribution on areas will also change when the reproductive volume is high, so that there will bee two distributions depending on high or low reproductive volume, $\mathrm{Re}\,cDist_{Area}^{High}(St, Ar)$ .and $\mathrm{Re}\,cDist_{Area}^{Low}(St, Ar)$ respectively. This gives the model for distribution of total biomass on areas and periods:

$$\mathrm{Re}\,c(St, y, q, Ar) =$$

$$\begin{cases} \mathrm{Re}\,c(St, y, \bullet, \bullet) * \mathrm{Re}\,cDist_{Area}^{High}(St, Ar) * \mathrm{Re}\,cDist_{Period}(St, q) \;\; if\; \varepsilon_{SR2}(st) \le 1/N_{\mathrm{Re}\,pVol}(St) \\ \mathrm{Re}\,c(St, y, \bullet, \bullet)) * \mathrm{Re}\,cDist_{Area}^{Low}(St, Ar) * \mathrm{Re}\,cDist_{Period}(St, q) \;\; if\; \varepsilon_{SR2}(st) > 1/N_{\mathrm{Re}\,pVol}(St) \end{cases} \quad (6$$

.3.5)

6.4. THE COMPLETE RECRUITMENT MODEL OF TEMAS

Eventually, we arrive at the S/R model for Baltic cod, which allow for more spawning success in the MPA, and account for years of outstanding reproductive volume.

$$\mathrm{Re}\,c(St, y, \bullet, \bullet) =$$

$$\begin{cases} STR_X(SSB_{VP}(St, y-1, \bullet, \bullet)) * \varepsilon_{SR1}(St) * R_{RV}(st) \;\; if\; \varepsilon_{SR2}(st) \le 1/N_{\mathrm{Re}\,pVol}(St) \\ STR_X(SSB_{VP}(St, y-1, \bullet, \bullet)) * \varepsilon_{SR1}(St) \qquad\qquad if\; \varepsilon_{SR2}(st) > 1/N_{\mathrm{Re}\,pVol}(St) \end{cases}$$

(6.4.1)

where $R_{RV}(St) > 1$ accounts for “outstanding recruitments” or “inflow years”

$$SSB_{RV}(St, y, \bullet, \bullet) = \sum_{q=1}^{q_{Max}} \sum_{Ar=1}^{Ar_{Max}} \sum_{a=0}^{a_{Max}(St)} N_{Mean}(St, y, a, q, Ar) * \quad (6.4$$

$$Wgt(St, y, a, q) * Mat(St, a, q) * RDist_{Period}(St, q) * RSF_{MPA}(St, Ar)$$

$$.2)\; RSF_{MPA}(St, Ar, \varepsilon_{SR2}(St)) = \begin{cases} rsf_{NotMPA}(St, Ar, \varepsilon_{SR2}(St)) \;\; if \;\; Ar \ne MPA \\ 1 \;\; if \;\; Ar = MPA \end{cases}$$

(6.4.3)

where the “Spawning success factor” is defined as

$$rsf_{NotMPA}(St, Ar, \varepsilon_{SR2}(St)) = \begin{cases} rsf_{NotMPA}^{High}(St, Ar, \varepsilon_{SR2}(St)) \;\; if\; \varepsilon_{SR2}(St) \le 1/N_{\mathrm{Re}\,pVol}(St) \\ rsf_{NotMPA}^{Low}(St, Ar, \varepsilon_{SR2}(St)) \;\; if\; \varepsilon_{SR2}(St) > 1/N_{\mathrm{Re}\,pVol}(St) \end{cases} \quad (6.$$

4.4)

where $0 \;\le\; rsf_{NotMPA}^{Low}(St, Ar, \varepsilon_{SR2}(St)) \;\le\; rsf_{NotMPA}^{High}(St, Ar, \varepsilon_{SR2}(St)) \le\; 1$

$$\mathrm{Re}\,c(St, y, q, Ar) =$$

$$\begin{cases} \mathrm{Re}\,c(St, y, \bullet, \bullet) * \mathrm{Re}\,cDist_{Area}^{High}(St, Ar) * \mathrm{Re}\,cDist_{Period}(St, q) \;\; if\; \varepsilon_{SR2}(st) \le 1/N_{\mathrm{Re}\,pVol}(St) \\ \mathrm{Re}\,c(St, y, \bullet, \bullet)) * \mathrm{Re}\,cDist_{Area}^{Low}(St, Ar) * \mathrm{Re}\,cDist_{Period}(St, q) \;\; if\; \varepsilon_{SR2}(st) > 1/N_{\mathrm{Re}\,pVol}(St) \end{cases} \quad (6.$$

4.5)

Tables 6.4.1-2. Illustrate of the stock recruitment model of TEMAS, by a numerical (hypothetical) example. Tables 6.4.1.a-e contain the input values to the model, and Table 6.4.2 shows the results.

The input parameters to the Baltic stock/recruitment model are

a *Spawning success factors* rsf^{High}_{NotMPA} and rsf^{Low}_{NotMPA}

b *Input: Temporal distribution:* $\mathrm{Re}\,cDist_{Period}(St,q)$

c *Reproductive Volume and B & H S/R Parameters,* R_{RV}, N_{RepVol}, STR_{11}And STR_{12}

d *High and low distribution on areas* $\mathrm{Re}\,cDist^{Low}_{Period}(St,Ar)$ and $\mathrm{Re}\,cDist^{Low}_{Period}(St,Ar)$

e *Input: Stochastic factors and SSB of six years* $\varepsilon_{SR1}(St,y)$, $\varepsilon_{SR2}(St,y)$ and $SSB_{Total}(St,y,\bullet,Ar)$

(y = 2005,…,2010) for the four areas “West”, “East” “Bornholm” and “Gotland”

and to these parameters should be added the usual stock parameters, including the migration coefficients.

	Area	
rsf^{High}_{NotMPA}	West	*0.8*
	East	*0.8*
	MPA=Bornholm	*1*
	Gotland	*1*
rsf^{Low}_{NotMPA}	West	*0.2*
	East	*0.2*
	Bornholm	*0.8*
	Gotland	*0.1*

Table 6.4.1.a. Input: Spawning success factor

$\mathrm{Re}\,cDist_{Period}(St,q)$

Per1	*0.25*
Per2	*0.70*
Per3	*0.05*
Per4	*0.00*

Table 6.4.1.b. Input: Temporal distribution

R_{RV}	*2.0*
N_{RepVol}	*5.0*
$1/N_{RepVol}$	*0.2*
STR_{11}	*2.0*
STR_{12}	*0.0001*

Table 6.4.1.c. Input: Reproductive volume and Beverton & Holt S/R Parameters

	$\mathrm{Re}\,cDist^{Low}_{Period}(St,Ar$	$\mathrm{Re}\,cDist^{High}_{Period}(St,Ar)$.
West	*0.05*	*0.05*
East	*0.05*	*0.06*
MPA=Bornholm	*0.65*	*0.43*
Gotland	*0.25*	*0.46*

Table 6.4.1.d. Input: High and low distribution on areas

Stochastic factors						
Year	2005	2006	2007	2008	2009	2010
$\varepsilon_{SR1}(St,y)$	*1.130*	*0.955*	*0.995*	*1.006*	*0.987*	*1.001*
$\varepsilon_{SR2}(St,y)$	*0.110*	*0.769*	*0.148*	*0.662*	*0.644*	*0.959*
Outstand. Year	Yes	No	Yes	No	No	No
$SSB_{Total}(St,y,\bullet,Ar)$						
West	*100*	*110*	*115*	*131*	*121*	*108*
East	*300*	*320*	*334*	*370*	*364*	*313*
MPA=Bornholm	*100*	*110*	*124*	*132*	*118*	*103*

Gotland	*100*	*105*	*111*	*121*	*139*	*112*
TOTAL	*600*	*645*	*684*	*754*	*742*	*636*

Table 6.4.1.e. Input: Stochastic factors and SSB of six years (hypothetical example)

To explain some of the caluculation in Table 6.4.2, consider the Beverton and Holt stock recruitment model in 2005:

Deterministic B&H: 1884.1 = 2.0*1040/(1+0.0001*1040)

(Deterministic B&H)* $\varepsilon_{SR1}(St,y)$ = 1884.1*1.130 = 2128.7

Note that 2005 is an inflow year (and outstanding year for the reproductive volume). Therefore the spatial distribution is made by the “High Reproductive Volume” distribution.

	Rec * $\mathrm{Re}\,cDist^{High}_{Period}(St, Ar)$.
West	0.05*2128.7 = 106.4
East	0.06*2128.7 = 127.7
MPA=Bornholm	0.43*2128.7 = 915.3
Gotland	0.46*2128.7 = 979.2

West	Rec* $\mathrm{Re}\,cDist^{High}_{Period}(St, Ar = West)$ * $\mathrm{Re}\,cDist_{Period}(St,q)$
Per1	0.25*106.4 = 26.61
Per2	0.70*106.4 = 74.50
Per3	0.05*106.4 = 5.32
Per4	0.00*106.4 = 0

Eventually recruits are distributed on time periods as shown in the right hand side of the text table.

Table 6.4.2. Illustration of the stock recruitment model of TEMAS, by a hypothetical example with input from Table 6.4.1.

	Year	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**
$SSB_{RV}(St,y,\bullet,\bullet)=\sum_{Ar=1}^{Ar_{Max}} SSB(St,y,\bullet,Ar)*RSF_{MPA}(St,Ar)$ $RSF_{MPA}(St,Ar,\varepsilon_{SR2}(St))=\begin{cases} rsf_{NotMPA}(St,Ar,\varepsilon_{SR2}(St)) \text{ if } Ar \neq MPA \\ 1 \text{ if } Ar = MPA \end{cases}$							
High Reproductive Volume $SSB * rsf^{High}_{NotMPA}$	West	100*08 =80	88	92	104.8	96.8	86.4
	East	300*0.8=240	256	267.2	296	291.2	250.4
	MPA=Bornholm	100*1.0=100	110	124	132	118	103
	Gotland	100*1.0=100	105	111	121	139	112
	TOTAL	**520**	**559**	**594.2**	**653.8**	**645**	**551.8**
	TOTAL*R_{RV}= TOTAL*2	**1040**	**1118**	**1188.4**	**1307.6**	**1290**	**1103.6**
Low Reproductive Volume $SSB * rsf^{Low}_{NotMPA}\cdot$	West	100*02=20	22	23	26.2	24.2	21.6
	East	300*02=60	64	66.8	74	72.8	62.6
	MPA=Bornholm	100*02=80	88	99.2	105.6	94.4	82.4
	Gotland	100*01=10	10.5	11.1	12.1	13.9	11.2
	TOTAL	**170**	**184.5**	**200.1**	**217.9**	**205.3**	**177.8**
	Outstanding year	Yes	No	Yes	No	No	No
Low Rep.. Vol.	**SSB_{RV}**	**NA**	**184.5**	**NA**	**217.9**	**205.3**	**177.8**
High Rep.. Vol.	**SSB_{RV}**	**1040.0**	**NA**	**1188.4**	**NA**	**NA**	**177.8**

	Year	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**
Low Rep.. Vol.	Deterministic BH model	NA	362.3	NA	426.5	402.3	349.4
High Rep.. Vol.	Deterministic BH model	1884.1	NA	2124.3	NA	NA	NA
Low Rep.. Vol.	BH * $\varepsilon_{SR1}(St,y)$	NA	346.1	NA	429.1	397.0	349.8
High Rep.. Vol.	BH* $\varepsilon_{SR1}(St,y)$	2128.7	NA	2113.0	NA	NA	NA

				Recruitment distributed on areas			
	Outstanding year	Yes	No	Yes	No	No	No
	Year	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**
Low Rep.. Vol.	West	NA	18.12	NA	21.33	20.12	17.47
	East	NA	18.12	NA	21.33	20.12	17.47
	MPA=Bornholm	NA	235.50	NA	277.23	261.52	227.10
	Gotland	NA	90.58	NA	106.63	100.58	87.35
High Rep.. Vol.	West	106.43	NA	105.65	NA	NA	NA
	East	127.72	NA	126.78	NA	NA	NA
	MPA=Bornholm	915.34	NA	908.60	NA	NA	NA
	Gotland	979.20	NA	971.99	NA	NA	NA

			Recruitment distributed on areas and periods				
	Outstanding year	Yes	No	Yes	No	No	No
Time period	**Year**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**
Q1	West	26.61	4.53	26.41	5.33	5.03	4.37
Q1	East	74.50	12.68	73.96	14.93	14.08	12.23
Q1	MPA=Bornholm	5.32	0.91	5.28	1.07	1.01	0.87
Q1	Gotland	0.00	0.00	0.00	0.00	0.00	0.00
Q2	West	31.93	4.53	31.70	5.33	5.03	4.37

Q2	East	89.41	12.68	88.75	14.93	14.08	12.23
Q2	MPA=Bornholm	6.39	0.91	6.34	1.07	1.01	0.87
Q2	Gotland	0.00	0.00	0.00	0.00	0.00	0.00
Q3	West	228.83	58.88	227.15	69.31	65.38	56.78
Q3	East	640.74	164.85	636.02	194.06	183.06	158.97
Q3	MPA=Bornholm	45.77	11.78	45.43	13.86	13.08	11.36
Q3	Gotland	0.00	0.00	0.00	0.00	0.00	0.00
Q4	West	244.80	22.64	243.00	26.66	25.15	21.84
Q4	East	685.44	63.41	680.39	74.64	70.41	61.14
Q4	MPA=Bornholm	48.96	4.53	48.60	5.33	5.03	4.37
Q4	Gotland	0.00	0.00	0.00	0.00	0.00	0.00
	TOTAL	2128.69	362.32	2113.02	426.51	402.34	349.39

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Annex 4: Description of the bioeconomic model BEMCOM

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A description of the bioeconomic model utilised in relation FP6 PROTECT relation to the EU 6th Framework Programme project *Marine Protected Areas as a Tool for Ecosystem Conservation and Fisheries Management* (PROTECT) is the topic of this annex.

The acronym of the bioeconomic model is BEMCOM (**B**io**E**conomic **M**odel to evaluate the **CO**nsequences of **M**arine protected areas), and it is programmed in GAMS (General Algebraic Modeling System).

BEMCOM is programmed in a generic way in order to be able to handle different case studies reflected through the utilised dataset and parameter values.

The description will enfold as follows. Section A.1 presents the general model dimensions, variables and parameters. An overview of how the variables interact in the model is given in section A.2, while specific model equations are specified in section A.3. Section A.4 focus on the objectives and restriction included in BEMCOM, and section A.5 gives some general comments to the model, including expansion possibilities. Finally, section A.6 specifies the dimensions and parameters for the case study related to the sand eel fishery in the North Sea primarily conducted by Danish fishermen.

A.1 Dimensions, variables and parameters

The bioeconomic model BEMCOM has in the general approach six dimensions in order to reflect different relevant aspects of the fishery. The dimensions are as follows:

- time	t =	1,…,T
- vessel/fleet segment	f =	1,…,F
- home port	h =	1,…,H
- area	a =	1,…,A
- species	s =	1,…,S
- cohort	c =	1,…,C

When describing the array of variables in BEMCOM, a deduction can be made between variables related to economic, biological and production issues.

The economic variables in BEMCOM are:

- profit	PROFIT
- total costs	TCOST
- variable costs	VCOST
- fuel and lubricants costs	FUEL
- provision costs	PROVISIONS
- ice costs	ICE
- sales costs	SALES
- crew costs	CREW
- fixed costs	FCOST
- maintenance costs	MAIN
- insurance costs	INSUR
- other fixed costs	OTHER

- investments	INVEST
- fish prices	PRICES

The biological variables are:

- fish stocks	STOCK
- recruitment	RECRU
- natural mortality	NAMOR

While the production variables are:

- catches	CATCH
- effort	EFFORT
- fleet size	FLEET
- change in fleet size	CFLEET
- landings	LANDING
- weighted landings	WLANDING
- discards	DISCARD

All variables are determined within the model framework, when the objective function is optimised.

It is also for the model parameters relevant to distinguish between those related to economic, biological and production issues, respectively.

The economic parameters in BEMCOM are:

- fuel price	fp
- provisions price	pp
- ice price	ip
- sales share	ss
- crew share	cs
- investment share	is

while the biological parameters are:

- catchability coefficient	cc
- recruitment coefficient	rc
- natural mortality coefficient	nmc
- discard coefficient	dc
- weight coefficient	wc

And finally the production parameters are:

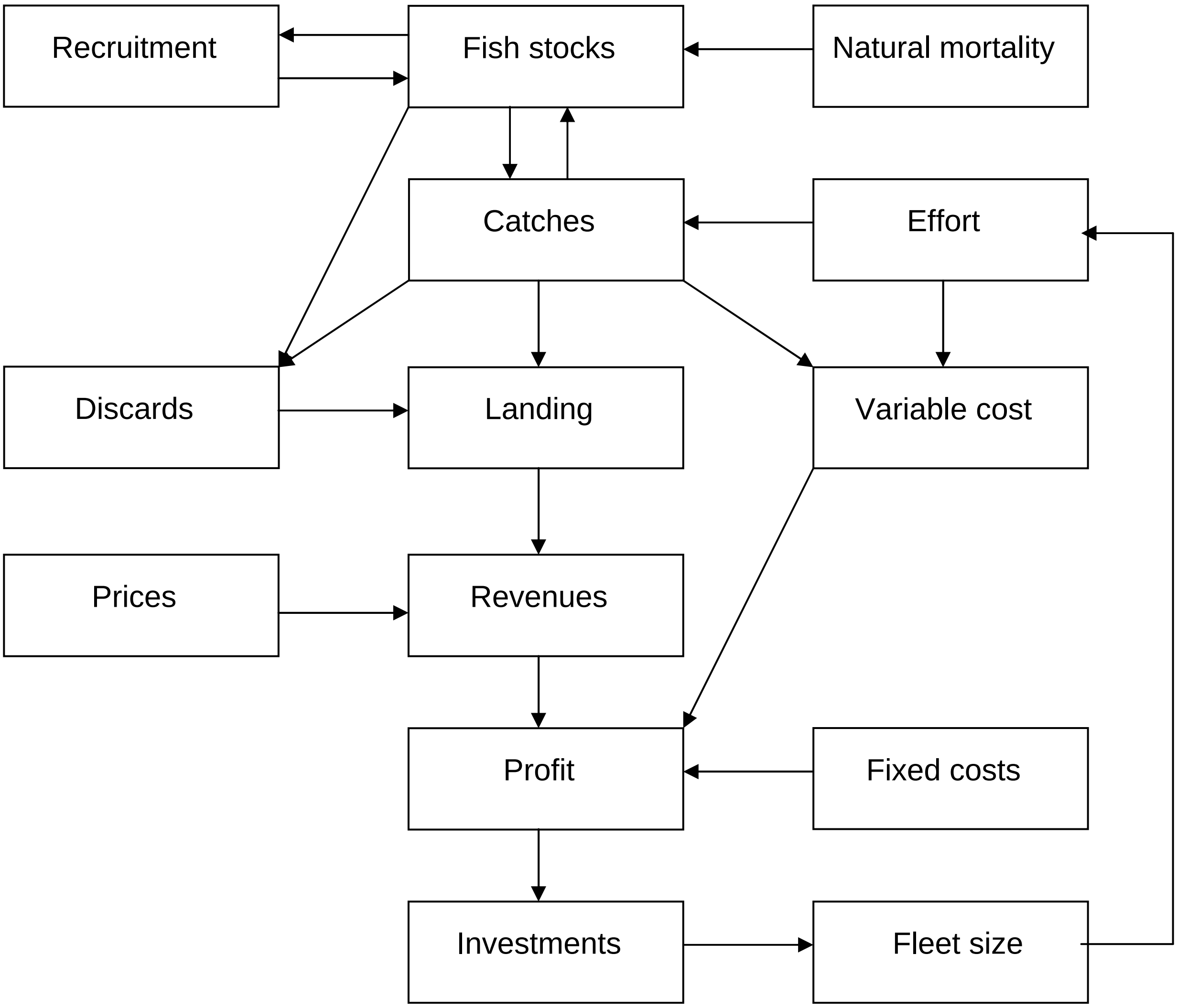
- fuel consumption	fc
- ice consumption	ic

The values of the parameters are determined outside the model either on the basis of the utilised dataset or by using knowledge about these.

The level of detail for the dimensions, variables and parameters can of course be reduced or expanded, if this is desired. The choice is dependent on several things including the available information.

A.2 Model overview

The figure below gives a simple overview of the interactions in BEMCOM. The model is more complex than shown, but it gives the basic flows.



These interactions can of course be discussed, but they concur with the general understanding within fisheries economics.

A.3 Equations

Turning attention towards the actual model equations, things becomes more complicated. A deduction is made between biological related equations, equations for the short term economic development of the fishing vessels, and long term equations for the development in the fishing fleet.

A.3.1 Biological equations

Starting with the biological equations, the number of fish recruited to cohort 1 of species *s* in fishing area *a* at time *t* depends on the size of the spawning stock biomass and the recruitment coefficient *rc*:

$$(3.1)\ \mathrm{RECRU}_{t,a,s,1} = (\mathrm{STOCK}_{t,a,s,3} + \mathrm{STOCK}_{t,a,s,4} + \ldots + \mathrm{STOCK}_{t,a,s,C-1} + \mathrm{STOCK}_{t,a,s,C}) \times \exp(rc^{obs}_{a,s})$$

The fish stocks can be reduced either through natural mortality or through fishermen catching the fish.

The number of fish in cohort *c* dying from natural causes of species *s* in fishing area *a* at time *t* depends on the stock size of the specific cohort and its natural mortality coefficient.

$$(3.2)\ \mathrm{NAMOR}_{t,a,s,c} = \mathrm{STOCK}_{t,a,s,c} \times \exp(-\mathrm{nmc}^{obs}_{a,s,c})$$

The catches are contrary to recruitment and natural mortality dependent on the type of vessels (selectivity, primary gear type etc.) and the home port of the vessels (fishermens age, experience etc.). The average catch in numbers at time *t* for a vessel in fleet *f* with home port *h* of cohort *c* of species *s* in fishing area *a* is determined by the catchability coefficient, the effort of the vessel and the stock size:

$$(3.3)\ \mathrm{CATCH}_{t,f,h,a,s,c} = \mathrm{cc}^{obs}_{f,h,a,s,c} \times \mathrm{EFFORT}_{t,f,h,a} \times \mathrm{STOCK}_{t,a,s,c}$$

The production relationship reflected in the catch equation follows the simple classic Schaefer approach, but other types of approaches could also be utilised, for instance a more complex Cobb-Douglas function than the one used above or a translog function.

With the above equations in mind, the development in stock size of cohort *c* of species *s* in fishing area *a* from time *t-1* to time *t* is as follows:

(3.4)

$$\mathrm{STOCK}_{t,a,s,c} = \mathrm{STOCK}_{t-1,a,s,c} + \mathrm{RECRU}_{t,a,s,c} - \mathrm{NAMOR}_{t,a,s,c} - \sum_{f=1}^{F} \sum_{h=1}^{H} \mathrm{CATCH}_{t,f,h,a,s,c} \times \mathrm{FLEET}_{t,f,h}$$

Unwanted catches are an almost unavoidable part of fisheries, and it is therefore necessary to divide the catches between a discarded and landed part.

The discarded part can for instance be of no value to the fishermen or illegal to land, and therefore thrown back into the sea. Unfortunately the survival rate of the discarded fish is very low, and these are therefore not included in the equation describing the development of the stock. The number of fish discarded at time *t* from the average vessel in fleet *f* with home port *h* in area *a* of cohort *c* of species *s* is:

$$(3.5)\ \mathrm{DISCARD}_{t,f,h,a,s,c} = \mathrm{dc}^{obs}_{f,h,a,s,c} \times \mathrm{CATCH}_{t,f,h,a,s,c}$$

The rest of the catch is therefore landed. At time *t*, an average vessel in fleet *f* with home port *h* thus land the following number of fish caught in area *a* of cohort *c* of species *s*:

$$(3.6)\ \mathrm{LANDING}_{t,f,h,a,s,c} = \mathrm{CATCH}_{t,f,h,a,s,c} - \mathrm{DISCARD}_{t,f,h,a,s,c}$$

By multiplying the landings in numbers with the weight coefficient, the landed weight is obtained:

$$(3.7)\ \mathrm{WLANDING}_{t,f,h,a,s,c} = \mathrm{wc}^{obs}_{a,s,c} \times \mathrm{LANDING}_{t,f,h,a,s,c}$$

A.3.2 Short term economic equations

The economic equations can be divided between revenue and cost, where the latter can further be separated into variable and fixed costs. The economic variables are characterised by not having are, species and cohorts as their dimensions, but they are of course determined by variables and parameters including these dimensions.

The revenue obtained from the landed catch is obtained by multiplying this with the fish price. The average revenue at time *t* for a vessel in fleet *f* with home port *h* is:

$$(3.8)\ \mathrm{REVENUE}_{t,f,h} = \sum_{a=1}^{A}\sum_{s=1}^{S}\sum_{c=1}^{C} \mathrm{WLANDING}_{t,f,h,a,s,c} \times \mathrm{PRICE}_{f,a,s,c}^{obs}$$

Landing prices for each species and cohort are not dependent on the total amount of fish landed. Prices on fish are generally considered to be determined on the global market. A more correct modelling approach would therefore be to consider total global landings, but this is a time consuming and cumbersome process, which has therefore been excluded from the model.

As mentioned, the average total costs at time *t* for a vessel in fleet *f* with home port *h* can be of either variable costs, which varies with the short term activity of the vessel, or fixed costs which varies with the long term activity of the vessel:

$$(3.9)\ \mathrm{TCOST}_{t,f,h} = \mathrm{VCOST}_{t,f,h} + \mathrm{FCOST}_{t,f,h}$$

Looking first at the variables costs, these consists of costs related to the use of fuel, provisions and ice, auction related costs and finally payment to the crew onboard:

$$(3.10)\ \mathrm{VCOST}_{t,f,h} = \mathrm{FUEL}_{t,f,h} + \mathrm{PROVISIONS}_{t,f,h} + \mathrm{ICE}_{t,f,h} + \mathrm{SALES}_{t,f,h} + \mathrm{CREW}_{t,f,h}$$

The fuel costs at time *t* for an average vessel in fleet *f* with home port *h* is determined by the fuel price fp, the fuel consumption fc and finally the effort of the vessel:

$$(3.11)\ \mathrm{FUEL}_{t,f,h} = \sum_{a=1}^{A} \mathrm{fp}^{obs} \times \mathrm{fc}_{f,h,a}^{obs} \times \mathrm{EFFORT}_{t,f,h,a}$$

The fuel costs are determined such that these varies with the choice of area to fish in. If the area is close to the vessels home port, fuel consumption is low and vice versa. No division is made between steaming time and fishing time.

It is necessary to bring supplies on every trip in order to feed the crew. The amount of supplies is related to the number of days at sea. The provisions costs at time *t* for an average vessel in fleet *f* with home port *h* is therefore determined as:

$$(3.12)\ \mathrm{PROVISIONS}_{t,f,h} = \sum_{a=1}^{A} \mathrm{pp}_{f}^{obs} \times \mathrm{EFFORT}_{t,f,h,a}$$

Observe that the provisions price vary between fleets, because the average number of crew members differ.

When catching fish, it is necessary to store the cold in order to preserve the fish. This requires ice, and the costs of ice at time *t* for an average vessel in fleet *f* with home port *h* is thus determined as:

$$(3.13)\ \mathrm{ICE}_{t,f,h} = \sum_{a=1}^{A}\sum_{s=1}^{S}\sum_{c=1}^{C} \mathrm{ip}^{obs} \times \mathrm{ic}_{f,a}^{obs} \times \mathrm{LANDING}_{t,f,h,a,s,c}$$

The ice costs are thus assumed dependent on the landed amount of fish, but is also influenced by the type of fish landed through the ice consumption parameter ic. This is due to the fact that industrial species requires lesser ice than consumption species.

When selling the landing, auctions usually requires a fee in order to cover costs for the auctioneer, packing and transporting. The fee is often a share of the revenue, and the sales cost at time *t* for an average vessel in fleet *f* with home port *h* is therefore:

(3.14) $\mathrm{SALES}_{t,f,h} = \sum_{a=1}^{A}\sum_{s=1}^{S}\sum_{c=1}^{C} \mathrm{ss}_{f}^{obs} \times \mathrm{REVENUE}_{t,f,h,a,s,c}$

The final variable cost is payment of the crew. Normally, this is considered to be a share of the revenue deducted some specific types of variable costs. However, for easiness the crew payment at time *t* for an average vessel in fleet *f* with home port *h* is calculated as:

(3.15) $\mathrm{CREW}_{t,f,h} = \mathrm{cs}_{f}^{obs} \times \mathrm{REVENUE}_{t,f,h}$

With the basic equations for the variable costs in place, it is time to consider the fixed costs. The fixed costs is considered to be composed of three elements: 1) maintenance costs, 2) insurance costs, and 3) other costs (rent of buildings on share, accountancy assistance etc.). Thus, the fixed costs at time *t* for an average vessel in fleet *f* with home port *h* is:

(3.16) $\mathrm{FCOST}_{t,f,h} = \mathrm{MAIN}_{t,f,h} + \mathrm{INSUR}_{t,f,h} + \mathrm{OTHER}_{t,f,h}$

None of these costs varies with the daily activity of the individual vessel, nor the areas and types of species caught, and is therefore determined through the values from the utilised data.

Having defined how revenue and cost is determined, the profit at time *t* for an average vessel in fleet *f* with home port *h* is defined as:

(3.17) $\mathrm{PROFIT}_{t,f,h} = \mathrm{REVENUE}_{t,f,h} - \mathrm{TCOST}_{t,f,h}$

A.3.3 Long term economic equations

The long term economic equations are related to the development in the fishing fleet, and thus the investment level among the vessel owners. Describing the investment environment is a complex case, where many factors should be considered, including the fisherman's individual desires towards investing in the fishery, the surrounding financial environment, and regulatory restrictions limiting the desired behaviour.

Investments is often considered to result in an expansion of the fishing fleet, however it can also result in the opposite, if the investments are negative. It is often assumed that the incentive for negative investments are more sluggish than the incentive for positive investments, and this can of course be included in the investment equations.

However, as a starting point it is simply assumed that the investments are determined by an investment share times the profit level. Thus, the investment at time *t* for an average vessel in fleet *f* with home port *h* is given as follows:

(3.18) $\mathrm{INVEST}_{t,f,h} = \mathrm{ir}_{f,h}^{obs} \times \mathrm{PROFIT}_{t,f,h}$

Investments in the fleet will result in changes in the fleet structure towards the most profitable fleet types, which is influenced by their catch and cost composition. It is therefore necessary to transform the investment level into changes in the fleet size. How do we approach this?

(3.19) …

Having transformed the investments into the number of actual vessels, the development in the fleet can at time *t* for fleet *f* with home port *h* be calculated as:

(3.20) $FLEET_{t,f,h} = FLEET_{t-1,f,h} + CFLEET_{t,f,h}$

A.4 Objectives and restrictions

Besides the relationships given in the equations presented in section 3, the model further includes an objective to be optimised and restrictions to be kept.

Within the economic literature, it is generally assumed that fishermen and society as well seek to obtain the highest profit. Thus, the objective function will be a summarization of all the individual vessels profit levels for all time periods and homeports:

(4.1) maximise TOTPROFIT = $\sum_{t=1}^{T}\sum_{f=1}^{F}\sum_{h=1}^{H} PROFIT_{t,f,h}$

Society may also have other objectives such as the highest possible employment, and this could justify the use of a multi-objective approach. This will however not be pursued further here.

Obtaining the highest profit for the society does not necessarily need to be done without including other considerations. This can be done through imposing restrictions on the endogenous variables in the model.

There are of course the fundamental restrictions, which for instance secure that catches and effort are not negative. Contrary to these restrictions are the

An obvious example of a restriction is the overall catch restriction given through the quotas, i.e.:

(4.2) $\sum_{f=1}^{F}\sum_{h=1}^{H}\sum_{c=1}^{C} LANDING_{t,f,h,a,s,c} \leq QUOTA_{t,a,s}$

Not all species are regulated by quotas, or alternatively the quotas a significantly higher than the possible catch amounts, and other measures must therefore be used for these species instead. This could for example be a fraction of the fish stock or a number corresponding with the catches in previous years.

Another restriction could relate to the effort of the vessels. Natural or observed limitations on the effort level can for instance be given as minimum or maximum values:

(4.3) $EFFORT_{t,f,h,a} \leq (\geq)\ EFFORT_{t,f,h,a}^{min(max)}$

Other types of restrictions could of course also be included in BEMCOM.

A.5 General comments on the model

Several general comments can be made to the above equations, besides the ones already mentioned. These are:

- Despite that BEMCOM is a dynamic model, there is no inclusion of inflation in the equations. However, it is relative straightforward to include such considerations by applying a parameter in each relevant equation, which accounts for this.

- Developments in the fish prices are as mentioned determined on the global market. A possibility to account for this to some extent is by including available price flexibilities.

- Because the catchability rate has cohorts as one dimension, it is possible to include the selectivity of different gears types in the model, thus facilitating evaluation of changes in gear selection.

- If the time horizon is long, it could be relevant to include a discount rate, and thus perform maximisation of the “net present profit”.

Annex 5: Hydroodynamic backtracking of fish larvae by individual- based modelling

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Hydrodynamic backtracking of fish larvae by individual-based modelling

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Abstract

We discuss methodological and implementation issues of spatial, temporal and combined spatio-temporal backtracking and illustrate larval backtracking for North Sea lesser sandeel *Ammodytes Marinus* larvae, using a combined hydrodynamical and individual-based model. It is found that dispersal effects are important for larval backtracking predictions. Our results show large differences in average transport distance as well as shape and extent of predicted hatch areas, when backtracking advected larval cohorts in different regions of the North Sea, thus emphasizing the importance of using realistic, spatially and temporally resolved diffusivity fields in larval transport simulations. In all cases, a biologically likely hatching area has been predicted. We discuss issues of methodological consistency and present a new scheme for including life-history growth stochasticity effects in backtracking in a consistent way, as well as procedures for assessing effects of larval mortality. Finally, fundamental limitations of larval backtracking are clarified, most importantly the time horizon and spatial resolution limit for backward prediction.

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1 Introduction

Backtracking fish larvae is a potentially powerful tool for understanding early life-history aspects quantitatively. Backtracking may be performed either explicitly in space and time or just in time - in the latter case with implicit spatial assumptions about the environmental parameters affecting larval growth and behavior. The goal of spatial backtracking is mapping hatching areas with high resolution, to understand conditions for survival (or larval excess mortality) and thereby develop minimal conceptual models of recruitment, based on key processes. Also, optimizing the design of closed marine habitats with respect to habitat definition and closure time is a potential application of backtracking.

Backtracking can also be used as an efficient biological parameter estimation tool, which may supplement forward tracking of larvae; the biological parameters of a larval tracking model should be fitted so that a simulation develops larvae into juveniles at the right area and time. If our knowledge on juveniles are more precise than on newly hatched larvae, it may require less trajectory simulations to backtrack from juveniles (because more trajectories will be successful). Additionally, biological parameter sensitivity may be different in backtracking, compared to forward tracking.

While backtracking is frequently used in other fields like pure computer sciences(Dechter and Frost, 2002), atmospheric sciences (Uliasz and Pielke, 1991) and pollution tracing(Spivakovskaya *et al.*, 2005), larval/zooplankton backtracking is still at a premature stage, seen from a methodological point of view. However, recently (Batchelder, 2006) applied a backtracking scheme to planktonic organisms in an coastal geometry and advocated backtracking as a tool for identifying hatching sites. We go one step further and consider growth backtracking as well in a fully realistic hydrodynamic setup. Compared to passive particles, larvae have ontogenetic development as well as active behavior, which complicates backtracking, due to the stochastic and nonlinear nature of these processes.

At first sight, the issue of tracing backwards the state of a larval sample in time appears simple: just advect larvae in a direction opposite the currents and shrink their size by the same amount as they were growing, if time is running forward(Allain, 2004; Pedersen *et al.*, 2000); however dispersion processes are present in the ocean and must be considered, as we will show later, because otherwise we have no idea about the error bar on answers and further simple backtracking opposite currents lines may have an error bias, since dispersion processes in the ocean are not spatially uniform. (Batchelder, 2006) similarly found that diffusion can not be ignored in backtracking. Alternatively, larval origin might be estimated by a forward approach by releasing a vast number of particles in a forward simulation at potential spawning grounds and focus at the small number that arrives in the area of interests(Allain *et al.*, 2003). While straightforward and dodging some methodological issues of backtracking, forward-tracking remains inefficient for spawning ground identification, because the majority of particle trajectories are useless (they end up at a position different from the area of interests). This is especially true, if arrival in the area of interest is a rare event (e.g. if the final area is small or the drift time is long); this point was also emphasized by (Batchelder, 2006). Further, backtracking may potentially point to unexpected spawning sites, whereas forward-tracking has less room for surprises, because potential spawning grounds are input to the simulation.

The aim of this paper is to advocate larval backtracking as a part of a testing suite for hydrodynamical individual-based models (IBM) for larvae, both as submodel screening and validation device as well as result-generating method, and clarify the formal basis of backtracking with focus on providing tools for consistent backtracking and identify limitations of backtracking. The latter aspect is very important, because inverse problems often has no unique solution.

2 Biological and Physical model

In Lagrangian transport simulations, the positions $\{x_i(t)\}$ of an ensemble of tracers $i = 1 \ldots N$ are monitored as a function of time t, along with the state variables $\{L_i(t)\}$ of the tracers. This ensemble represents the fish larvae, with their length L_i as state variable, in our case. Each larva in the ensemble are propagated from time t to $t + dt$ by the dynamical equations

$$\begin{aligned} dL_i &= G(L_i, x_i, t)dt && (1) \\ dx_i &= \{u(x_i, t) + a(x_i, L_i, t)\}dt + d\Omega(x_i, t, dt) && (2) \\ d\Omega(x_i, t, dt) &= \nabla K(x_i, t)dt + \sqrt{2K(x_i + \frac{1}{2}\nabla K(x_i, t)dt, t)W(dt)} && (3) \end{aligned}$$

where dt is a small time increment. Eq. (1) describes the larval growth $L_i \rightarrow L_i + dL_i$, which depends on larval size, position and time in season through a deterministic relation G. It is also possible to model G as a stochastic relation, and we will return to this in Sec. 4. Eq. (2) describes the larval transport $x_i \rightarrow x_i + dx_i$, where $u(x_i, t)$ is the smooth advective field derived by linear interpolation in grid resolved currents, obtained from the hydrodynamical model described below, and $d\Omega$ is a random walk process modelling the effect of local turbulent fluctuations(Visser, 1997). $a(x_i, L_i, t))$ is the active larval motion velocity, including possibly buoyant velocity. $d\Omega$ in Eq. (3) is controlled by the local diffusivity field $K(x_i, t)$ and $W(dt)$ is a stochastic process with $\langle W(dt)\rangle = 0$ and $\langle W(dt)^2\rangle = dt$. (Because of this, the limit $\frac{d\Omega}{dt}$ does not exist, and Eqs. (1-3) can not be stated as ordinary partial differential equations, but is kept in Itô form(Rogers and Williams, 1987) as above.) The spatial distribution of an ensemble of larvae moving by Eq. (2) can also be described by the standard diffusion equation with local diffusivity K(Taylor, 1921; Visser, 1997). Eulerian versus Lagrangian approaches are reviewed elsewhere in the present issue, and we will focus on Lagrangian approaches here, noting *en passant* that equivalent results are obtained generally, if consistent dynamical equations are used. The local diffusivity field K describes subgrid advective processes, like turbulence, in statistical sense. Eqs. (1-3) describe virtual larval trajectories, i.e. without mortality. We return to mortality aspects in Sec. 5.

2.1 Forward time model

A flexible, spatially explicit sandeel IBM has been developed on the base of the ECOSMO model framework(Schrum and Backhaus, 1999; Schrum *et al.*, 2006; Hochbaum, 2004). The ECOSMO hydrodynamic model setup has in detail been validated against available observations(Janssen *et al.*, 2001; Janssen, 2002). In the present work, a biological model of the early larval life-stages of the lesser sandeel has been added, based on the underlying biophysical processes. The setup is sketched in Figure 1.

The hydrodynamic part of the ECOSMO model is based on a staggered Arakawa C-grid with a 5 nm horizontal resolution, free surface and 5 m layers down to a depth of 40 m (and 8 m layers below 40 m depth). A database(Schrum *et al.*, 2003) of 3D physical fields from the ECOSMO model (currents $u(x, t)$, temperature $T(x, t)$, and local diffusivity $K(x, t)$) has been stored as daily averaged fields (for data compression purposes) and these fields are used for the IBM simulations. $K(x, t)$ is parameterized using an analytical k-ϵ approach, considering counteracting effects of local shear and

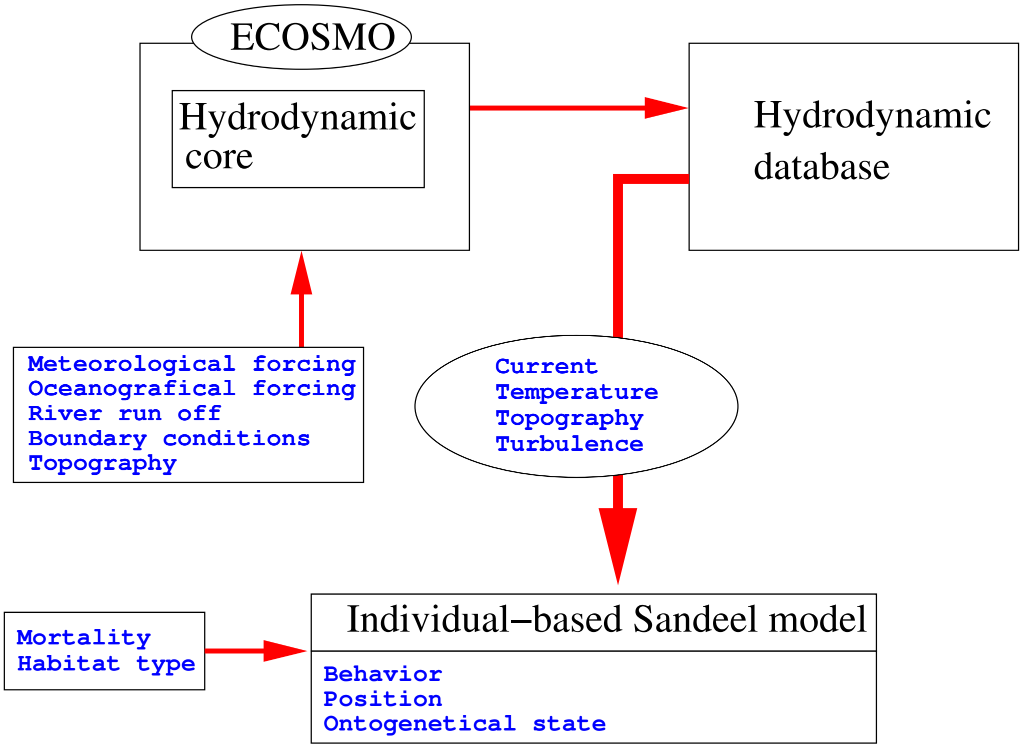

Figure 1: Sketch of coupled 3D biophysical model

stratification(Schrum, 1997).

Horizontally, the larvae are described as passive floaters, with no explicit active vertical/horizontal migratory behavior, i.e. $a = 0$ in Eq. (2). This is a good approximation during night time, but during light hours there is a predominance of larvae in the water layers with high zooplankton abundance(Jensen *et al.*, 2003). However, until the vertical behavior becomes accurately quantified for lesser sandeel, we use $a = 0$.

Only vertical turbulent dispersal is taken into account, i.e. $d\Omega$ is a vertical vector, since the dominant dispersal mechanism in the North Sea is turbulent diffusion, coupled to current layer shear(Zimmerman, 1986; Van Dam *et al.*, 1999). As larvae hatch and are advected, they disperse relative to each other. Within the dispersed patch - which is of scale kilometers - there may be small scale patchiness, due to larval schooling behavior and subscale environmental patchiness. We will not consider these additional small scale variations in the larval spatial distribution in this work, but rather concentrate on the kilometer-scale features in transport and dispersal of larval patches from same area.

The larval population is mathematically sampled by a set of representative tracers (a virtual population) which each represents a constant number of individuals (the ratio of real physical larvae per tracer needs not be stipulated, since density effects are not addressed explicitly in this study, only relative numbers matter). The boundary condition $dK(x,t)/dz = 0$ along with tracer reflection is imposed vertically at the surface and bottom to avoid artificial aggregation of tracers at the surface or bottom of the water column. The vertical random walk of Eq. (3) is implemented as $W(dt) = u\sqrt{3dt}$, where u is a uniform random distribution on $[-1, 1]$, corresponding to $\langle W(dt)^2 \rangle = dt$, so that local jump amplitudes reproduces local Eulerian field dispersal rates correctly, i.e. proportional to the square root of the local diffusivity K(x,t)(Taylor, 1921; Maier-Reimer, 1973; Hunter *et al.*, 1993).

The Lagrangian simulations were performed with a time step dt=30 minutes, using Euler forward integration. In Sec. 3 we will show that higher order horizontal integration schemes changes tracer trajectories negligibly for dt=30 minutes, i.e. trajectories are appropriately integrated numerically with dt=30 minutes, when using current fields averaged over tidal periods. Longer time steps in conjunction with higher order horizontal trajectory integration was not attempted, since this would imply large vertical jumps in the stochastic modelling of turbulent dispersal, Eq. (3).

The larval growth model in Eq. (1) is parameterized to the functional form

$$G(L,T) = \lambda(T)(\frac{L}{L_0})^{\gamma}(1 - \frac{L}{L_\infty}) \tag{4}$$

where $T = T(x,t)$ is the local temperature experienced by each larva. The data set used for parameterization is North Sea length at age samples for *Ammodytes Marinus* obtained by MIK trawl data from the years 1995 and 1996 (pooled together)(Jensen, 2001). Larval ages were obtained by otolith analysis(Jensen, 2001). The temperature modulation $\lambda(T)$ is approximated by a quadratic polynomial:

$$\lambda(T) = a_0 + a_1 T + a_2 T^2. \tag{5}$$

In Fig. 2, we show the data and model fits, based on analytical integration

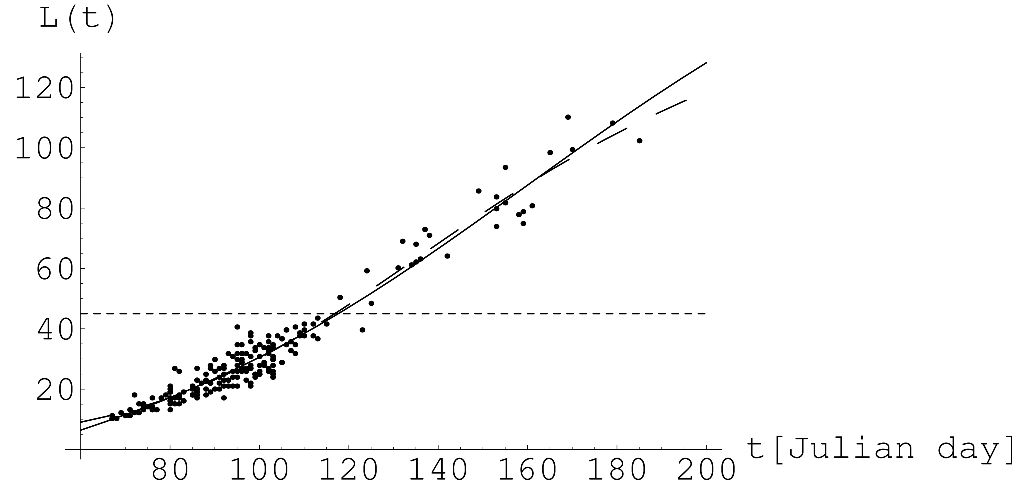

Figure 2: Pooled length-age data (points) used for parameterizing the lesser sandeel growth model. The horizontal line at $L_m = 45$ mm corresponds to the approximate length of metamorphosis. The full line corresponds to model v1, the dashed line model b1, see Table 1.

of Eqs. (1) and (4), including the North Sea average seasonal temperature variation; it is important to include the average seasonal temperature variation in the parameter estimation process, since the average temperature rises from around 5 °C at hatching time to around 12-14 °C at the time where metamorphosis is complete, ($L_m \sim 45$ mm)(Wright and Bailey, 1996; Jensen, 2001); otherwise model forward simulation will not reproduce the growth pattern in Fig. 2. It appeared difficult to simultaneously resolve $\lambda(T)$ and the length scaling exponent γ, because the seasonal temperature variation and larval length are strongly correlated in the data set. There is a shallow residual minimum in the fit at $\gamma \sim 0.96$, but fits constrained to $\gamma = 0$ (model v) or $\gamma = 1$ (model b) produce essentially equally good fits for 0-group sandeels, as shown in Fig. 2. The parameters of the respective fits are given in Table 1, along with the estimated hatch lengths L_0, all of which are consistent with observations(Winslade, 1971; Smigielski *et al.*, 1984) of $L_0 \sim 5 - 7$ mm. We use $L_\infty = 218$ mm(Macer, 1966), which is not included as a free parameter when fitting data in Fig. 2, since it is of minor importance for larval growth (since ($L_m << L_\infty$) and because the data set only has observations for $L << L_\infty$ (but L_∞ adds a little concavity to the growth curve).

model	γ	L_0	a_0	a_1	a_2	ρ
μ	-	mm	-	$(°\mathrm{C})^{-1}$	$(°\mathrm{C})^{-2}$	mm
v1	0	6.40	-0.354	0.167	0	10.8
v2	0	8.91	-1.81	0.515	-0.0196	11.9
b1	1	9.05	0.422	-0.0205	0	12.0
b2	1	9.15	0.401	-0.0153	-0.000272	12.1

Table 1: Parameters for the alternative sandeel growth models. The fitting residual is normalized as $\rho = \sqrt{\frac{1}{M}\sum_{i=1}^{M}(L^\mu(t_i) - L_i)^2}$, where $\{L_i, t_i\}_{i=1\dots M}$ is the length-age data set and $L^\mu(t)$ is the growth curve for model μ.

From Fig. 2 it is seen that seasonal temperature variation is a surprisingly good proxy for the complex bioenergetic effects (spectrum, abundance and distribution of food, as well as complex food switching patterns)(Letcher *et al.*, 1996; Baron, 2004). The quantitative growth predicted by Eq. (4), as parameterized here, is in good quantitative agreement with that of Gallego(Gallego *et al.*, 2004) presented recently. Since the model parameterizes *in situ* data, the model also integrates the prey dependence on temperature variations, at a crude level. Currently, the model does not address density effects (e.g. food competition and cannibalism) explicitly, although they are believed to affect recruitment(Daan *et al.*, 1990; Kishi *et al.*, 1991; Kimura *et al.*, 1992; Arnott and Ruxton, 2002). There are also indications that population density effects may affect individual sandeel growth, currently reported for sandeel populations in the North Sea(Bergstad *et al.*, 2002) and Ise Bay(Nagoshi and Sano, 1979), but the influence has not yet been sufficiently quantified for larval stages.

2.2 Reversed time model

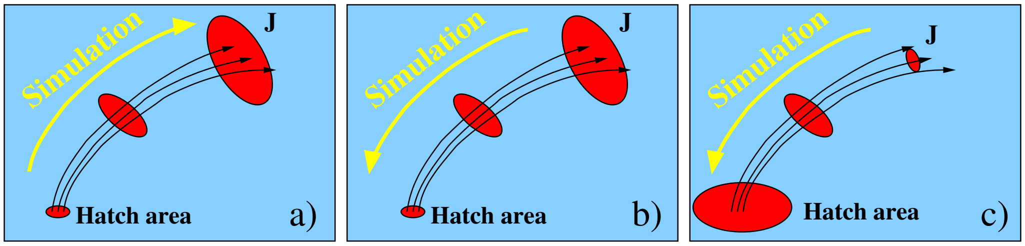

Figure 3: Sketch of different types of hydrodynamical simulations. Arrows indicate current steam lines. a) Normal forward time simulation of a larval patch (in red), starting from the hatch area with larval ending with a juvenile distribution at J. b) Inverse time simulation of a larval patch (in red), starting from a juvenile sample at J. c) Reversed time simulation of a larval patch (in red), starting from a juvenile sample at J.

Let us start at the heart of the problem, which is sketched in Figure 3. Figure 3a illustrates the familiar forward time modelling situation, where a larval patch (in red) is traced from the hatching area to the patch J at a later time (which could symbolize the distribution of juvenile larvae). The larval patch increases in spatial extent, as the patch is advected along current lines, due to spatial dispersive processes. If we now catch a portion of the larvae in the patch J and ask the model where they came from, many people anticipate the situation in Figure 3b: the model, when "run backwards", should converge to the hatch area, where the forward simulation started from. We call this *inverse time* simulation. Since the Fourier spectrum of inverse time dispersion diverges for small spatial scales, the inverse time simulation is not numerically stable(Hadamard, 1923). The physical meaning of this is that forward-time diffusion quickly smears out fine-scale features, while inverse time simulations blow them up. Therefore, a given final distribution can originate from very different initial distributions by forward-time diffusion. We want to calculate the expectation of all these possible initial distributions leading to a given final distribution; this is called *reverse time* simulation and is sketched in Figure 3c. The major difference, compared to inverse time simulation in Figure 3b, is that the larval patch will disperse in space when traced backwards in time. The point is that dispersive (and generally stochastic) processes delete knowledge on initial state when time progresses and therefore the uncertainty on the larval origin also increase when they are traced back in time. In other words, given a larva in a specific place and state, we can only give a spatial probability distribution of places where it is likely that it has hatched, because it can end up in a specific place and state along many different life-history paths, when dispersive (and generally stochastic) effects are present in the model. This limitation is fundamental and insurmountable, and has important consequences. First, if we neglect spatial dispersive processes (if this is possible) and just backtrack along current lines, we will end up at a point somewhere in the hatch area distribution in Figure 3c, but we have no guaranty that the point is at the center of the hatch area distribution (and it will not be at the center, when the dispersive processes has spatial gradients, which is usually the case) and we have no idea about the characteristic size of the hatch area distribution. Secondly, it implies a characteristic past time horizon, beyond which we will not be able to back-trace, because the possible starting places covers all possible spawning areas. We will return to this issue in Sec. 5.

Having now established that reversed time simulation is the appropriate methodology to identify hatch areas / hatch schedule probability distributions from a given larval catch, we will focus on reversed time simulation in the rest of this paper. We also note probability distributions are obtained from larval ensemble trajectories by any standard smoothing technique. The spatial extent of larval distribution are obtained either from form parameters of the smooth distributions or identifying areas where probabilities are larger than a given tolerance level. All qualitative conclusions below are unaffected by these technical steps and choices. Formally, Eqs. (1-3) cover two process classes: deterministic advection processes (by fields G, u, a) and local dispersal $d\Omega$. Both these process classes are unambiguously reversible: deterministic advection terms change sign, whereas local dispersal $d\Omega$ keeps its sign. This is due to the fact that random walk processes are fundamentally time reversible: it is impossible to judge whether the clock runs forward or backward from a random walk trajectory $x(t)$ in a stationary (or slowly varying) diffusivity field; more directly, $x(t)$ and $x(-t)$ are equally likely. Further, since the stationary state of a dispersive process is spatially uniform, the backward dynamics dispersion amplitude corresponds to the forward dispersion amplitude K. In other words, $d\Omega$ is formally invariant under time reversal. Hence, each larva in the ensemble is propagated from time t to $t - dt$ by the dynamical equations

$$\begin{aligned} dL_i &= -G(L_i, x_i, t)dt && (6) \\ dx_i &= -\{u(x_i, t) + a(x_i, L_i, t)\}dt + d\Omega(x_i, t, dt) && (7) \\ d\Omega(x_i, t, dt) &= \nabla K(x_i, t)dt + \sqrt{2K(x_i + \frac{1}{2}\nabla K(x_i, t)dt, t)W(dt)} && (8) \end{aligned}$$

where $dt > 0$ is a small time step backward. We note that Eqs. (6-8) are the consistent way of running the forward model, Eqs. (1-3), backwards. When any aspect of the forward model is changed, the corresponding change must be performed in the reversed time model. In Sec. 4 we will discuss the impact and complications on backtracking, arising when G in Eq. (6) is generalized to a stochastic function reflecting life history stochasticity.

As an implementation remark, we note that reverse time particle tracking are most simply performed offline, i.e. current fields etc. are be taken from a precalculated database, generated by running a hydrodynamical setup forward in time covering the period of interest. It is possible to run a hydrodynamical model backward in time as well(Griffin and Thompson, 1996), using adjoint primitive equations. This avoids large amounts of data I/O, but may be more CPU intensive, depending of the hydrodynamical resolution. However, many realistic, operational hydrodynamic setups does not offer this advanced feature.

Apart from this, implementation of Eqs. (6-8) is a straightforward modification of the implementation of Eqs. (1-3): they are solved by trajectory integration backward in time, exactly like Eqs. (1-3) forward in time.

3 Spatial and temporal backtracking

To illustrate the approach, we perform backtracking of three representative samples of larvae caught at different locations in the North Sea in

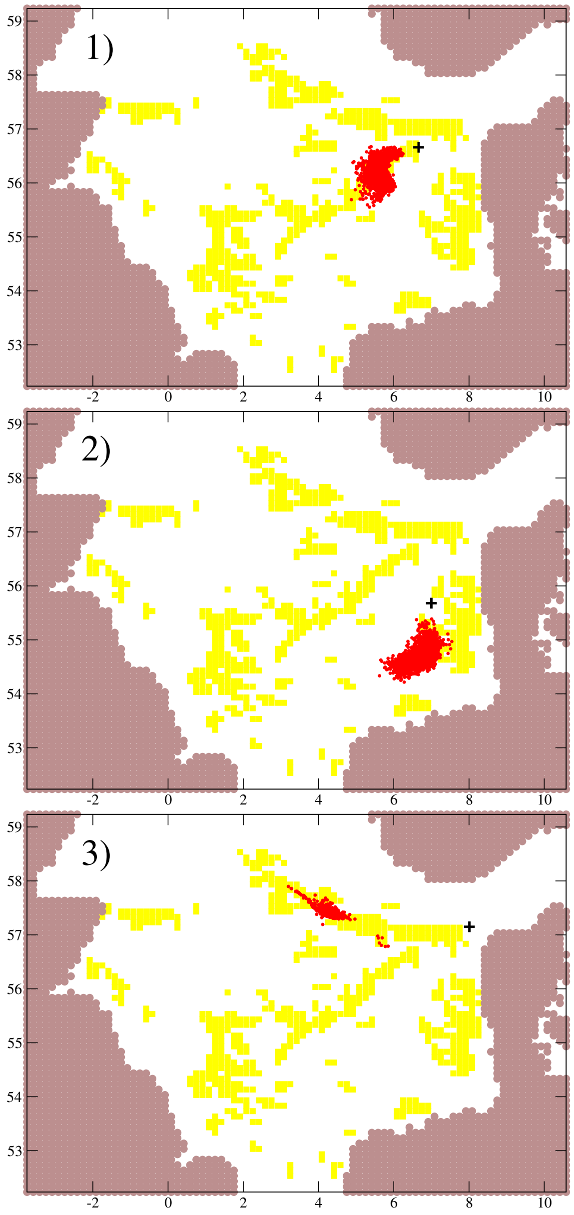

Figure 4: Backtracking of larval samples in Table 2, with sample number indicated as figure label, using biological model v1. Inland areas are colored brown, sand banks suitable for sandeel habitats are colored yellow. Larval catch position is indicated with + symbol. Back-traced larval ensemble positions are small, red circles that indicate potential hatch positions.

sample	catch date	catch position	catch statistics		
			$\langle L \rangle$ [mm]	$\sigma(L)$ [mm]	$\langle \text{age} \rangle$ [days]
1	April 22 (2001)	6.66°E, 56.66°N	18.2	1.6	23
2	April 27 (2001)	7.00°E, 55.68°N	19.9	1.1	33
3	May 25 (2001)	8.01°E, 57.15°N	22.0	1.0	32

Table 2: Catch data for three representative larval samples used for spatial backtracking examples.

2001(LIFECO, 2004). The catch samples are summarized in Table 2. In Fig. 4, we show the results of backtracking larval samples in Table 2. The figures show larval catch position, along with the spatial hatch probability distribution. The ensembles have been initialized at the catch position at catch time with normal length distribution, with form parameters from Table 2, and traced backward, until they have hatch length $L = L_0$, using growth model v1. The figures are overlayered with identified sandeel fishing banks, as obtained from detailed fishery loggings(Jensen and Rolev, 2004). Habitat data has been projected onto the hydrodynamic grid, so that length scale features below approximately 10 km are not resolved. It is assumed that larvae must originate from some of these sand banks, as sandeel spawn demersal eggs within their habitats(Reay, 1970). Each ensemble size in Fig. 4 contains 2000 individuals, in order to roughly map the spatial hatch probability distribution. Sample 1 is caught at the eastern tip of the central bank system and is under influence of cyclonic North Sea circulation system, which in this area normally results in north-easterly transport.

The sample is traced back to the eastern central bank system around the "Tail end" fishing area, approximately (5.5°E, 56°N), i.e. retainded on the same major bank system. Sample 2 has been advected along the northward Jutland coastal current, and is traced back to the south western part of the Jutland sand bank system, approximately (7°E, 54.5°N). Sample 3 is under influence of the Norwegian trench inflow and traced back to the middle of the northern major sandeel bank system, around the "Klondyke" fishing area, approximately 200 km west of catch position.

Comparing the three examples, we see prominent differences in shape and extent of the predicted spatial hatch probability distributions, and also a clear difference in the advection distance, i.e. distance between most likely hatch position and catch location. This puts clear emphasis on the necessity on explicitly including realistic advection ($u(x,t)$) and dispersal fields ($K(x,t)$) when back-tracing larval ensembles. In all considered cases, the hatch probability distribution has a significant overlap with a sand bank system suitable for sandeels habitat(Jensen and Rolev, 2004).

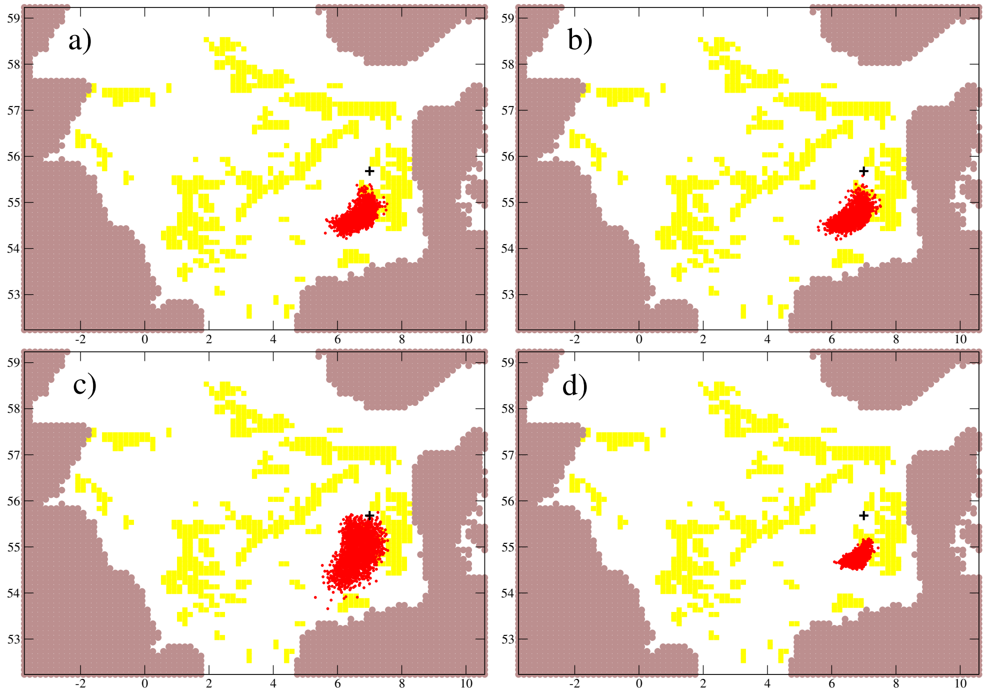

Figure 5: Sensitivity test on model parameters, shown for backtracking of larval sample 2 in Table 2. Inland areas are colored brown, sand banks suitable for sandeel habitats are colored yellow. Larval catch position is indicated with + symbol. Back-traced larval ensemble positions are small, red circles. Biological models/tracing algorithms are a) v1/Euler b) v1/Runge-Kutta 2^{nd} order c) v2/Euler d) b1/Euler

In Fig. 5, we test the sensitivity of central model parameters for sample 2 in Table 2. Comparing Figs. 5(a) and 5(b) we see that the effect of changing the horizontal trajectory integration algorithm from Euler to Runge-Kutta 2^{nd} order(Press *et al.*, 1992) (with same time step) is negligible. If tidal current fluctuation were not averaged out, Euler trajectory integration would display much larger errors for same time step, due to the rotating nature of and high amplitudes of tidal current fluctuation in the North Sea. Comparing Figs. 5(a) and 5(c) we see the effect using a quadratic temperature modulation $\lambda(T)$ (model v2) instead of a linear temperature modulation (model v1): the predicted hatch areas are similarly centered, but the dispersal is significantly larger. Finally, comparing Figs. 5(a) and 5(d) we see the effect of using a length scaling exponent $\gamma = 1$ (model b1) instead of $\gamma = 0$ (model v1). Again, the predicted hatch centers are the same, but the dispersal somewhat smaller. Generally, we predict approximately the same hatch centers for all models, but with some fluctuation in dispersal. There is no systematic bias in dispersal patterns, when comparing models. Generally, we find that the effect of changing horizontal trajectory integration algorithm from Euler to Runge-Kutta 2^{nd} order is negligible.

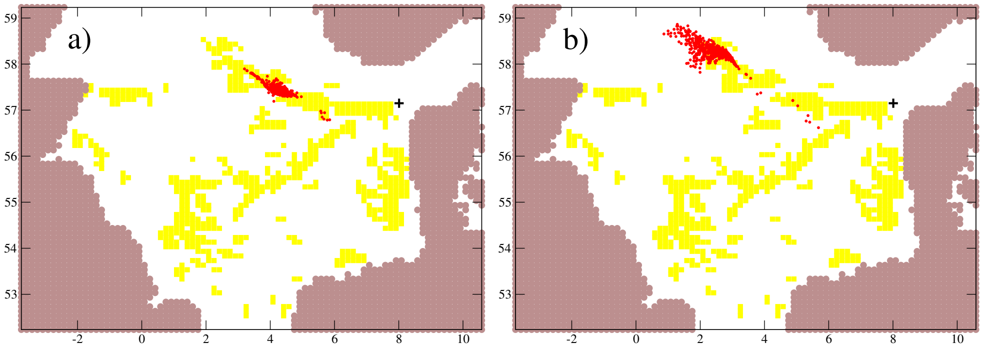

Figure 6: Anomalous large parameter sensitivity: sample 3, using a) model v1 and b) model b1. Inland areas are colored brown, sand banks suitable for sandeel habitats are colored yellow. Larval catch position is indicated with + symbol. Back-traced larval ensemble positions are small, red circles.

Fig. 6 displays an anomalous large parameter sensitivity in one of our simulations: sample 3 using model v1 versus model b1. In this case there is an exceptional large offset between predicted hatch areas, in the order of 100 km, but both within same major bank system. This is the largest variation found, and we stress that this case is isolated; the normal variability picture is as illustrated in Fig. 5. However, these isolated cases can also be very useful, because they provide a clear prediction for validating specific models, when additional data is present to compare with. Unfortunately, we do not have data to ground truth larvae in samples 1-3 by secondary means.

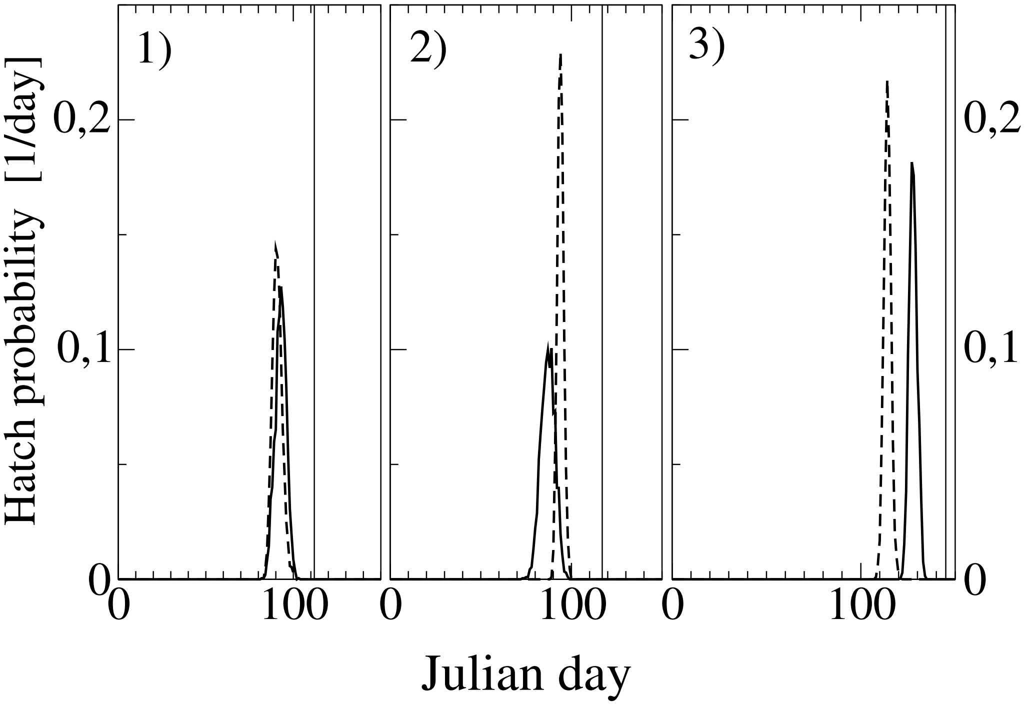

Figure 7: Hatch time distributions from backtracking of larval samples in Table 2, sample number indicated by figure label. Full line is model v1, dashed line is model b1. Thin vertical lines indicates catch time of sample.

Fig. 7 shows the hatch time distributions from backtracking of larval samples in Table 2, where the spatial and temporal variability of local physical environment has been included, i.e. these hatch time distributions correspond to the spatial distributions in Fig. 4. Although models v1 and b1 overlap on the spatial prediction of hatch area, somewhat larger relative differences appear on the prediction of drift period, with sample 3 again being an outlier, but the overall predicted hatch periods are in reasonable agreement with age assessment from otolith reading (Table 2). Both growth models almost have the same prediction of the hatch peak for sample 1, model v1 performs a little better for sample 2, whereas model b1 is a little better for sample 3, so there is no apparent bias toward length of drift period between models v* and b*, but both models indicate that there was large spatial heterogeneity in hatch periods over the North Sea in 2001, consistent with survey observations(LIFECO, 2004).

4 Growth stochasticity

Growth variability within an ensemble of larvae arises from many sources, e.g. genetic variability, food patchiness and other environmental fluctuation on a subgrid scale. Growth variability over larger spatial scales will also appear indirectly, if spatial effects are not explicitly represented. From Fig. 2 we can roughly estimate the relative growth variability (i.e. $\sigma(G)/\langle G\rangle$) to be in the order of 20 % for sandeel larvae in the North Sea (this estimate is an upper limit, because it aliases some spatial and temporal variability as fundamental growth variability, because data in Fig. 2 is pooled). However, it is reasonable to expect different levels of variability for other fish species and other areas.

At this point it is convenient to shift to a discrete representation in time and larval size, so the larval ensemble is characterized by a distribution vector p_t giving the size distribution of larvae in a suitable set of length classes at time t. If Eq. (1) is integrated forward by a fixed, small time step $dt = h$, the larval length distribution development is characterized by the matrix Γ^h

$$p_{t+h} = \Gamma^h p_t \tag{9}$$

which can be considered a Markov process, when the time scale of average temperature changes is large compared to h. Uniform mortality can be handled by multiplying a prefactor to Eq. (9) - this does not alter the qualitative discussion. Put in another way, we focus on the relative length characteristics of a larval ensemble. The matrix Γ^h_{ij} gives the transition probability between length classes j to i during the time step h. Bayes' theorem provides the time reversed process characterized by the matrix Q^h

$$\begin{aligned} p_t &= Q^h p_{t+h} &\quad (10) \\ Q^h &= (\kappa * \Gamma^h)^T &\quad (11) \end{aligned}$$

where κ_{ij} is the ratio of prior probabilities in states j, i and $*$ means element-by-element matrix product. For regular Markov processes (as spatial dispersion) κ is straightforward, but for oriented stochastic processes, like growth, the larval/juvenile length classes have zero probability in the stationary state (all larvae have become adults) and therefore the prior ratio κ is ill-defined. We have recently proposed an alternative direct scheme(Christensen, 2007) to compute κ, which resolves the problems with the usual definition. A peculiarity of Eq. (11) is that a process which is deterministic in forward time direction may become stochastic in reversed time (or vice versa) due to the transpose of the forward Markov matrix Γ^h.

In Fig. 8 we illustrate the effect of growth stochasticity by backtracking larval ensembles for 40 days under different premises. Growth stochasticity is modelled adding a stochastic width of $\sigma(G) = 0.2G$ to the average growth G in Eq. (4), consistent with Fig. 2 (this means absolute growth fluctuations are smaller for early larvae than juveniles). The growth Markov matrix Γ^h in Eq. 11 in this example is obtained by integrating Eq. (4) corresponding to $h = 2$ days and projecting onto size class bins of 1/3 mm. The backward growth Markov matrix Q^h has then been generated using Eq. 11. Fig. 8(a,b) shows that growth stochasticity adds width (full line) to the sharp length distribution obtained by deterministic backtracking (dashed line) of a narrow length distribution; the length distribution of growth model b1 (Fig. 8(b)) is more narrow than that of growth model v1 (Fig. 8(a)) and also biased off center. We see that model v1 displays larger sensitivity than model b1, which can be traced to the fact that model b has smaller temperature sensitivity, because length scaling of growth explains growth variation of very small larvae.

Fig. 8 (c) shows that effects of typical growth variability are dominated by typical variability in length distribution. This conclusion, however, need not be true for other species and other areas, especially if backtracking is performed for a longer part of their life history. We also emphasize the

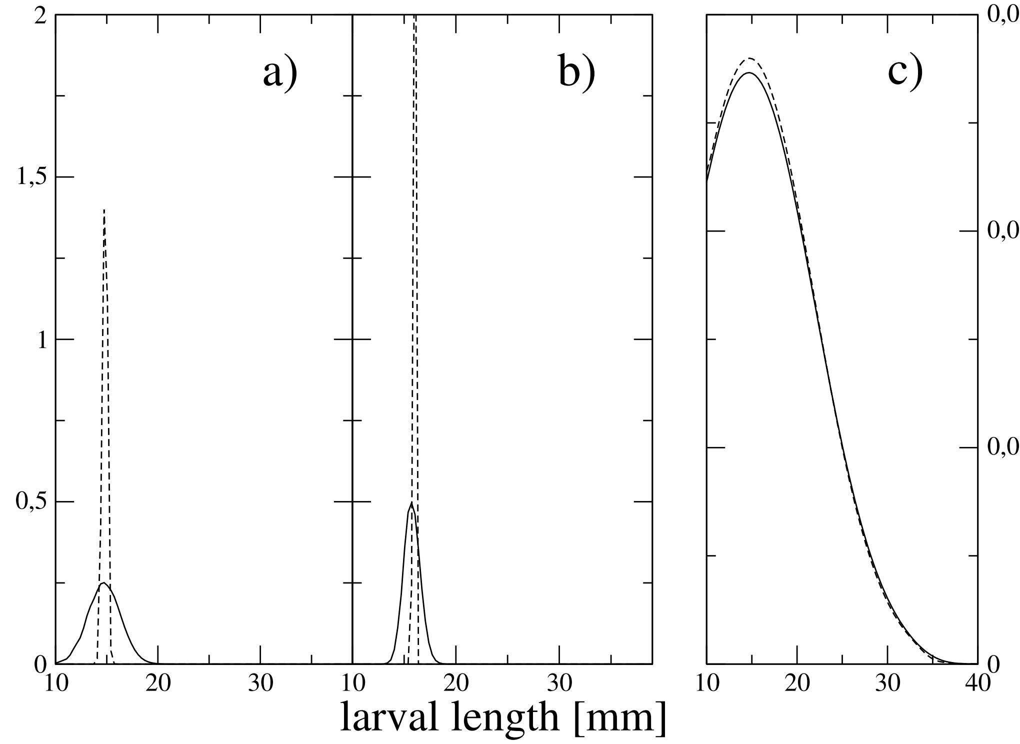

Figure 8: Larval ensembles backtracked for 40 days, starting on Julian day 120 with initial average length $\langle L \rangle = 50$ mm. a) initial length RMS $\sigma(L) = 0$ mm and growth model v1 b) initial length RMS $\sigma(L) = 0$ mm and growth model b1. c) initial length RMS $\sigma(L) = 10$ mm and growth model v1. Growth models are discretized in size classes of 1/3 mm, according to Eq. (9). Full lines show ensemble backtracking, using Eq. (11), with relative growth stochasticity $\sigma(G) = 0.2\bar{G}$ and dashed lines deterministic ensemble backtracking, using Eq. (6), i.e. $\sigma(G) = 0$.

importance of a realistic prior κ, if the objective is to recover time scales, e.g. hatch schedules.

5 Discussion

In dispersal processes, transport distance R and transport time t scales(Taylor, 1921) are linked by

$$R = \sqrt{\bar{D}t} \tag{12}$$

on a coarse scale, where $\bar{D}$ is a characteristic average horizontal dispersal rate. For the North Sea in the larval drift period, Figs. 4 indicate that roughly $\bar{D} \sim 20$ km^2/day ~ 230 m^2/s, which is within the normal range, $\bar{D} \sim 100 - 1000$ m^2/s, of encountered dispersal rate in tidal dominated waters(Zimmerman, 1986). This supports the previous assertion(Zimmerman, 1986) that the dominant horizontal dispersion pathway was vertical diffusivity coupled to vertical current shear. Explicit inclusion of other horizontal dispersion pathways can be expected to increase the average horizontal dispersal rate $\bar{D}$ somewhat, so our diffusivity fields are lower bounds on the total diffusivity. However, we believe that it will not change our conclusions, nor the qualitative picture we draw in our paper, because they hold also for increased spatial diffusivity fields. In the future, we plan to include other horizontal dispersion pathways for the sake of completeness.

In our context, this relation has two important implications:

- A time horizon t_c: if we want to backtrack larvae and localize them with a spatial accuracy R (or better), we can only simulate backwards for $t < t_c = R^2/\bar{D}$, before dispersion exceed the desired spatial accuracy R. It should also be observed that R exceeds the twice the spatial grid resolution, which in our case is 2×10 km (this is the resolution limit of current structures) - otherwise the spatial grid resolution should be increased correspondingly. This lower grid resolution limit corresponds to roughly 20 days backtracking in the North Sea.

- Spatial resolution limit R_c: if we want to simulate backwards for a specific time period t the spatial accuracy on the answer is $R_c = \sqrt{\bar{D}t}$. For lesser sandeels in the North Sea, the drift period is of order two months, so that the fundamental backtracking resolution for settled juveniles is roughly $R_c \sim 30$ km. This is smaller than typical area resolution in Fig. 4 - this is due to the fact that larval cohorts in Fig. 4 have a significant length variance, see Table 2. On the other hand, including cohort length distribution directly gives the time distribution of probable hatch.

Spatial backtracking provides an initial probability distribution. However, care must be taken in the interpretation these probability fields. Batchelder (Batchelder, 2006) suggested that the fraction of origin of a planktonic can be determined by considering overlap with backtracked probability distribution; this can only be accomplished, if all sites of origin have a uniform prior distribution of the planktonic organisms in question. Further, the mortality must be spatially uniform. Batchelder also suggested time round trip experiments as an appealing and intuitive validation step of backtracking schemes. Here, a given situation is first run forward in time for a selected interval. Then the resulting spatial distribution run backward in time for the same time interval, and final spatial distribution overlaid with the initial state. In the light of our comments above, we want to emphasize that the final spatial distribution (i.e. after the time round trip) should not generally be expected to be strongly overlapping with the initial state. The overlap can be made arbitrary small or large, depending on the choice of round trip time interval. It is not a question about good or bad performance, but speed of information loss, which is given by the local diffusivity. There is only a negative validation, if there is strictly no overlap between final spatial distribution and initial state. We also want to note, that the center of gravity in the final spatial distribution should not be expected to coincide with initial state center of gravity, when the diffusivity is spatially heterogeneous.

Another interesting mechanism for particle dispersal is the potential presence of Lagrangian chaos(Zimmerman, 1986). The presence of Lagrangian chaos is determined completely by the structure of the advective flow field transporting the particles. If it is due to spatial current structures above grid scale, it is automatically picked up by trajectory integration. However, if it is due to subgrid scale spatial current structures, it must appear as a contribution to the spatial turbulent density $K(x,t)$. An interesting aspect in this context is that diffusive trajectories starting close to each other diverge as square root of time, whereas chaotic trajectories diverge exponentially with time. Since we have demonstrated the importance of using realistic dispersal fields $K(x,t)$, future studies should address the nature and parameterization of subgrid scale Lagrangian chaos contributions to particle dispersal, as well as disentangling the overlap with standard turbulent dispersal parameterizations, to avoid double accounting.

Even though the hydrodynamic database underlying our study has in detail been validated against available observations(Janssen *et al.*, 2001; Janssen, 2002), as pointed out in Sec. 2, some level of circulation field uncertainty must be expected to be present. At the most crude level, this can be modelled as an additional, homogeneous diffusivity. In this case, it will not affect the qualitative conclusions of this paper. At a more sophisticated level, this can be modelled as a spatially and temporally dependent autocorrelated random walk process, overlayered on the particle tracking described above. However, it is beyond the scope of the present paper to parameterize such an error model process.

We have not yet discussed the choice of larval ensemble size N, because we have focused on qualitative results rather than quantitative results. Two guiding figures apply here. If we want to observe a biological event in our simulations (e.g. a larva crossing a critical point in its life cycle) which has an approximate probability p, then $\frac{1}{p} < N$ should be satisfied. Conversely, if we are limited by computer resources to an ensemble size of N_c, we can only expect to observe events more likely than $\frac{1}{N_c} < p$. If we want to estimate p numerically in our particle tracking (with dispersal) we are bounded by the counting noise, which gives the relative variance our estimate $\frac{\sigma(p)}{p} \sim \frac{1}{\sqrt{pN}}$. This criterion is harsh to meet in reality, if p is small, and acceleration techniques, like e.g Brownian Bridges(Rogers and Williams, 1987) may become necessary.

Finally we want to discuss the influence of mortality on our results. We have focused on virtual larval trajectories. If particles are passive and without internal states (e.g. size and condition), (or mortality is independent of internal states) these trajectories can be corrected *a posteori* by any mortality schedule. If mortality is spatially homogeneous, mortality effects will not affect many relative properties (like relative survival of ensemble sub groups) or survival chance will trivially depend on drift time. In the case of a spawning site distribution back-tracked from a catch location, like in Fig. 6, spatially homogeneous mortality will not change the predicted spawning site distribution. Thus one could say that all our examples in the present paper are also valid for a constant mortality level. If the mortality level is spatially and/or temporally varying, changes in the predicted spawning site distribution must be expected.

An interesting example be to study the impact of spatial predator distribution; in this case survival chances along virtual larval trajectories (both forward and backward in time) can be assessed as path integrals along virtual particle trajectories of the spatial predator distribution. Virtual particle trajectories have another advantage, when comparing different mortality scenarios: the same set of particle trajectories can be computed once, stored, and used for all mortality scenarios. Reusing trajectories has the added advantage that the $\frac{1}{\sqrt{N}}$ sampling noise level is suppressed. If particles are active and activity depends on internal states, *a posteori* is not possible, and a comprehensive discussion of the issue is beyond the scope of the present paper. If particles with internal states in forward-tracking die, they are removed from the ensemble. What happens in back-tracking? Then they must be added to the ensemble at a rate corresponding to the local mortality. The less trivial question is what internal state un-died (added) particles should be assigned. Clearly more research is needed to address this open question.

6 Conclusions

Forward simulation has often been used as a device to discriminate different potential underlying biological mechanisms. We have demonstrated larval backtracking as a versatile tool, complementary to normal forward simulations for model validation. Backward processes may exhibit a sensitivity not present in the forward processes. Model sensitivity is traditionally considered a weakness, because it makes assumptions important. Conversely, strong model sensitivity can be considered a powerful model validation asset, in the presence of auxiliary data - we advocate the latter point of view, used carefully, as a constructive attitude.

We have illustrated larval backtracking for North Sea lesser sandeel larvae and tested several alternative biological growth models. We have found that dispersal effects are important for larval backtracking predictions, with large differences in shapes and extent of predicted hatching areas for larval patches originating from different regions of the North Sea, as well as large differences in the average advection distance. This emphasizes the need for future studies on the quality of sub grid scale turbulence parameterization, including possibly the effect of sub grid scale Lagrangian chaos. We have found backward prediction of hatch area more robust to model parameters than backward prediction of temporal hatch schedule. In all considered cases, a biological reasonable hatching area has been predicted, without nudging the model in this direction. We have generally found reasonable agreement between backward predicted temporal hatch distribution and otolith reading data.

We have clarified fundamental limitations of larval backtracking due to information loss in stochastic processes, most importantly the time horizon and spatial resolution limit for backward hatch area prediction. For juvenile sandeel larvae in the North Sea, the lower backtracking resolution limit is 30 km, increasing to 50-100 km, if growth stochasticity or/and cohort length variance is included. The accuracy of backtracking is bounded by turbulence processes on long (monthly) time scales and spatial hydrodynamic resolution short (weekly) time scales.

Finally, we have presented a new general scheme for deriving the consistent backtracking equations for a general stochastic larval growth model. We have found that growth stochasticity adds uncertainty to the backtracked hatch estimate, similarly to hatch area prediction. For North Sea lesser sandeel, we estimate the relative growth rate variability to be in the order of 20 %. If the larval ensemble has a broad length distribution, it may be sufficient to use deterministic backtracking (i.e. use average growth instead of stochastic growth), provided growth stochasticity is small or moderate, which is found to be the case for North Sea sandeels.

In backtracking perspective, the most rewarding development on the biological side will be linking growth variability to the local biophysical environment, in order to capture effects of regional, seasonal and interannual differences in environmental conditions with respect to growth variability. Also more work is needed in elaborating unresolved processes and uncertainties on both the biological side, with emphasis on active behavior and physical cue responses, and on the hydrodynamic side, with emphasis on improving the parameterization of hydrodynamic dispersal fields.

Acknowledgments

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Annex 6 Variation in the abundance of sandeels Ammodyes marinus off southeast Scotland: an evaluation of area-closure fisheries management and stock abundance assessment methods

Greenstreet, S., E. Armstrong, H. Mosegaard, H. Jensen, I. Gibb, H. Fraser, B. Scott, G. Holland and J. Sharples,

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Variation in the abundance of sandeels *Ammodytes marinus* off southeast Scotland: an evaluation of area-closure fisheries management and stock abundance assessment methods

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In 2000, a sandeel fishery off SE Scotland, which commenced in the early 1990s, was closed in response to concerns that the fishery was having a deleterious effect on seabird breeding performance at colonies in the vicinity of the fishing grounds. Fishery-derived catch per unit effort (cpue) data are used together with three different fishery-independent survey techniques (acoustic, demersal trawl, and nocturnal grab survey) to assess variation in sandeel *Ammodytes marinus* population biomass in the area over the period 1997−2003, a period that included the last three years that the fishery was operating and the first four years of the sandeel fishing moratorium. Temporal trends in estimates of sandeel biomass derived from these different assessment methods were inconsistent and, on the basis of these alone, it was not possible to determine whether sandeel population biomass in the area had increased following the closure of the fishery. The different survey methods assess different components of the sandeel population; acoustic survey and fishery cpue quantified sandeels active in the water column, whilst demersal trawl survey quantified sandeels buried in the sediments. These data were collected at a time of year when sandeels were moving between the seabed sediments and the overlying water column. A grab survey also quantified sandeels buried in the sediment, but these data were collected at a time of year when the entire population should have been buried in the sediment. Differences between the different time-series were reconciled by taking account of the cumulative total primary production in each year prior to the surveys. On the basis of this, a model was developed that utilized acoustic and demersal trawl survey data to estimate the total sandeel population biomass. This model was validated using the nocturnal grab-survey data. The modelled data indicated that the biomass of sandeels 1+ years old increased sharply in the first year of the closure and remained higher in all four of the closure years than in any of the preceding three years, when the fishery was operating. The biomass of 0-group sandeels in three of the four closure years exceeded the biomass present in the three years of commercial fishing. Whereas the response of 1+ sandeels may have been a direct consequence of the closure, this is not likely to have been the case in respect of 0-group sandeels. The closure appears to have coincided with a period of enhanced recruit production.

Keywords: fishery closure effects, fishery-dependent assessment, fishery-independent assessment, local stock biomass assessment, sandeel behaviour modelling.

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Introduction

Fishing for small fish for industrial purposes in the North Sea commenced in the 1970s and expanded rapidly during the 1980s. Sandeels *Ammodytes marinus* are the principal species targeted by this industrial fishery. In recent decades annual landings of sandeels have topped one million tonnes on occasion, and have rarely dropped below 0.5 million tonnes (ICES, 2002, 2004). Sandeels are therefore the target of the largest single-species fishery in the North Sea (Gislason and Kirkegaard, 1998), and they also constitute important prey for many top predators. They are an important part of the diet of commercial fish species, such as cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), whiting (*Merlangius merlangus*), saithe (*Pollachius virens*), and mackerel (*Scomber scombrus*; Daan, 1989; Daan *et al.*, 1990; Hislop *et al.*, 1991; Hislop, 1997; Greenstreet *et al.*, 1998). Several marine mammals feed intensively on sandeels during spring and early summer. Examples are grey seals (*Halichoerus grypus*; Hammond *et al.*, 1994), harbour seals (*Phoca vitulina*; Pierce *et al.*, 1991; Tollit and Thompson, 1996; Tollit *et al.*, 1997; Brown *et al.*, 2001), and harbour porpoise (*Phocoena phocoena*; Santos and Pierce, 2003). Sandeels are also the principal prey of many of the most abundant seabird species that feed in the North Sea, including common guillemots (*Uria aalge*), razorbills (*Alca torda*), Atlantic puffins (*Fratercula arctica*), northern gannets (*Morus bassanus*), and black-legged kittiwakes (*Rissa tridactyla*), particularly during the breeding season (Tasker and Furness, 1996; Wanless *et al.*, 1998; Furness and Tasker, 2000).

The North Sea sandeel stock is assessed annually and, until recently, these assessments suggested that current levels of fishing mortality were not excessive (ICES, 2002). However, there is increasing evidence that sandeels in the North Sea do not constitute a single homogenous stock, but may instead consist of several discrete stocks (Proctor *et al.*, 1998; Pedersen *et al.*, 1999). Fishing effort for the species is also patchily distributed, raising the possibility that overexploitation of sandeel stocks may have taken place at local spatial scales (Frank and Brickman, 2001). With so many marine top predators heavily reliant on sandeels, the potential for local overexploitation has increasingly given cause for concern (Ormerod, 2003). Many sandeel fishing grounds are close to major seabird colonies, so the overlap between the distributions of post-settlement sandeels, seabirds at sea, and industrial fishing activity is considerable (Jensen *et al.*, 1994; Wright and Begg, 1997). Consequently, concern has focused on the potential impact of sandeel fishing on seabird stocks (Monaghan, 1992; Furness and Tasker, 2000; Tasker *et al.*, 2000; Furness, 2002, 2003). On two occasions, in the Shetland Isles in the mid-1980s and off the Firth of Forth in SE Scotland in the mid-to-late 1990s, declines in seabird breeding success were linked to nearby sandeel fishing activity (Monaghan *et al.*, 1989; Hamer *et al.*, 1991, 1993; Furness, 1996; Wright, 1996; Harris and Wanless, 1997; Rindorf *et al.*, 2000). In both instances, a precautionary approach was adopted and the sandeel fishery in question was closed.

Such concerns, and the management response to them, highlight the necessity for monitoring changes in the abundance of sandeels in areas where marine predators might be at risk. First, there is the obvious need to monitor change in the abundance of sandeels, so that circumstances wherein the fishery may start to limit sandeel availability to predators can be recognized. Second, when fisheries are closed to alleviate such situations, the effectiveness of the management action will need to be monitored and assessed. Such information should be a prerequisite where management intent is ultimately to re-open the fishery following stock recovery. Traditionally, most information on the abundance of sandeels has originated from scientific monitoring of the fishery (Wright, 1996; Furness, 2002). Complete closures of specific sandeel fisheries will compromise the availability of such data, making monitoring of the effectiveness of a closure difficult. Furthermore, results of the analysis of fishery-derived data in the form of traditional stock assessments (ICES, 2002, 2004), or even catch per unit effort (cpue) data, are rarely available in real time. The first indications that a particular local sandeel stock might be in trouble may come after the event, by which time marine top predators in the region may already be having difficulties obtaining prey. Fishery-derived indices also suffer from the disadvantage of not being sampled randomly. This can lead to the maintenance of artificially high abundance indices if only high-density areas are sampled (Hilborn and Walters, 1992), and these are constantly replenished from more marginal habitats following some form of “ideal free” redistribution process (Fretwell and Lucas, 1970; Partridge, 1978). Such a situation has been postulated for sandeel fishing grounds off the Scottish east coast (Jensen *et al.*, 2001).

Assessing local sandeel population biomass is difficult because of the species’ capacity to move freely between the seabed sediments and the overlying water column. In fact, sandeels spend most of their time buried in the sediments (Reay, 1970; Winslade, 1974a, b, c; Pinto *et al.*, 1984), emerging briefly to spawn in mid-winter (Macer, 1966; Gauld and Hutcheon, 1990; Bergstad *et al.*, 2001), and to feed in large schools during part of the day over a more prolonged period in late spring and early summer (Macer, 1966; Reay, 1970; Winslade, 1974a, b, c; Freeman *et al.*, 2004). Sampling sandeels in the sediment at night, using grabs and dredge gears for example, therefore, presents the best opportunity for assessing the entire population in one medium at one time. However, at the time of the seabird breeding season, and at latitudes that include the major seabird breeding colonies, light levels drop below 20 lux for <5 h each night. At higher light levels, emergence activity is stimulated (Winslade, 1974b). This makes it difficult to carry out sufficient nocturnal survey work in the time available. Late autumn and early spring are more appropriate for nocturnal grab or dredge surveys, when the entire population should be in the sediment and the night-time periods are longer. However, those times are well outside both the seabird breeding and main fishing seasons, making real-time management of a fishery based on such assessment methods difficult.

Assessing sandeel abundance at around the time that the fishery might operate, and when seabirds are likely to be most dependent on the species, should provide a more informative basis for a broader ecosystem approach to the management of sandeel fisheries. However, this would tend to involve surveying outside the hours of darkness at a time of year when sandeels were moving frequently between the sediment and the water column. Assessment methods relying solely on "sampling" sandeels in the water column, for example acoustic survey techniques and commercial pelagic industrial trawling cpue, are therefore open to the criticism that they fail to sample the entire population. The proportion of the sandeel population active in the water column varies considerably in response to several influences, such as variation in water temperature, light level, and zooplankton abundance, and the risk from predation (Reay, 1970; Winslade, 1974a, b, c). To add to the difficulties involved, different components of the population, for example 0-group sandeels and older sandeels, may also spend different fractions of the available time feeding in the water column and buried in the sediment (Reeves, 1994; Kvist *et al.*, 2001; Bergstad *et al.*, 2002). In terms of specific growth rates, 0-group sandeels need to gain weight to a much greater extent than older sandeels during the feeding period, so as to achieve a minimum body mass sufficient to survive the ensuing winter (Winslade, 1974c). Distinguishing interannual variation in sandeel population biomass (of both age groups) from between-survey differences in the proportion of the sandeel population active in the water column, therefore, presents difficulties in interpreting the results of such surveys when carried out in isolation.

The timing and intensity of primary production, which provides food for the main zooplankton prey of sandeels (Covill, 1959; Macer, 1966; Reay, 1970; Meyer *et al.*, 1979; Monteleone and Peterson, 1986), must strongly influence the availability of food to sandeels, and consequently affect the amount of time sandeels spend foraging in the water column (Winslade, 1974a, c). Variability in both the timing and the intensity of the spring bloom in primary production at any given location in the North Sea is driven by the degree of mixing within the water column (Le Févre, 1986; Pingree *et al.*, 1975; Simpson, 1981). As water depth and tidal current speeds at any location are deterministic (Pingree *et al.*, 1978, Simpson and Bowers, 1981), any variation in water column mixing, and hence primary production, is due to interannual differences in local meteorological forcing. Based on an earlier model (Sharples, 1999), Sharples *et al.* (2006) developed a one-dimensional, coupled biophysical model that uses daily local meteorological data to determine seasonal variation in water column mixing and primary productivity in the Marr Bank and Wee Bankie region off Scotland, areas with notable sandeel concentrations. Annual variation in the timing of the spring bloom determined by this model is correlated with kittiwake breeding success at local seabird colonies (Scott *et al.*, 2006). Since kittiwakes in this area are heavily dependent on sandeels (Wanless *et al.*, 1998; Rindorf *et al.*, 2000; Lewis *et al.*, 2001), this relationship is presumably mediated through an effect of primary productivity on sandeel behaviour. This model could therefore provide the necessary additional information required to interpret interannual variation in, for example, acoustic survey data and thus enable variation in total population biomass to be distinguished from variation in the proportion of the population active in the water column.

The sandeel fishery off SE Scotland was closed in 2000. Here we present data collected on the main fishing grounds, the Wee Bankie, Marr Bank, and Berwick's Bank, over the period 1997—2003. Our analyses include the last three years of the fishery and the first four years of the closure. Our principal aim is to determine the effectiveness of the fishery closure as a means of restoring a potentially depleted sandeel stock. First we present the basic catch, effort, and cpue data reported to the Danish Institute of Fisheries Research (DIFRES) by Danish sandeel fishers in order to determine the extent to which the closure actually modified fishing activity in the area. A limited scientific fishery was permitted to enable cpue data to be collected for scientific monitoring of the sandeel population within the closed area. We then use three fishery-independent survey methods to assess the population biomass or abundance of sandeels within the study area. We explore the relationships between these fishery-independent biomass estimates and the fishery-based cpue. In the process, we illustrate the difficulties involved in interpreting the results of the different survey techniques, caused primarily by the species' behavioural characteristic of moving between the water column and seabed sediments, and the extent to which the different survey techniques detect sandeels either in the water column or in the sediment. To overcome these problems, we use the output from a one-dimensional coupled biophysical model to reconcile the differences between temporal trends derived from the four biomass assessment techniques. We then develop a simple statistical model that utilizes the data collected from two fishery-independent survey techniques to estimate first, the proportions of the sandeel population that are either active in the water column or buried in the seabed sediments, and second, the total or absolute biomass of sandeels in the study area in each year. The model results are validated with data collected by a third fishery-independent assessment method. This model could provide biomass assessment information in real time, allowing managers the option of regulating fisheries at critical times for predator populations.

Study area and methods

The study area covered most of two ICES statistical rectangles, 41E7 and 41E8, between 56°0′N and 56°30′N, and longitudes 03°0′W and 01°0′W (Figure 1). Fishery-dependent catch and effort data are reported for both rectangles. Fishery-independent acoustic, demersal trawl, and nocturnal grab surveys were undertaken using the Scottish FRV "Clupea". All acoustic and demersal trawl survey work was carried out between 03:00 and 15:00 GMT, between late May and early July of each year from 1997 to 2003 (Table 1), and grab-survey work was undertaken between 20:00 and 04:00 GMT in October of each year from 1998 to 2003. The acoustic and demersal trawl surveys followed immediately one after the other, with one day in between to change fishing gears. The order of the surveys varied between years with the objective of minimizing variation in the timing of the acoustic survey. Because the "Clupea" could not operate throughout an entire 24-h period, it was necessary to anchor close inshore for at least 10 h each day. As a result, it was not possible to cover the two ICES rectangles completely, so an area down the eastern edge of ICES rectangle 41E8 had to be excluded. However, the area that was covered by the two surveys (east to longitude 01°30′W in the north, and to 01°10′W in the south) included the main sandbanks, the Wee Bankie, Marr Bank, and Berwick's Bank, where most sandeel fishing activity in these two ICES rectangles took place (Figure 1). The 4720-km^2 area covered by the acoustic survey included the section of water inside the Firth of Forth. This area, in the middle of a busy shipping lane, was inappropriate for demersal trawling, so was excluded from the demersal trawl area. Few sandeels were observed there during the acoustic surveys, and no commercial sandeel fishing took place there. The area covered by the demersal trawl survey was therefore bounded to the west at longitude 02°40′W and amounted to approximately 4529 km^2.

Acoustic survey

The study area was divided into 50 rectangles of 5′ latitude by 10′ longitude (approximately 9.27 × 10.30 km). Transects were steamed through the centres of these rectangles in an east−west direction (Figure 1), approximately perpendicular to the general orientation of the coastline, so tending to run across depth contours rather than parallel to them (MacLennan and Simmonds, 1991). Acoustic data were integrated over 5-min periods of passage along each transect, and 6−8 "samples" of 5 min were collected in each

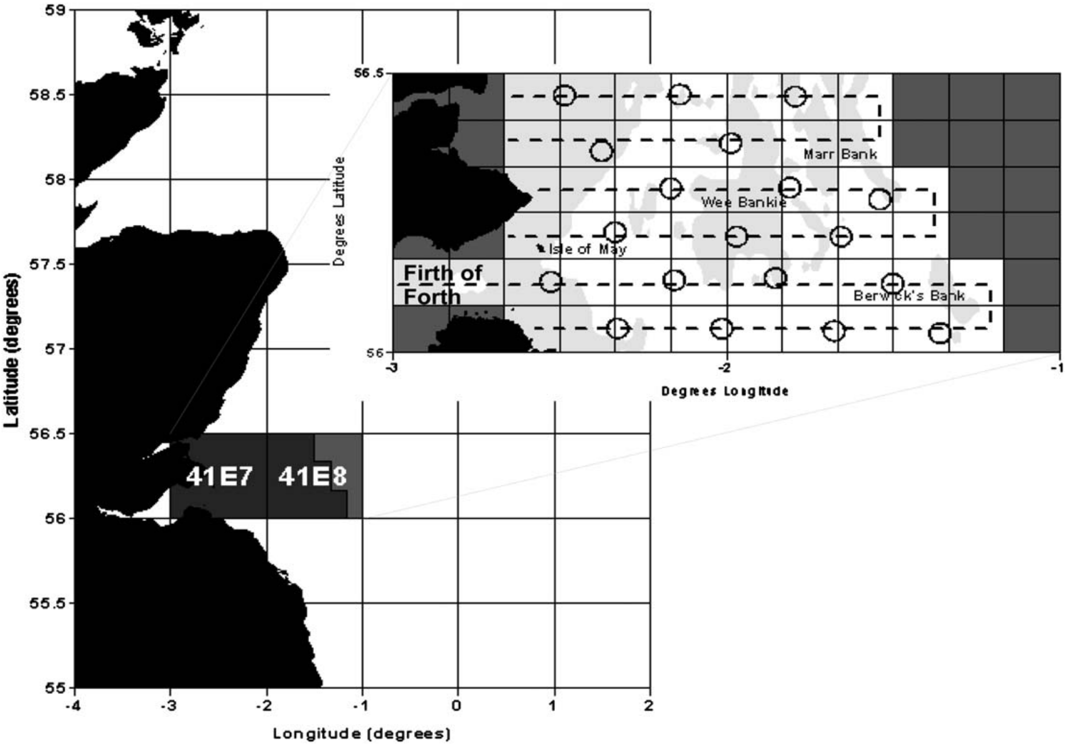

Figure 1. The location of the two ICES statistical rectangles that constituted the study area off the east coast of Scotland. The expanded section shows the division of the area into 50 rectangles of 5′ latitude by 10′ longitude, six main acoustic survey transects, and 19 demersal trawl stations. Light shading indicates regions of 50-m depth or less, and the locations of the main sandbanks are depicted. Heavy shading delimits the area included in the acoustic survey and demersal trawl biomass estimates.

Table 1. Dates and median Julian day when acoustic and demersal trawl surveys were carried out each year. The actual area covered by the acoustic survey in each year, and the raising factor required to equate the acoustic biomass estimates to that expected had the area covered in 1997 been covered in each subsequent year are also provided.

Year	Acoustic survey dates	Median Julian day (AS)	Area surveyed (km^2)	Raising factor	Demersal trawl survey dates	Median Julian day (DT)
1997	21–25 June	174	4721	1.0000	27 June–3 July	180
1998	13–16 June	165	3392	1.3918	18–22 June	171
1999	2–5 July	184	4645	1.0164	7–11 July	190
2000	14–19 June	167	4612	1.0236	20–24 June	173
2001	4–9 June	157	4671	1.0107	30 May–3 June	152
2002	15–19 June	168	4529	1.0424	9–13 June	162
2003	4–17 June	166	4529	1.0424	6–11 June	159

rectangle. Major concentrations of pelagic fish encountered during the course of each acoustic survey were sampled using an International Young Gadoid pelagic trawl fitted with a 6-mm codend (Figure 2A). The samples obtained were used to confirm species composition, to determine the size and age composition of the fish, and to determine sandeel length–weight relationships for each year. A 38-kHz split-beam transducer and a 120-kHz single-beam transducer were mounted in a towed body deployed forward of the propeller from a boom mounted near the bow of the vessel. The body, towed at a nominal speed of 18 km h^{-1} approximately 5 m below the sea surface, provided a more stable platform in rough weather and avoided the problems of interference from air bubble formation under the hull often associated with hull-mounted transducers. Only data from the 38-kHz transducer were used in the biomass estimation process. Although not ideal for sandeel discrimination, 38 kHz is certainly adequate, and the only sandeel target strength data available were obtained at this frequency (Armstrong, 1986). Sandeels provide a better acoustic return at higher frequencies, so data collected from the 120-kHz transducer were used to aid identification and discrimination of their shoals. Where there was doubt regarding the identity of the fish in a particular mark, the species composition in the appropriate pelagic trawl sample was used to attribute the integral values to species. Integral values for herring and sprat were assigned *pro rata* with their relative proportions (by weight) in the catch. This approach was considered inappropriate for assigning sandeel integrals because of their likely low catchability in the fishing gear compared with clupeids. Instead, if sandeels were caught in the trawl along with clupeids, both the 38-kHz and 120-kHz echo traces were re-examined to identify the marks most likely to consist of sandeels.

The 38-kHz transducer and echosounder were calibrated using a tungsten carbide sphere of known target strength suspended in the centre of the acoustic axis. The available target strength value for sandeels (TS, db kg^{-1} = −50) is independent of fish length; 1 kg of sandeels is assumed therefore to have a target strength of −50 db at 38 kHz no matter what size the fish are (Armstrong, 1986). Once surveyed, each transect's acoustic record was examined and the integral values obtained from the 38-kHz transducer for each 5-min run period were assigned to species. Knowing the target strength and identity of fish targets in the swath of water column ensonified by the acoustic pulse, the density (g m^{-2}) of sandeels present could be determined. For each rectangle of 5′ latitude by 10′ longitude, 6–8 density estimates, each the result of 5 min of survey, were obtained, and an overall estimate of the mean density of sandeels in the rectangle could be derived. Raising these by the area of sea in each rectangle provided estimates of sandeel biomass in each rectangle. Summing the results over all the rectangles, provided estimates of the total biomass of sandeels in the surveyed area during each cruise. In June 1998, poor weather conditions prevented the two most northerly transects from being surveyed. In 2000 and 2001, additional part-transects were steamed between the main transects over the major sandbanks or around the major seabird colony on the Isle of May (Figure 2A). Differences in the area actually covered by each survey needed to be taken into account in order to compare biomass estimates between years. The survey in 1997 covered the largest area. In subsequent years raising factors (RF_X) were determined to adjust the biomass estimate obtained in each year (x): $RF_X = Area_{1997}/Area_X$, where $Area_{1997}$ is the area covered in 1997 and $Area_X$ is the area covered in each of the other years (Table 1). Implicit in this is the assumption that the average density of sandeels in the area covered in each survey equalled the average density in the parts of the study area not covered. Trawl length frequency distribution data and survey weight-at-length relationships and age–length keys were used to break these biomass estimates down into particular size and age categories.

Acoustic survey techniques only detect sound returned from sandeel targets that are in the water column. Most of the sonar pulse is returned by the seabed, and the echosounder is incapable of quantifying sandeel biomass buried in the seabed. Consequently, biomass estimates obtained from the acoustic surveys were considered to be estimates only of the biomass of sandeels active in the water column.

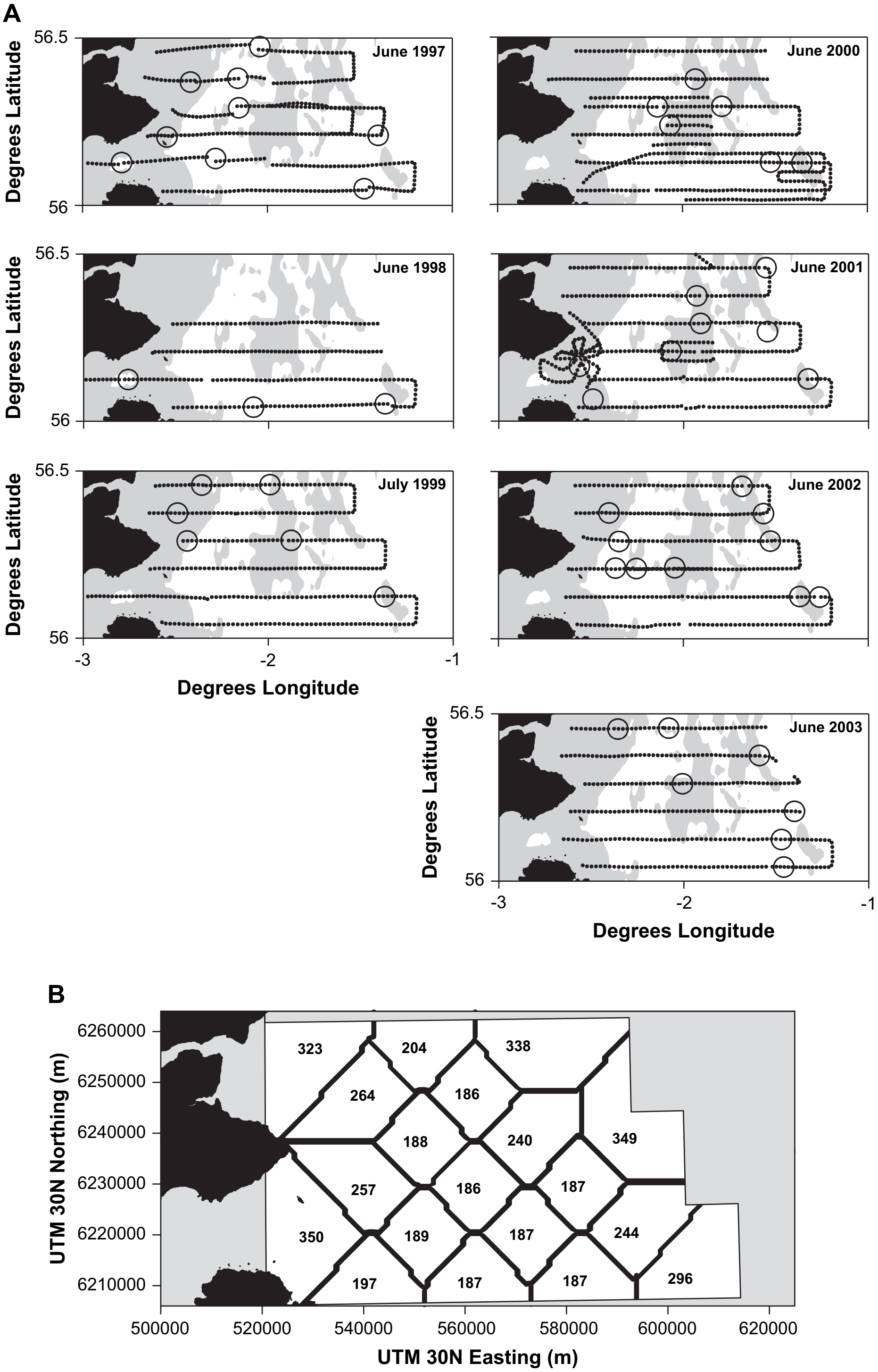

Figure 2. A. The actual acoustic survey track coverage in each year and the locations of the pelagic trawl sampling. B. The sea area associated with each demersal trawl station derived by “nearest neighbour” tessellation.

Demersal trawl survey

A Jackson rockhopper demersal trawl with a codend of 10-mm mesh was towed for 30 min at a speed of approximately 4 km h^{-1} at each of 19 evenly spaced sample stations (Figure 1). Net geometry monitoring equipment (SCANMAR, Norway) recorded the width and height of the trawl opening every 30 s. The ship's position, determined by Differential Global Positioning System (DGPS), was recorded simultaneously, so for each trawl sample, the area of seabed swept by the gear could be calculated. The total catch of sandeels in each trawl sample was quantified (number caught per 0.5-cm size class). Length-stratified subsamples were weighed to determine weight-at-length relationships and otoliths were extracted to determine age−length keys for each cruise. These were used to convert sandeel numbers at length to sandeel weight at length in each trawl. Dividing the number at length and weight at length of fish in each catch by the area of seabed swept by the trawl on each occasion converted these to density-at-length estimates (number km^{-2}, and kg km^{-2}). Multiplying the density-at-length estimates by the area of seabed associated with each trawl station determined by simple nearest-neighbour interpolation, or tessellation (Figure 2B), provided estimates of the total number and weight of sandeels, of each 0.5-cm length class, in each trawl station subarea. Summing these subarea population-at-length estimates across all trawl station subareas provided estimates of the total numbers and biomass at length of sandeels in the whole study area. Application of the age−length keys determined for each year enabled these estimates of total numbers and biomass at length to be assigned to the required age categories.

The low headline height of the demersal trawl (3.31 ± 0.06 m, $n = 133$) meant that only a small fraction of the water column was sampled on each deployment. However, the heavy contact of the ground gear on the seabed was likely to drive sandeels out of the sediment and into the path of the trawl (Meyer *et al.*, 1979; Hain *et al.*, 1995). Sandeel catches in heavy ground gear demersal trawls are higher at night when sandeels are buried in the sediment than by day when they are more active in the water column (Temming *et al.*, 2004). Consequently, the demersal trawl survey was considered primarily to provide an index of the biomass of sandeels in the sediment.

Nocturnal grab survey

A stratified random design was used for the nocturnal grab survey, in which sampling effort was disproportionately directed towards the sandy, silt-free habitats preferred by sandeels compared with the relative amount of such habitat within the study area (Holland *et al.*, 2005). Between 137 and 195 stations were sampled each year. At each station, a day grab sampling an area of 0.096 m^2 was deployed up to three times in order to obtain a single valid sample. Grab samples that contained less than 8-cm depth of sediment within the jaws were considered to be invalid in terms of estimating sandeel density. From each grab, sediment samples were extracted to determine whether the samples had been collected from suitable sediment types, insofar as sandeel habitat preferences were concerned. A detailed analysis of the relationship between sandeel density and size and sediment character revealed eight classes of sediment habitat. One of these habitat classes was considered to be unsuitable habitat for sandeels, and of the remaining seven habitat types, four were clearly preferred by sandeels (Holland *et al.*, 2005). Here we present sandeel density data by age for these four habitat types only. Sandeel density was determined by passing the sediment collected in each grab sample through a 5-mm mesh sieve to extract all sandeels. These were then counted, measured (to 0.5 cm below), weighed (to 0.1 g), and had their otoliths removed for age determination. Grab catches could therefore be quantified by both numbers and weight at both length and age. For further details of the grab-survey methodology and discrimination of the different habitat classes, see Holland *et al.* (2005).

The sandeel grab density data provided an index of the abundance of sandeels buried in the sediment at times of year and day when the entire population was likely to be buried in the sediment.

Cpue data

Off the Firth of Forth, sandeels are only fished by day from spring to early summer. Although the commercial industrial fishery in the area was closed from 2000 onwards, a limited experimental fishery was operated to allow the collection of the scientific data required by the Danish Institute of Fisheries Research (DIFRES) to monitor the sandeel population in the area. The cpue for a boat or a fleet is calculated by dividing the size of the catch, in weight or number of fish, by the amount of effort required to take the catch. Cpue is assumed to vary in proportion to variation in the abundance of the targeted fish population (King, 1995). For comparison with the fishery-independent survey data, the cpue data used in this study were extracted from logbook data for the Danish sandeel fishery operating on the Wee Bankie, Marr Bank, and Berwick's Bank (i.e. ICES statistical rectangles 41E7 and 41E8; Figure 1) over the period 1997−2003. The logbooks contain information about sandeel catches and fishing effort at the level of a fishing trip, in terms of catch weight and the number of days in which fishing took place within each ICES rectangle (30 × 30 nautical miles) visited. For each fishing trip, the trip median Julian day was determined from the Julian date of departure and the Julian landing date. Julian weeks were defined such that 1 January in each year (Julian day 1) was the first day of Julian week 1. Therefore, simply dividing each trip median Julian day by seven and rounding up allowed the trip to be assigned to a particular Julian week. The catch and effort information for all trips was in this way assigned to specific Julian weeks.

The pattern of fishing activity within the Danish sandeel fleet, in terms of vessel size, has changed over the years; the number of smaller vessels has decreased and the number of larger vessels has increased. Standardization of effort was required to compare catches and effort across all vessel size categories. Information about vessel size in the logbook data was provided in gross tonnage (GT) intervals of 10 GT for vessels up to 50 GT, and in 25-GT intervals for vessels >50 GT. Fishing effort for all vessels of any given gross tonnage ($\text{effort}_{\text{GT}}$) was therefore standardized to a 200-GT vessel equivalent (effort_{200}) using the equation

$$\text{effort}_{200} = \text{effort}_{\text{GT}} \frac{\text{GT}^b}{200^b},$$

where $b = 0.45$ and GT is the gross tonnage of the vessel concerned. Parameter b was estimated from:

$$\log(\text{cpue}_y) = a_y + b \log(\text{GT}),$$

where cpue denotes mean catch per unit effort (t d^{-1} fishing) for a trip, a_y denotes sandeel abundance in year y, and b denotes the standardization exponent for all years. For further details, see STECF (2004, 2005).

Preliminary examination of the Danish logbook data revealed that over the years 1997–2003, the main fishing activity in the study area was in Julian weeks 20–26 (20 May–7 July). To illustrate annual variation in fishing activity, catch and effort data were extracted for all trips with trip median Julian days falling within this 7-week period, and an annual cpue index was calculated. For a more direct comparison of the cpue index with the combined acoustic and demersal trawl survey estimates, catch and effort data were extracted for fishing trips with a median Julian day greater than or equal to the median Julian day of the first of these two surveys, and less than or equal to the median Julian day of the second of the two surveys (see Table 1), and again annual cpue indices were calculated.

The industrial fishery for sandeels uses large pelagic trawls that are prevented as far as possible from contacting the seabed for fear of sustaining gear damage. As a result, the fishery is unlikely to "sample" sandeels buried in the sediment. Commercial sandeel catches in the area consisted almost entirely of sandeels aged 1+ years (DIFRES, unpublished information). Cpue was therefore considered to provide an index only of the biomass of 1+ sandeels active in the water column.

1-D coupled biophysical model

Coupled biological and physical oceanographic modelling has advanced sufficiently in recent years so as to accurately capture important dynamics, such as the area-specific quantity of primary production, at temporal and spatial scales appropriate to the feeding behaviour of individual fish (Franks, 1992; Sharples, 1999; Waniek, 2003). These types of models, in particular the 1-D coupled biophysical model of Sharples (1999) and Sharples *et al.* (2006), can reproduce biologically important details of the vertical structure of the water column and the daily primary production, represented as chlorophyll concentrations for a given location. Biological information gained from this model, such as the annual timing of the spring bloom, has improved our understanding of local top predator population dynamics (Scott *et al.*, 2006). The physical component of the model, driven by local values for tidal forcing, surface heating, and surface winds, calculates for each day the vertical structure of currents, temperature, and light through the water column for the years in which the appropriate meteorological data are available. A turbulence closure scheme (Canuto *et al.*, 2001) is used to calculate the rates of turbulent mixing driven by tidal and wind stresses. The biological component calculates the response, in terms of chlorophyll concentration, of a single phytoplankton species to the light and nutrient environment, with the turbulent mixing controlling the vertical fluxes of phytoplankton and dissolved inorganic nutrients.

The tidal current data needed to parameterize the 1-D biophysical model for the study area were obtained from a hydrographic mooring located within the area at 56°15′N, 01°15′W, and a water depth of 65 m. The mooring provided information, at 10-min resolution, on the changes in vertical structure (at 5–10-m intervals), such that it was possible to define the depth of the surface mixed layer and the strength of the thermocline at any point in time. The mooring had two current meters, one fluorometer, and eight mini-loggers (temperature recorders), and operated from March to October of both 2001 and 2002. The temperature and fluorometry data obtained from the mooring were used to validate the model (Sharples *et al.*, 2006). The daily meteorological data, the daily mean values for solar irradiance, windspeed and direction, humidity, air temperature, and air pressure, needed to run the model for thermally stratified regions within the study area were collected at the Leuchars and Mylnefield Meteorological Stations in SE Scotland. These were obtained from the British Atmospheric Data Centre (BADC).

Results

Trends in fishing activity

As expected, catches in 1997 and 1998 were considerably larger than in any year during the closure period. However, in 1999 the catch was much less, despite the fishery being officially open (Figure 3). In fact, the scientific catch in 2001, when the fishery was closed, actually exceeded that taken in 1999. Given the small sample size, all data during the fishing period had to rank either higher or lower than all data in the closure period in order to obtain a significant Mann–Whitney test result. Simply considering the ranked annual landings in this way failed to produce a significant Mann–Whitney test comparing fishery years with closure years (Mann–Whitney $U = 1$, $p = 0.08$). Fishing effort

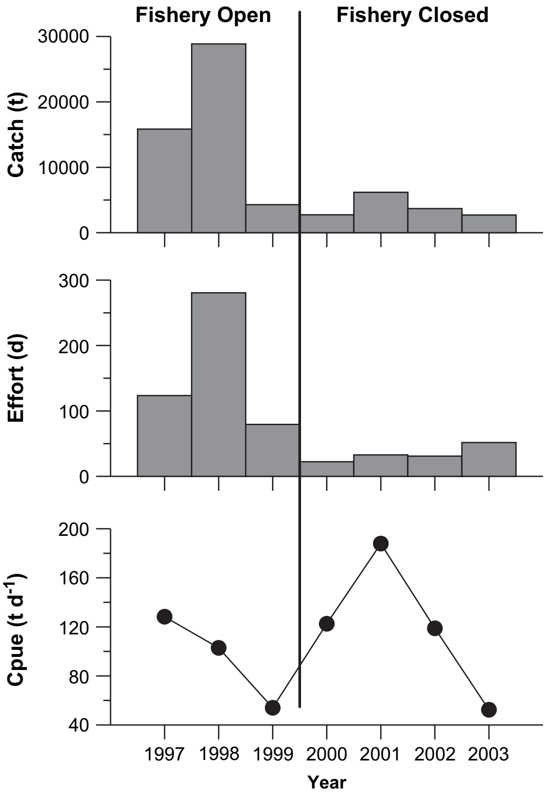

Figure 3. Annual variation in the Danish sandeel fishery, catch, effort, and cpue data for ICES statistical rectangles 41E7 and 41E8, determined for the period including Julian weeks 20−26.

was also lower in 1999 than in the two preceding years (Figure 3), but this was almost certainly a case of fishers responding to the unusually low cpue by stopping fishing. Fishing effort in all three fishery years was higher than in all four of the closure years (Mann−Whitney $U = 0$, $p = 0.03$), so closing the fishery certainly reduced fishing effort in the area. There was no indication from the cpue data to suggest that sandeel biomass in the area was any higher during the period of the moratorium than during the earlier period when the fishery was active (Figure 3; Mann−Whitney $U = 5$, $p = 0.72$).

Individual biomass index trends

Variation in the biomass/abundance of both 0-group and 1+ sandeels indicated by each assessment method is shown in Figure 4. None of the Mann−Whitney tests comparing fishery years with closure years were statistically significant. Even so, some suggestion of a recovery in the biomass of 1+ sandeels following the fishery's closure was apparent in the three fishery-independent survey trends. For both the demersal trawl and acoustic surveys, the lowest biomass estimate obtained during the closure years was lower than the highest biomass estimate obtained during the fishery years, sufficient to give a non-significant Mann−Whitney test result ($U = 1$, $p = 0.08$ for both indices). However, in both instances this was only just the case (Figure 4). The grab-survey density estimates obtained in 1998 and 1999 were lower than all four estimates obtained during the years of the closure, but the reduction in the sample size (no autumn grab survey was carried out in 1997) reduced statistical power to the point where even this result was not statistically significant ($U = 0$, $p = 0.06$). On the other hand, the cpue index calculated for the same period as the acoustic/demersal trawl surveys showed no indication of any major effect of the fishery closure on 1+ sandeel biomass ($U = 5, p = 0.72$). None of the three fishery-independent assessment methods indicated any effect of the fishery closure on 0-group sandeel biomass in the area.

Otherwise, consistency between the behaviour of the different index values was low; none of the correlation comparisons between the three fishery-independent assessment method abundance estimates of either 0-group or 1+ sandeels were statistically significant (Table 2). Only the two indices of 1+ sandeel biomass in the water column, acoustic survey and cpue calculated for the period coinciding with the acoustic/demersal trawl surveys, revealed temporal trends that were significantly correlated (Figure 4, Table 2; $r^2 = 0.74$, $p = 0.013$). Trends in cpue determined for the whole period (Figure 3) and for the period immediately coinciding with the combined acoustic/demersal trawl surveys (Figure 4) were similar ($r^2 = 0.77$, $p < 0.01$), but of course these two indices were not independent of each other. The acoustic survey biomass estimate was actually significantly correlated with both cpue indices, although the relationship was closer for cpue calculated for the period coinciding with the surveys ($r^2 = 0.74$, $p = 0.013$) than for the whole fishing season ($r^2 = 0.66$, $p < 0.03$).

Primary productivity passing through the food chain fuels the food supply to sandeels. Variation in the chlorophyll concentration in the water column, therefore, provides an indication of the "feeding opportunity" for sandeels. The 1-D coupled biophysical model was used to estimate the daily total water column chlorophyll concentration in each year (Figure 5). Integrating these curves up to the date of the combined acoustic demersal trawl survey in each year, i.e. the Cumulative water Column Chlorophyll Concentration (CCCC), provided an indication of the total "feeding opportunity" available to sandeels prior to each survey (Table 3). Timing of the combined acoustic/demersal trawl surveys in each year varied by 32 days, while the start date of the spring bloom varied by 16 days. As a result, the timing of the combined acoustic/demersal trawl survey relative to the start of the spring bloom varied by a factor of two, from as little as 38 days later in 2001 to as long as 76 days later in 1999 (Table 3). Consequently, the "feeding opportunity" available to sandeels prior to each survey, indicated by CCCC, varied by a factor of 1.6; from 9709 to 15 508 mg d m^{-2}. Perhaps a certain growth target or body condition level acts as a trigger, causing sandeels to cease feeding in the water column and enter the

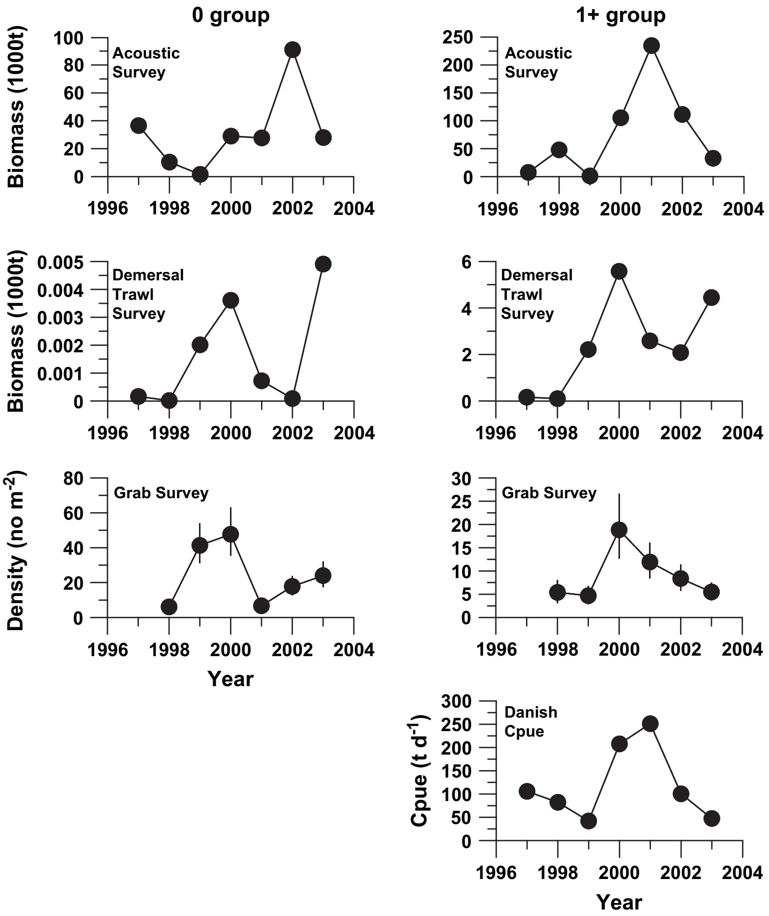

Figure 4. Between-year variation in the biomass/abundance of 0-group and 1+ sandeels indicated by three fishery-independent assessment methods (acoustic survey, demersal trawl survey, and autumn nocturnal grab survey) and fishery-dependent cpue data determined for the period immediately coinciding with the timing of the combined acoustic and demersal trawl surveys.

over-wintering phase of their life cycle buried in sediment (Winslade, 1974c). Then, such a variation in "feeding opportunity" could have led to considerable variation in the proportions of sandeels active in the water column or buried in the sediment at the times of the combined acoustic/demersal trawl surveys. Given that the different fishery-independent assessment methods sample sandeels in the water column and in the sediment to differing extents, the lack of significant correlations between them is therefore entirely to be expected.

As acoustic surveys only detect sandeels active in the water column, whereas demersal trawl surveys primarily sample sandeels buried in the sediment, these two biomass assessment methods are complementary. By taking account of sandeel emergence behaviour and reconciling the differences between the two indices, they can be combined to provide an index of total sandeel biomass in the area. The grab sampling carried out at night in autumn should have sampled the population at a time when all sandeels should have been buried in the sediment. This survey, therefore, also provides an index of total population abundance that can be used to verify the combined acoustic and demersal trawl survey estimate.

Development of a sandeel biomass model

1+ sandeels

The total biomass B_{TOT} of sandeels at a given time is the sum of the biomass in the water column (B_{WC}) and the biomass buried in the sediment (B_{SED}). The acoustic survey provides an estimate of the absolute biomass of sandeels in the water column (B_{AS}), so that $B_{\mathrm{AS}} = B_{\mathrm{WC}}$. We assume, as with any trawl survey or cpue index, that the demersal trawl survey index (B_{DT}) varies proportionally with variation in the biomass of sandeels in the sediment, so that $B_{\mathrm{DT}} = q_{\mathrm{DT}} B_{\mathrm{SED}}$, where q_{DT} is the catchability coefficient of the demersal trawl for sandeels (King, 1995). Thus,

$$B_{\mathrm{TOT}} = B_{\mathrm{AS}} + B_{\mathrm{DT}} q_{\mathrm{DT}}^{-1}. \qquad (1)$$

From this it follows that at the time of each survey, the proportion of the total sandeel biomass buried in the sediment (P_{SED}) may be expressed as

Table 2. Results of correlation analyses comparing the temporal trends for each sandeel biomass/abundance index. Significant correlations are emboldened.

Age class	Biomass index comparison	r^2	p
0	Acoustic survey vs. demersal trawl survey	−0.076	0.550
	Acoustic survey vs. autumn nocturnal grab survey	−0.034	0.728
	Demersal trawl survey vs. autumn nocturnal grab survey	0.381	0.192
1+	Acoustic survey vs. demersal trawl survey	0.065	0.581
	Acoustic survey vs. autumn nocturnal grab survey	0.326	0.237
	Demersal trawl survey vs. autumn nocturnal grab survey	0.406	0.174
	Danish cpue vs. demersal trawl survey	−0.092	0.508
	Danish cpue vs. autumn nocturnal grab survey	0.397	0.180
	Danish cpue vs. acoustic survey	**0.738**	**0.013**

$$P_{\mathrm{SED}} = \frac{B_{\mathrm{DT}} q_{\mathrm{DT}}^{-1}}{B_{\mathrm{AS}} + B_{\mathrm{DT}} q_{\mathrm{DT}}^{-1}}.$$

We posit that optimal sandeel behaviour requires 1+ sandeels to emerge to feed in the water column during a period when foraging is most profitable. However, while active in the water column, sandeels experience higher rates of mortality, principally through raised predation risk. The optimum life history strategy is therefore to keep the feeding period as short as possible to minimize mortality, but long enough to acquire sufficient energy reserves to reproduce and survive the ensuing non-feeding winter period. Therefore, as the feeding period progresses, with increasing cumulative opportunity to feed, an increasing number of sandeels will attain the body condition level required, terminate their feeding activity, and bury themselves in the sediment. As a proxy for this sandeel feeding opportunity we use the CCCC (C in our model), and indeed a significant fraction of between-year variation in the acoustic survey 1+ sandeel biomass was related to variation in CCCC over the period preceding each survey (Figure 6A). The logistic equation describes just such a gradual increase in the proportion P_{SED} of inactive sandeels in the sediment,

$$P_{\mathrm{SED}} = \frac{1}{1 + \exp(ab - bC)},$$

where a is the point of equal proportions in the sediment and the water ($P_{\mathrm{SED}} = 0.5$), and b is the rate of change from active to buried behaviour in the population equal to the slope of the regression of the logit transformed proportions vs. CCCC: $\ln(P_{\mathrm{SED}}/1 - P_{\mathrm{SED}}) = i + bC$ (logit transformations linearize the logistic equation and normalize errors). We therefore assume the following relationship:

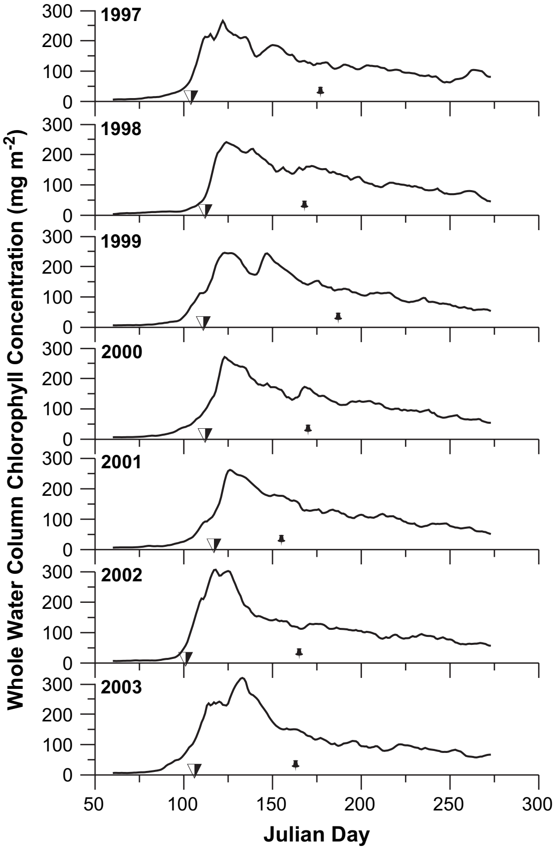

Figure 5. Daily variation in whole water column chlorophyll concentration throughout each of the years 1997−2003. The start of the spring bloom and the date of the combined acoustic and demersal trawl surveys in each year are indicated by inverted triangles and pin symbols, respectively.

$$\frac{B_{\mathrm{DT}} q_{\mathrm{DT}}^{-1}}{B_{\mathrm{DT}} q_{\mathrm{DT}}^{-1} + B_{\mathrm{AS}}} = \frac{1}{1 + \exp(ab - bC)}. \tag{2}$$

By rearranging and log-transforming Equation (2), we obtain the relationship

$$\ln\left(B_{\mathrm{DT}y}\right) - \ln\left(B_{\mathrm{AS}y}\right) = \ln(q_{\mathrm{DT}}) - ab + bC_y + \varepsilon_y.$$

Letting $\lambda = \ln(q_{\mathrm{DT}}) - \gamma$, where $\gamma = ab$, and $Y_y = \ln\left(B_{\mathrm{DT}y}\right) - \ln\left(B_{\mathrm{AS}y}\right)$, b and λ may be estimated by the linear regression of $Y_y = \lambda + bC_y$ for positive values of $B_{\mathrm{DT}y}$ and $B_{\mathrm{AS}y}$ (Figure 6B).

Given the estimate of $b = 0.0008919$, then a and $\ln(q_{\mathrm{DT}})$ remain linked within the derived value for $\lambda = -14.6356$. The range of reasonable values for q_{DT} is limited because a very small value would imply exceptionally poor catchability combined with no influence of the chlorophyll

Table 3. Julian dates of the combined acoustic/demersal trawl surveys, i.e. the Julian date of the day in port between the two surveys, and the start of the spring bloom, which is defined as the first day in which surface water chlorophyll concentration exceeded 2 mg m^{-3} for five consecutive days (after Scott *et al.*, 2006). The time span between these two dates is also shown together with the cumulative daily, whole water column, chlorophyll concentration over the part of each year preceding each annual combined survey.

Year	Julian day of combined acoustic/ demersal trawl survey	Julian day of start of spring bloom	Number of days between start of bloom and combined surveys	Cumulative daily whole water column chlorophyll concentration (mg d m^{-2})
1997	177	104	73	13 866
1998	168	112	56	10 959
1999	187	111	76	15 508
2000	170	112	58	12 412
2001	155	117	38	9 709
2002	165	101	64	12 948
2003	163	106	57	13 906

signal, whereas $q_{\mathrm{DT}} > 1$ indicates a concentration effect of the trawl. We considered a realistic interval, restricting catchability between $q_{\mathrm{DT}} > 0.0005$ and $q_{\mathrm{DT}} \leq 1$. The corresponding interval for a is then approximately 8000−16 500 (CCCC values). To find the most likely value of a within this interval that would also fit with other available information on the 1+ sandeel population, we used a linear correlation analysis to compare the total biomass of age 1+ sandeels estimated from the combined acoustic and demersal trawl surveys ($B_{\mathrm{TOT}y}$) with the grab-survey index ($B_{\mathrm{G}y}$), where $B_{\mathrm{TOT}y} = B_{\mathrm{DT}y} q_{\mathrm{DT}}^{-1} + B_{\mathrm{AS}y}$, allowing q_{DT} to vary with variation in a over the range of CCCC values 8000−16 000 mg d m^{-2}, according to the relationship $q_{\mathrm{DT}} = \exp(\lambda + ab)$. The curve describing the correlation coefficient for the linear relationship between $B_{\mathrm{TOT}y}$ and $B_{\mathrm{G}y}$ was dome-shaped over this range of values of a, with a maximum $r^2 = 0.674$ ($p = 0.045$) at an intermediate value of $a = 12\,045$ (Figure 6C). Figure 6D shows the actual relationship between B_{TOT} and B_{G} obtained using a value of a of 12 045. This optimal solution for a gives an estimate of the catchability of 1+ sandeels in the demersal trawl of $q_{\mathrm{DT}} = 0.0204$. Applying these optimal solutions for q_{DT} and a first to Equation (2) and then to Equation (1) allows variation in the proportion of 1+ sandeels buried in the sediment with increasing CCCC to be modelled (Figure 6E), and provides an estimate of total 1+ sandeel population biomass in each year as the sum of the fractions active in the water column and buried in the sediment (Figure 6F). Error bars in Figure 6F indicate the range of 1+ sandeel biomass predicted by the model over the range of values of a that gave a statistically significant r^2 in Figure 6C. The sandeel biomass model indicated that 1+ sandeel biomass in all the four years that the fishery was closed exceeded the biomass present in the area during the three years that the fishery was commercially active (Mann−Whitney $U = 0$, $p = 0.03$).

0-group sandeels

A similar approach was adopted to model the biomass of 0-group sandeels based on the combined acoustic survey and demersal trawl 0-group sandeel biomass indices such that Equations (1) and (2) were again considered to represent the situation. However, important differences in the biology of 1+ and 0-group sandeels necessitated some modifications to certain aspects of the model and caused some problems with parameter estimation. First, 0-group sandeels metamorphose from the larval stage in late May (Wright and Bailey, 1996); here, we assume a metamorphosis date of Julian day 145. Prior to this date, zooplankton production fuelled by primary productivity, as indicated by CCCC, would not be utilized by post-metamorphosis 0-group sandeels. Consequently we recalculated a CCCC value more appropriate to 0-group sandeels, the Cumulative total water Column Chlorophyll Concentration from Julian day 145 onwards (CCCC_{145}). Second, the length of 0-group sandeels immediately following metamorphosis is approximately 4.5 cm (Wright and Bailey, 1996). In the Firth of Forth, 0-group sandeels appear to need to attain a length of 7.5−8.0 cm before burying in the sediment (Holland *et al.*, 2005; GJH and SPRG, unpublished data), which at published growth rates would require 80 days or more (Wright and Bailey, 1996) leading to burial at around Julian day 225, i.e. mid-August. It was likely therefore that a large proportion of the 0-group sandeel population would have remained active in the water column at the time that the combined acoustic and demersal surveys were carried out in all the years that our study was conducted (Figure 5, Table 3). The lack of a significant relationship between the acoustic survey index and CCCC_{145} tended to confirm this (Figure 7A). The polynomial fit, excluding the 2002 datum, driven by the low value acoustic biomass estimate obtained in 1999 when the survey was carried out latest in the year and at its highest CCCC_{145} value, suggests that 0-group sandeels remained active in the water column until CCCC_{145} exceeded at least 5000 mg d m^{-2}. This caused problems in estimating both b and a_{145} parameters, because many of the data points lay in the region where $P_{\mathrm{SED}} \approx 0$. Therefore, although regression of $Y_y = \lambda + bC_y$ provided estimates of $b = 0.0007921$ and $\lambda = -13.808412$, the regression was not statistically significant (Figure 7B). Moreover, a zero value demersal trawl index in 1998 provided an additional problem, because positive values were required from both indices in order to perform the analysis. Excluding that year's data from the analysis would have meant also discarding the perfectly valid 1998 acoustic survey information. To avoid this, we arbitrarily assigned a value of 0.000016 as the 1998 demersal trawl index value,

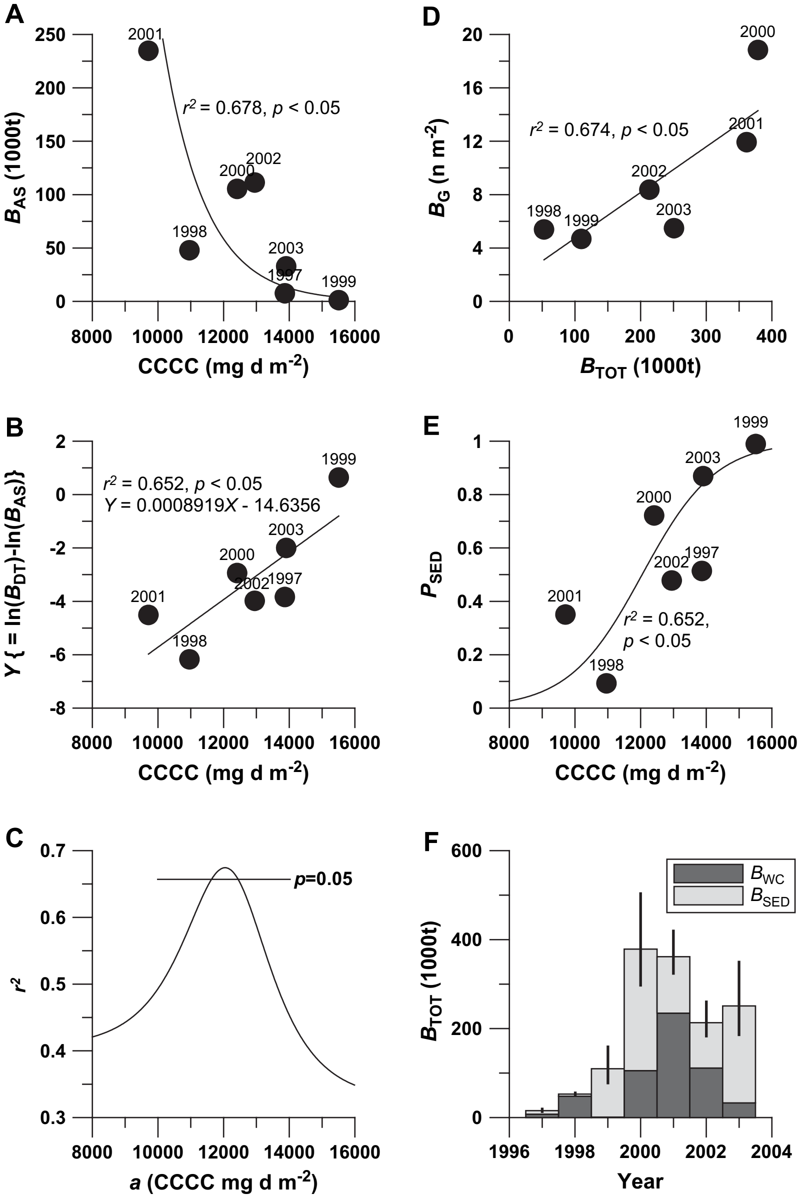

Figure 6. 1+ sandeel model. A. The relationship between the acoustic survey biomass estimate and CCCC. B. Linear regression of CCCC on Y to estimate b and λ parameters, where $Y_y = \ln(B_{\mathrm{DT}y}) - \ln(B_{\mathrm{AS}y})$, and b and λ are the slope and constant, respectively, of the regression analysis. C. The relationship between r^2 values of the correlation between the modelled estimate of 1+ sandeel biomass and the autumn grab-survey estimate of 1+ sandeel density as the value of the a parameter was varied. D. The best fit correlation between the modelled estimate of 1+ sandeel biomass and the autumn grab-survey estimate of 1+ sandeel density obtained at $a = 12\,045$. E. The modelled variation in the proportion of 1+ sandeels buried in the sediment with increase in CCCC, showing fit to the data. F. The annual variation in the modelled estimate of 1+ sandeel biomass, both in the water column and in the sediment. Bars indicate the range of modelled biomass in the sediment for the range of a values that gave a significant r^2 in panel C.

i.e. one-hundredth of the mean of all seven demersal trawl, 0-group sandeel, biomass indices.

Because in most years, 0-group sandeels remained active in the water column well after the combined acoustic/demersal trawl surveys have taken place, predation and other density-dependent processes would have continued to operate to reduce their abundance. As a result, the relationship between modelled sandeel biomass in June and July and the observed densities in the following autumn grab survey used to fix the value for a in the 1+ sandeel model would, in the case of 0-group sandeels, have been decoupled. Instead, we posit that the difference between

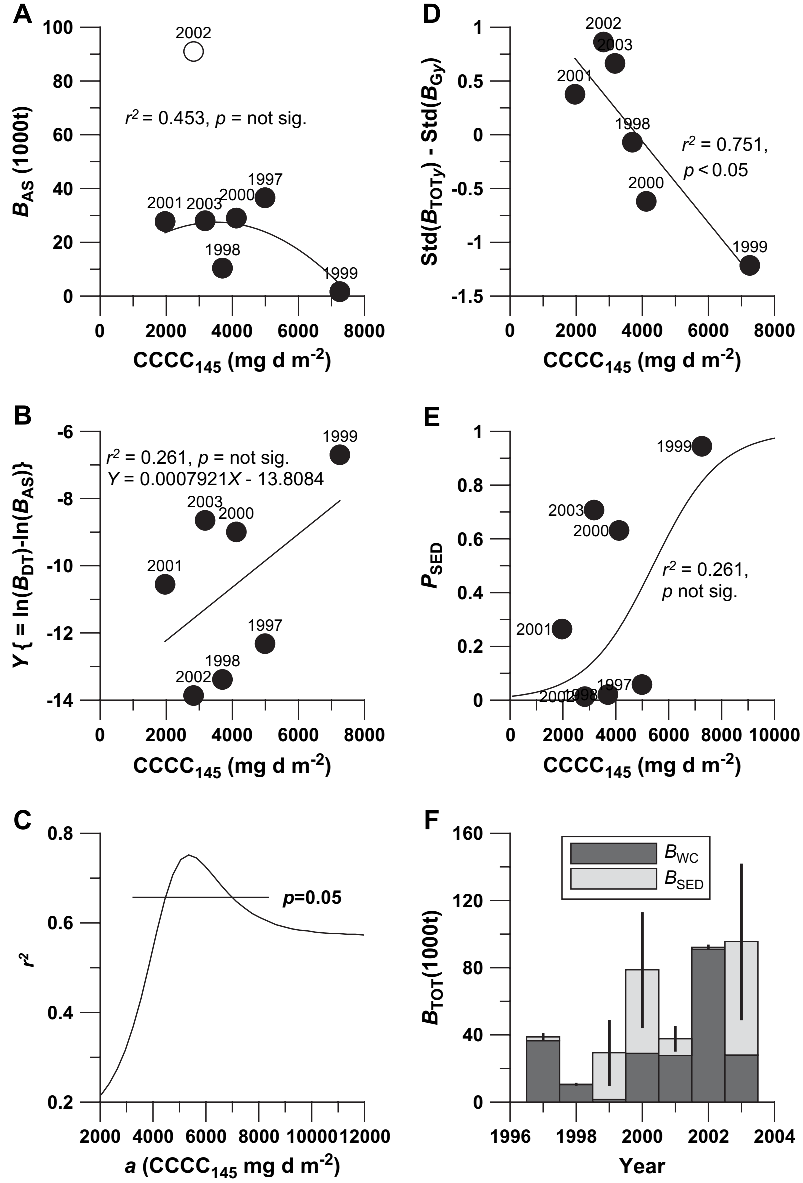

Figure 7. 0-group sandeel model. A. The relationship between the acoustic survey biomass estimate and CCCC$_{145}$. B. Linear regression of CCCC$_{145}$ on Y to estimate b and λ parameters, where $Y_y = \ln(B_{\mathrm{DT}y}) - \ln(B_{\mathrm{AS}y})$, and b and λ are the slope and constant respectively of the regression analysis. C: The relationship between r^2 values of the correlation between the difference between the standardized modelled estimate of 0-group sandeel biomass and the standardized autumn grab-survey estimate of 0-group sandeel density ($\mathrm{Std}B_{\mathrm{TOT}y} - \mathrm{Std}B_{\mathrm{G}y}$) with CCCC$_{145}$, as the value of the a_{145} parameter was varied. D. The best fit correlation between $\mathrm{Std}B_{\mathrm{TOT}y} - \mathrm{Std}B_{\mathrm{G}y}$ and CCCC$_{145}$ obtained at $a_{145} = 5400$. E. The modelled variation in the proportion of 0-group sandeels buried in the sediment with increase in CCCC$_{145}$ showing fit to the data. F. The annual variation in the modelled estimate of 0-group sandeel biomass, both in the water column and in the sediment. Bars indicate the range of modelled biomass in the sediment for the range of a_{145} values that gave a significant r^2 in panel C.

the two abundance estimates should be inversely proportional to the CCCC$_{145}$ at the time of the acoustic/demersal trawl surveys. If the acoustic/demersal trawl surveys took place early in the season (low CCCC$_{145}$), modelled 0-group sandeel biomass at this time should be higher relative to the grab-survey index than would be the case in years when the acoustic/demersal trawl surveys occurred later in the season (higher CCCC$_{145}$), because any population-reducing processes would have had longer to operate.

We tested this hypothesis over a range of a_{145} corresponding to $q_{\mathrm{DT}} = 0.000005$ to $q_{\mathrm{DT}} = 0.02$, assuming that 0-group sandeel catchability in the demersal trawl gear would in all likelihood be lower than the catchability of 1+ sandeels. For each value of a_{145} and q_{DT}, we determined the resulting modelled 0-group sandeel total population biomass and standardized these data ($\mathrm{Std}B_{\mathrm{TOT}y}$) by dividing the modelled biomass in each year ($B_{\mathrm{TOT}y}$) by the mean of the modelled biomass in all years ($\overline{B}_{\mathrm{TOT}}$). The grab-survey data were standardized in the same way ($\mathrm{Std}B_{\mathrm{G}y} = B_{\mathrm{G}y}/\overline{B}_{\mathrm{G}}$). We then carried out a linear correlation analysis to examine how the relationship between $\mathrm{Std}B_{\mathrm{TOT}y} - \mathrm{Std}B_{\mathrm{G}y}$ and CCCC_{145} varied with increase in a_{145} (Figure 7C). The best fit correlation (Figure 7D), obtained at $a_{145} = 5400$ (Figure 7C), provided an optimal estimate of $q_{\mathrm{DT}} = 0.0000726$, but the correlation r^2 was significant over the range $a_{145} = 4750$ to $a = 6850$, corresponding to $q_{\mathrm{DT}} = 0.0000434$ to $q_{\mathrm{DT}} = 0.0002288$. These values of a_{145} and q_{DT} were substituted in Equations (2) and (1) to model variation in the proportion of 0-group sandeels in the sediment with increasing CCCC_{145} (Figure 7E), and to estimate the total 0-group sandeel biomass present each year at the time of the combined acoustic/demersal trawl surveys (Figure 7F). The error bars in Figure 7F indicate the range of 0-group sandeel biomass predicted by the model over the range of a_{145} values that gave statistically significant r^2 in Figure 7C. Modelled 0-group biomass in three of the four fishery-closure years was higher than in the preceding three years when the fishery was in operation, giving a Mann–Whitney test that was not quite statistically significant ($U = 1$, $p = 0.08$).

Discussion

Closing fisheries to protect overexploited stocks is a well established fisheries management option (Pauly *et al.*, 2002). It is likely to be particularly successful with short-lived species with high recruitment potential (Gell and Roberts, 2003; Sale *et al.*, 2005), such as sandeels. On a previous occasion where closure of a sandeel fishery was initiated to protect stocks around Shetland, the local population size increased substantially within just a few years of the start of the moratorium (Wright, 1996; ICES, 2002). The main purpose of this paper has been to establish what effect a more recent sandeel fishery closure has had on a sandeel population off SE Scotland. In the past, the information used to monitor changes in sandeel stock size has generally originated from the fisheries themselves. Such data will invariably be available, given the need for active management of fisheries required under the Common Fisheries Policy. However, with the exception of the Shetland sandeel stock, any assessment of sandeels in the North Sea has to date been carried out at a whole North Sea scale, despite indications that sandeels in the North Sea almost certainly consist of several separate sub-populations (Proctor *et al.*, 1998; Pedersen *et al.*, 1999). Individual, age-based stock assessments at this local population scale have yet to be undertaken on a regular basis, and it is not certain that the data are available that would allow this to be done with adequate precision from a management perspective (Lewy *et al.*, 2004). This is regrettable, because the distribution of one of these local sandeel populations underpinned the demarcation of the sandeel fishery closure area off SE Scotland, and such an assessment would perhaps have provided the best indication of the impact of the management action on the local sandeel stock.

At present Denmark holds, by some considerable margin, the largest North Sea sandeel quota. Logbook data provided by Danish fishers allowed variation in cpue on the fishing grounds off SE Scotland to be examined. In the absence of such a fishery, no fishery-dependent assessment methods would be applicable for monitoring the effects of management through fisheries moratoria. Therefore, if fishery-dependent data underpin the advice that culminates in decisions to close fisheries, maintenance of these data time-series will require some form of scientific or experimental fishery to be contained to provide the information required to monitor the effectiveness of fishery closures. This would be particularly critical if the data collected while a fishery was in operation provided some sort of reference point that was to be used to inform future decisions to re-open fisheries. In the case of the sandeel fishery closure off SE Scotland, such data were only available for the period of the closure, because a strictly limited scientific fishery was continued throughout. The implication is that it could be very difficult for managers to impose complete closures of fisheries.

While closure of the sandeel fishery certainly reduced fishing effort in the area, the effect on catches was less clear-cut. The catch in 1999, the last year that the commercial fishery was open, was in fact very much on a level with catches by the scientific fishery during the four closure years. Only in 1997 and 1998 were catches substantially higher. The cpue in 1999 was considerably lower than in the two preceding years, indicative of a much lower 1+ sandeel biomass in the area. It is possible that this low cpue was not economically viable, forcing fishers to abandon fishing activity in the area in the last year before closure. Variation in cpue calculated over the entire season in each year suggested no major recovery in the 1+ sandeel stock following closure of the fishery, and the same was true for a cpue index determined for the period immediately coinciding with the combined acoustic and demersal trawl surveys. Trends in the biomass of both 0-group and 1+ sandeels derived from three fishery-independent stock assessment methods were also examined. Two of these assessments, the acoustic survey and the demersal trawl survey, were carried out in early summer, at approximately the same time of year as the main fishery. The third method, the nocturnal grab survey, was undertaken in autumn of each year. Like the two cpue indices, none of these fishery-independent methods indicated a clear-cut response of either age class of the sandeel population to the fishery closure, although in the case of 1+ sandeels, all three fishery-independent assessments tended to suggest that there may have been one.

For neither age group did the fishery-independent indices vary consistently over time. Sandeels spend most of their time buried in the sediment, emerging briefly to spawn in mid-winter and again in late spring/early summer to feed on the burst of zooplankton production triggered by the annual plankton bloom (Macer, 1966; Winslade, 1974c; Gauld and Hutcheon, 1990; Bergstad *et al.*, 2001). This characteristic of moving between the seabed sediment and the overlying water column almost certainly explained the inconsistency between the three assessment methods, each of which detected sandeels in the two parts of the marine environment to a greater or lesser extent. Danish cpue and the acoustic survey provided an index of variation in the biomass of 1+ sandeels in the water column; that these two indices should provide the only significant correlations provided strong support for this contention. Moreover, because the proportion of sandeels active in the water column is likely to vary over time, the closer relationship between the acoustic survey index and cpue calculated for the period coincident with the combined acoustic/demersal trawl survey, rather than with cpue calculated over the whole season, further supports this argument.

Although the need to emerge in winter to reproduce is a prerequisite for the continued survival of the species, the strategy by which individual sandeels reach this critical point is much more elective. Like many other organisms subject to high predation mortality, sandeels have to balance the need to obtain food against the need to reduce predation risk (Lima and Dill, 1990). When foraging exposes animals to predation risk they should forage for the minimum time possible and then return to refugia (Pearson *et al.*, 1984). Although not totally immune from predators while buried (Girsa and Danilov, 1976; Hobson, 1986; Temming *et al.*, 2004), it is widely assumed that the seabed sediments provide such a refuge (Reay, 1970). It is during late spring and early summer, when sandeels are most active in the water column, that they feature most strongly in the diets of many fish, seabird, and marine mammal predators (Bailey *et al.*, 1991; Greenstreet *et al.*, 1998; Brown *et al.*, 2001).

If sandeels are most vulnerable to predation while active in the water column, then during the feeding period of late spring to early summer, they should feed only for so long as is necessary to achieve an adequate body condition to ensure their survival through winter and to meet their gonad production requirements. Once these objectives are met, sandeels increase their chances of survival by returning to the sediment and reducing their risk of mortality from predation. Winslade (1974c) suggests that this is indeed the case, i.e. once sandeels attain a “certain fat content”, the over-wintering phase of their life cycle is triggered, causing them to cease feeding in the water column. Certainly the period that sandeels are active in the water appears to be relatively short. Commercial pelagic catches start to increase in April, peak in June, then decline in July (Macer, 1966; Winslade, 1974c; Reeves, 1994). The increase in commercial landings closely tracks the increase in copepod abundance early in the season (Winslade, 1974c), consistent with the observation that emergence only takes place when prey are available in the water column (Winslade, 1974a). In July, however, sandeel landings decline at a much faster rate than expected given the slower decline in copepod abundance (Winslade, 1974b). Somatic growth of sandeels is also highly seasonal, being strongest from March to June and almost ceasing in 1+ sandeels by July (Bergstad *et al.*, 2002). Meeting the increased costs of metabolism associated with the continuing rise in sea temperature into July and August appears to inhibit continued growth. Without the benefit that further increase in body size and condition might endow, the risk from predation associated with continued feeding in the water column would appear to be sufficient to cause sandeels to cease feeding and to bury in the sediments, once they have reached an adequate condition to survive winter. In Japanese waters a slightly different situation exists, but a similar logic applies. Rising seawater temperatures cause a closely related species, *Ammodytes personatus*, to cease feeding and to aestivate in the sediment in order to conserve energy. With increasing temperature, larger sandeels within an age class enter into an aestivation state before smaller fish do, indicating the importance of fat reserves as a stimulus (Tomiyama and Yanagibashi, 2004). Previous studies had also demonstrated that the more opportunity these sandeels had had to feed, the earlier they started to aestivate (Yanagibashi *et al.*, 1997; Yamada *et al.*, 1999).

As the feeding period progressed, the proportion of the sandeel population active in the water column in our study area is likely to have declined while the proportion buried in the sediment would have increased. Consequently, the different assessments undertaken in late May or early June would have been affected not only by the variation in total sandeel biomass, but also by differences in the proportions of the population in the water column and in the sediment. Thus, for example, 68% of variation in the acoustic survey index of 1+ sandeel biomass could be explained by variation in CCCC, a proxy for the total feeding opportunity available to sandeels prior to each of the combined acoustic/demersal trawl surveys. By combining the information provided by two complementary assessment methods carried out at the same time of year, acoustic and demersal trawl surveys, a model was developed to estimate the total biomass of each sandeel age class in the study area at the time of the combined surveys.

The model assumes that the acoustic survey provides an estimate of the absolute abundance of sandeels in the water column. If we consider the acoustic survey in the same way that we have used the demersal trawl survey, i.e. $B_{\mathrm{AS}} = B_{\mathrm{WC}} q_{\mathrm{AS}}$, then we have in effect assumed that $q_{\mathrm{AS}} = 1.0$. This may not be the case. It is possible that other reflecting targets have been mis-classified as sandeels, leading to q_{AS} values >1.0 or alternatively that sandeels have been mis-classified as something else, leading to q_{AS} values <1.0. These potential errors may tend to cancel each other out. However, because assessment of the echo-integral records was undertaken by the same two considerably experienced scientists (SPRG and EA) in each year, there may be a greater potential for bias, in that the same errors were likely to be consistently made in each year. In many ways, this is the preferable case, because a consistent bias means that, if later found necessary, an acoustic survey q_{AS} parameter can be added to revise the model output accordingly. Recent and continuing work to develop an objective algorithm to assign echo-integral data to sandeel and other fish or plankton categories indicated close agreement with the subjective, experience-based method used here (Mosteiro *et al.*, 2004). The model further assumes that the demersal trawl index varies as a linear function of the biomass of sandeels buried in the sediment, such that $B_{DT} = B_{SED}q_{DT}$, where q_{DT}, the catchability coefficient of sandeels in the demersal trawl, is constant. In this study, the same demersal trawl was fished by the same officers and crew operating the same research vessel each year. Moreover, the Jackson rockhopper trawl had been in use for many years prior to this particular study, so no element of learning was involved early in the study. Furthermore, the same 19 stations were fished each year, so the probability of sampling specific sandeel habitats should have remained constant. Hence, variation in the area actually swept by the gear each year was the only source of between-survey variation in effort, but this was taken into account in the formulation of the demersal trawl biomass index. The demersal trawl survey, therefore, violated none of the assumptions underpinning this relationship (King, 1995), so there was no reason to believe that the sandeel catchabilty coefficient for the Jackson rockhopper trawl was not constant across all surveys.

The model states that total sandeel biomass in the area is the sum of sandeel biomass in the water column and sandeel biomass in the sediment (Equation (1)), and that the proportion of sandeel biomass buried in the sediment varies as a logistic function of CCCC prior to the combined acoustic/demersal trawl surveys (Equation (2)). By determining values of the a and b parameters for the logistic curves describing the burial behaviour of each sandeel age class, estimates of the catchability coefficients of both 1+ and 0-group sandeels in the demersal trawl could be derived. For 1+ sandeels this was straightforward. The linear regression used to determine b was statistically significant. Comparison of the various modelled 1+ sandeel biomass estimates with the grab-survey 1+ sandeel abundance index, because both a and q_{DT} were allowed to vary over a predefined range of possible values, was straightforward and provided a relatively limited range of a values giving a significant correlation. Applying the determined value for b and the optimal solution for a in Equation (2) produced a logistic curve of variation in the proportion of sandeel biomass in the sediment with increasing CCCC prior to each combined survey that was a significant fit to the observed data. Using the range of values of a giving a significant correlation between the modelled biomass and the grab index to provide a range of possible q_{DT} values to apply in Equation (1) had a negligible effect on the ranked order of each year's modelled 1+ sandeel biomass. No matter which value of a and associated q_{DT} was used, our estimates of 1+ sandeel biomass derived from the model suggested that the biomass in all four years that the fishery was closed was higher than in any of the preceding three years when the fishery was operating, a result that produced a significant Mann–Whitney test result.

Modelling 0-group sandeel biomass was more problematic. First, the b parameter could not be determined with any great confidence because the linear regression used to estimate it was not statistically significant. Second, because of differences in the biology of 1+ and 0-group sandeels, CCCC values cumulated from a more appropriate start date had to be calculated, and an alternative hypothesis relating modelled 0-group biomass to the 0-group grab-survey index had to be employed. Significant correlations were obtained, but over a relatively wide range of possible values of a and associated estimates of q_{DT}. As a result, the logistic curve describing variation in the proportion of 0-group sandeel biomass buried in the sediment as a function of CCCC failed to provide a significant fit to the observed data. The range of possible q_{DT} values was such that the modelled estimate of 0-group sandeel biomass buried in the sediment was considerable, sufficient to affect the ranked order of each year's estimate of total biomass. Over the full range of possible a and associated q_{DT} values, the same three years, 2000, 2002, and 2003, were always ranked 1–3 (highest biomass). Therefore, whichever value of q_{DT} was applied, modelled 0-group biomass was higher in three of the four fishery-closure years than in each of the preceding three years when the fishery was in operation, a result that failed to produce a significant Mann–Whitney test result. The remaining fishery-closure year was always ranked fifth, being beaten by either 1997 or 1999, depending on the value of q_{DT} used.

The problems experienced in modelling 0-group sandeel biomass almost certainly reflect major differences in the processes that influence 0-group and 1+ sandeel feeding and burial behaviour. While our model appears to have captured the situation regarding 1+ sandeel reasonably well, this appears not to be the case with respect to 0-group sandeels. Numerous factors may affect 0-group sandeel behaviour that at present our model cannot take into account. In determining the parameters a and b of the logistic function we only have one datum for each year. We use all seven years of data to estimate these parameter values, and in doing so we make the assumption that the shape of the curve and its position relative to CCCC is constant over all years. Although this assumption appears to have held reasonably well for 1+ sandeels, there are clear indications that this was not the case for 0-group sandeels. There may be several reasons for this. Variation in mean egg-hatch date between years in turn leads to differences in the timing of metamorphosis from the larval phase (Wright and Bailey, 1996). In our study, we assumed a constant metamorphosis date of Julian day 145 in determining the CCCC values appropriate to 0-group sandeels. Variation in the date of metamorphosis would, in effect, alter the location of the logistic curve describing the proportion of 0-group sandeels in the sediment; effectively altering the *a* parameter of the model. Sandeel growth rates display density-dependence, being lower in years of particularly high cohort strength (Bergstad *et al.*, 2002). In 2002, the acoustic survey estimate of 0-group biomass was higher than in any other year, by a factor of three. This may have impeded growth rates so that, by the time of the survey, the sandeels may not have reached the body size and condition expected given the feeding opportunity apparently available to them. Some studies have suggested a negative density-dependent relationship between 0-group and older sandeels (Arnott and Ruxton, 2002; Furness, 2002). Therefore, in years when 1+ sandeel biomass was elevated, high densities of these older fish in the sediment may have inhibited the burial behaviour of 0-group sandeels. Such factors may well have affected the slope of the logistic curve, altering the *b* parameter, as well as the curve's location relative to CCCC.

The 0-group acoustic survey data suggested that 0-group sandeels remained active in the water column later in the year, and at higher CCCC values, than 1+ sandeels. The metamorphosis dates and growth rate information published by Wright and Bailey (1996) suggest that this was likely, and it is consistent with the observations that 0-group sandeels continue to be caught in the pelagic trawls used by the industrial fishery (Macer, 1966; Winslade, 1974c; Reeves, 1994), and to continue to increase body condition (Bergstad *et al.*, 2002), much later in the year than their older conspecifics. Winslade (1974c) suggested that older sandeels have slower growth rates, so they reach their fat-level thresholds and cease activity in the water column earlier than younger fish. Consequently, older sandeels generally spend less time in the water column (Kvist *et al.*, 2001). Similarly, in Japan, 0-group *A. personata* remained active in the water column longer and started aestivation later than older fish (Tomiyama and Yanagibashi, 2004). This tendency for 0-group sandeels to remain in the water column later in the year may also have contributed to the greater difficulty in parameterizing the 0-group model. Timing of the combined acoustic/demersal trawl surveys was opportune for estimating *b* in the 1+ sandeel model. The data were distributed across a range of CCCC, and related biomass proportions in the sediment, that was ideal for the linear regression analysis. This was less so for 0-group sandeels. At the time of year that the surveys were generally carried out there was a greater tendency for data to be collected at CCCC values well below the value of the *a* parameter, and with low biomass proportions in the sediment, a data distribution that was not conducive to satisfactory linear regression.

The approach to modelling 0-group sandeel biomass needed revision to take account of the fact that 0-group sandeels remained active in the water column sometimes well after the combined acoustic and demersal trawl surveys were completed. In such circumstances, 0-group sandeels would have been prone to greater and continued mortality from predation (Temming *et al.*, 2004; Greenstreet, 2006). Therefore, we had no reason to expect a straightforward relationship between modelled 0-group biomass and the later grab-survey estimates of 0-group sandeel population density. Instead, we proposed that the difference between these two population size estimates, subtracting the standardized grab-survey index from the standardized modelled biomass, was inversely proportional to variation in CCCC, and this was indeed the case.

Three final points are worthy of some consideration. First, the model we apply here effectively tunes the demersal trawl estimates of sandeel biomass in the sediment to the acoustic survey. Therefore, the choice of target strength used in the analysis of the acoustic integral data is critical. Consequently, while the modelled biomass estimates may certainly be considered as estimates of relative biomass, their use as estimates of absolute abundance is critically dependent on the sandeel target strength of -50 db kg^{-1} used here. Armstrong (1986) suggests that sandeel target strength lies within the range -46.9 to -54.7 dB kg^{-1} at 38 kHz. A 3 dB kg^{-1} difference from the target strength used here, i.e. -47 dB kg^{-1}, would result in a halving of our absolute biomass estimates. The relative proportions in the sediment and in the water column predicted by the model would be unchanged, so changes in relative biomass predicted by the model would also remain unaffected by this increase in target strength. Second, by the Julian day of the combined acoustic and demersal trawl surveys in 1997 and 1998, approximately 65% and 90% of the total catches of 16 200 t and 29 300 t of sandeels, respectively, had already been taken by the fishery (HJ and HM, unpublished data). In effect, therefore, in 1997 approximately 10 530 t and in 1998 approximately 26 370 t of sandeels present earlier in the season were not accounted for by the model. In 1999, the sandeel fishery was over by the time the surveys took place, but in that year landings only amounted to 4000 t, so the effect was much reduced. Similarly, in all subsequent years, the fishery was officially closed and scientific landings never exceeded 6500 t, a very small fraction of the modelled biomass in each year. A detailed analysis of seasonal variation in fishery cpue data in relation to the modelled sandeel emergence behaviour is planned for the future. Third, the catchability of 0-group sandeels in the demersal trawl (0.000073) was two orders of magnitude lower than that of 1+ sandeels (0.0204). The demersal trawl simply did not sample 0-group sandeels well.

As 0-group sandeels spend a longer period in the water column, acoustic surveys may instead provide the best assessment of the biomass present in an area in a particular year, particularly if these surveys could be consistently undertaken relatively early in the year, when CCCC is still relatively low, and the proportion of 0-group sandeels likely to be buried in the sediment is small.

The industrial sandeel fishery that operated off SE Scotland targeted 1+ sandeels almost exclusively, in common with the North Sea sandeel fishery generally (Gislason and Kirkegaard, 1998; Kvist *et al.*, 2001). Therefore, it might be expected that this age class should respond to the fishery closure, and the model results confirm that this was the case. In fact 1+ sandeel biomass increased markedly in 2000, the first year of the closure, when perhaps a lag of one or two years might have been anticipated to allow successive recruitment to build stocks. No such lag was apparent because, although allowed, very little fishing actually took place in the area in 1999. Moreover, 0-group sandeel biomass determined by the model in 1999 was higher by a factor of two than in the previous year. Therefore, the first year of the closure was actually preceded by a year of low fishing mortality, combined with reasonable recruit production. Predatory fish biomass in the area was also lower in 1999 than in the two preceding years, presumably resulting in reduced natural mortality, particularly of 0-group sandeels (Greenstreet, 2006). This combination of circumstances allowed 1+ sandeel biomass to increase immediately in the first year of the closure, and to remain at high levels thereafter. The relative contributions of closure of the sandeel fishery, or these other factors, in causing the increase in 1+ sandeel biomass therefore remains debatable. Although not targeted by the fishery, and not appearing in the landings to any significant extent, the model suggested that 0-group sandeel biomass was also higher in three of the four fishery-closure years than in any of the preceding three years, when the fishery was active. These results suggest that the timing of the closure coincided with a period of increased recruit production, and enhanced levels of recruitment to the local sandeel population would certainly have contributed to the increase in 1+ sandeel biomass. At present we are not in a position to speculate as to whether the raised 1+ aged sandeel biomass subsequently contributed to the increase in 0-group sandeel biomass.

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Annex 7. Draft report on the assessment of the effects of the Firth of Forth sandeel fishery closure on breeding seabirds

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1 Introduction

2 Methods

3 Results

4 Discussion

1. Introduction

1.1 The Firth of Forth sandeel closure

The industrial fishery for (mainly lesser) sandeels started in the North Sea in the 1950s and gradually developed into the largest single-species fishery in the region, with landings exceeding 1 million t in some years (Furness 1999). The complex of sand banks off the Firth of Forth in SE Scotland (Wee Bankie, Marr Bank etc.) was not exploited by the sandeel fishery until 1990, when Danish vessels started to fish here. Landings then quickly grew to more than 100,000 t in 1993, a level that was considered to have negative effects on local sandeel stock size as well as breeding productivity of black-legged kittiwakes (Rindorf et al. 2000). In the late 1990s, concern arose after several years of very poor kittiwake breeding productivity on the Isle of May and other colonies in the Firth of Forth area. A zone around the Firth of Forth, including the Wee Bankie, was therefore closed to the sandeel fishery from 2000 (Camphuysen 2005). The closure was initially for a three-year period, but was later extended and will be re-evaluated in 2006. A limited-scale survey fishery by commercial fishing vessels has been maintained throughout the closure period (Camphuysen 2005).

Although the fishery in the Wee Bankie area started in 1990 and was closed from 2000, we here define the fishery as having taken place in 1991-1998. This definition was adopted because sandeel fishery effort and landings in the area were very low in 1990 and 1999, comparable to the commercial survey fishery in 2000-2005 (Fig. 3.2, Rindorf et al. 2000, Frederiksen et al. 2004, DIFRES unpubl. data).

1.2 Aims of this assessment

The original justification for the Firth of Forth sandeel closure was that a reduction in breeding productivity of black-legged kittiwakes had been observed at nearby colonies, and that it was considered likely that this was linked to the fishery through a decline in food availability. Any assessment of the ecological effects of the closure should thus include kittiwakes. Here, we analyse the available data from seabird colonies along the UK North Sea coast, inside and outside the closure zone, to assess whether the fishery and/or the closure has had an effect on breeding seabirds, including, but not restricted to, black-legged kittiwakes.

2. Methods

2.1 Study area

The closure zone for the sandeel fishery from 2000 extended from 55°30’N to 58°N, and from 1°W to the UK coast (Fig. 2.1). The control zone used here extended from 52°N to 55°30’N, and at least 75 km out from the UK coast to encompass the foraging range of most seabirds (Fig 2.1). The exact configuration of the control zone was constrained by ICES statistical ‘squares’ (0.5° latitude by 1° longitude), the smallest scale on which data on fishery effort and landings are available. Within both closure and control zones, we used data from all regularly monitored seabird colonies for the selected species (Fig. 2.1, Table 2.1; see also section 2.3.1).

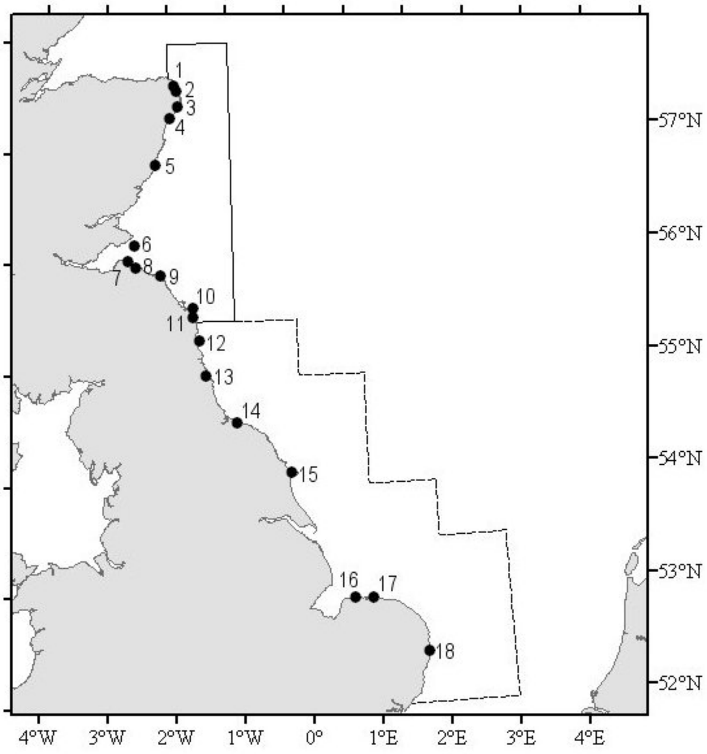


Figure 2.1. Map of the study area off the UK E coast. The sandeel fishery closure zone is indicated by the solid line, and the control zone by the dashed line. Numbers refer to the colonies included in this assessment, see Table 2.1.

Table 2.1. Seabird colonies on the UK E coast included in the assessment of effects of the sandeel fishery closure. Colony numbers refer to Fig. 2.1, and 'status' indicates whether the colony is inside or outside the closure zone.

Number	Colony	Status	Species included
1	Loch of Strathbeg	Inside	Sandwich tern
2	St Fergus	Inside	Arctic tern
3	Bullers of Buchan	Inside	Black-legged kittiwake
4	Sands of Forvie	Inside	Black-legged kittiwake, Sandwich tern
5	Fowlsheugh	Inside	Black-legged kittiwake
6	Isle of May	Inside	Northern fulmar, European shag, black-legged kittiwake, Arctic tern
7	Tantallon	Inside	Northern fulmar
8	Dunbar	Inside	Black-legged kittiwake
9	St Abb's Head	Inside	Northern fulmar, European shag, black-legged kittiwake
10	Farne Islands	Inside	Northern fulmar, European shag, black-legged kittiwake
11	Long Nanny	Inside	Arctic tern
12	Coquet Island	Outside	Northern fulmar, black-legged kittiwake, Sandwich tern, Arctic tern
13	North Shields	Outside	Black-legged kittiwake
14	Saltburn	Outside	Black-legged kittiwake
15	Bempton	Outside	Black-legged kittiwake
16	Scolt Head	Outside	Sandwich tern
17	Blakeney Point	Outside	Sandwich tern
18	Lowestoft	Outside	Black-legged kittiwake

The closure and control zones largely correspond to two of the sandeel aggregations identified in the North Sea (Pedersen et al. 1999, Frederiksen et al. 2005). The southern limit of the closure zone also corresponds exactly to the limit between two clusters of black-legged kittiwake colonies with similar dynamics within clusters, but separate dynamics between clusters (Frederiksen et al. 2005).

2.2 Fisheries data

Data on the distribution of sandeel fishery effort and landings in the closure and control zones 1986-2005 form Danish vessel log books were supplied by the Danish Institute for Fisheries Research (DIFRES). Effort was measured for each vessel as the product of vessel size (engine power in kW) and the number of days spent in the area, summed over all vessels (H. Jensen, unpubl. data). The resulting variable, with the unit kW-days, represents a partially standardised measure of effort.

2.3 Seabird data

2.3.1 Seabird monitoring in the UK

Seabird monitoring in the UK is coordinated by the Joint Nature Conservation Committee (JNCC), in collaboration with the Royal Society for the Protection of Birds (RSPB) and the Shetland Oil Terminal Advisory Group (SOTEAG). Although monitoring at some sites has been running since the 1970s, the comprehensive Seabird Monitoring Programme (SMP) was set up in 1986. Data on population numbers and breeding productivity are collected at a large number of colonies by volunteers, reserve wardens etc., using standardised methods (Walsh et al. 1995). Coverage varies among species, with some (such as black-legged kittiwake) being monitored at scores of colonies, and others (such as Atlantic puffin) at very few colonies. More detailed monitoring is carried out under contract to JNCC at four geographically dispersed sites: the Isle of May (SE Scotland), Fair Isle (Shetland), Canna (NW Scotland) and Skomer (Wales). Results of both extensive and intensive monitoring are collated by JNCC and published in annual reports (e.g. Mavor et al. 2005).

For this assessment, we use data on breeding productivity of 5 species monitored at 18 colonies in the closure and control zones along the UK E coast during 1986-2005 (Tables 2.1, 2.2, Fig. 2.1). Only species, which were monitored regularly (data available for min. 13 years) in at least 3 colonies, were included. These species differ in several aspects of foraging ecology, such as foraging range and the extent of their dependence on lesser sandeels, and are thus differentially sensitive to changes in sandeel abundance and availability (Table 2.2, Furness & Tasker 2000); see Table 2.3 for details on the five species monitored on the Isle of May. A varying proportion of data were missing because species were not monitored at all colonies in all years.

Table 2.2. Overview of available data on breeding productivity of the seabird species monitored by JNCC and included in the assessment of the sandeel fishery closure.

Species	Number of colonies (inside/outside closure zone)	Foraging range and depth	Sandeel dependence
Northern fulmar	5 (4/1)	Offshore, surface	Low
European shag	3 (3/0)	Inshore, 10-30 m	High
Black-legged kittiwake	12 (7/5)	Offshore, surface	High
Sandwich tern	5 (2/3)	Inshore, surface	High?
Arctic tern	4 (3/1)	Inshore, surface	High?

2.3.2 Isle of May

Monitoring of seabirds on the Isle of May commenced in 1973 with a study of Atlantic puffins by the Nature Conservancy Council (NCC). Under NCC's successors, the Institute of Terrestrial Ecology (ITE) and later the Centre for Ecology and Hydrology (CEH), the monitoring programme developed during the 1980s and 1990s to encompass five seabird species and a large range of demographic, behavioural and foraging parameters. Since 1986, the Isle of May has been one of JNCC's key monitoring sites under the SMP. Standardised methods are used to monitor demographic and foraging parameters such as breeding productivity, return rates of adults, nest attendance and fledging mass of chicks (Harris et al. 2005). Results of the monitoring programme are published in annual reports (e.g. Harris et al. 2005) and have formed the basis for a large number of scientific papers.

The five seabird species monitored in detail on the Isle of May all feed extensively on lesser sandeels during the breeding season (Wanless et al. 1998, Daunt et al. submitted). However, they use a range of different foraging and chick provisioning strategies (Table 2.3), and are thus differentially sensitive to changes in sandeel abundance and availability (Furness & Tasker 2000).

For this assessment, we use breeding productivity (number of fledged chicks per occupied nest) of these five species, as well as the mean body mass of near-fledged common guillemot and Atlantic puffin chicks. These data were available for all years (1986-2005).

Table 2.3. Main features of the foraging ecology during the breeding season of the five seabird species monitored in detail on the Isle of May.

Species	Foraging range	Foraging technique	Prey transport	Sandeel dependence	Key references
European shag	Inshore (< 20 km)	Pursuit diver (0-60 m)	Multiple prey in stomach	> 90% most years, mostly 1+ group	(Harris & Wanless 1991, Wanless et al. 1991a, Wanless et al. 1991b, Rindorf et al. 2000, Camphuysen 2005)
Black-legged kittiwake	Offshore (< 80 km)	Surface feeder	Multiple prey in stomach	Mostly > 80%, 1+ group in early season, 0 group later	(Rindorf et al. 2000, Lewis et al. 2001, Daunt et al. 2002, Bull et al. 2004, Camphuysen 2005)
Common guillemot	Offshore (< 50 km)	Pursuit diver (0-70 m)	Single prey in bill	Variable (0-80%), 1+ group in early season, 0 group later	(Rindorf et al. 2000, Wilson et al. 2004, Camphuysen 2005, Wanless et al. 2005)
Razorbill	Offshore (< 50 km)	Pursuit diver (0-35 m)	Multiple prey in bill	Mostly > 80%	(Wanless et al. 1988, Harris et al. 1990, Wanless et al. 1990, Harris et al. 2005)
Atlantic puffin	Offshore (< 50 km)	Pursuit diver (0-35 m)	Multiple prey in bill	Mostly > 75%, 1+ group in early season, 0 group later	(Wanless et al. 1988, Harris et al. 1990, Wanless et al. 1990, Wanless et al. 2004)

2.4 The before-after control-impact (BACI) design

Proper assessment of the effect of any anthropogenic impact on the environment requires data from before and after the impact takes place, and from locations affected and unaffected by the impact. Stewart-Oaten et al. (1986) suggested the before-after control-impact (BACI) design as an appropriate way to collect and analyse data on environmental impacts. In this design, data from single locations affected and unaffected by the impact are collected on several occasions before and after the impact and analysed in a two-way analysis of variance (ANOVA), and a statistically significant interaction between period and location indicates an effect of the impact. This design was elaborated by Underwood (1994) to include several control locations.

In this assessment, we treat the presence or absence of a fishery in the closure zone as the impact, years as temporal replicates within each period and colonies as spatial replicates within each zone. We use data from 1986-2005, i.e. the period when the SMP has been running. Because data are available for years both before the fishery started and after it was closed, we could in principle evaluate both the effect of the fishery and the closure. However, in the statistical analyses presented here we have pooled years before and after the fishery was active, and thus we test for the overall effect of having an open fishery on lesser sandeels within foraging range of seabird colonies depending on this resource, assuming that effects operated without lags, i.e. came into force as soon as the fishery opened and ceased when it was closed.

2.5 Statistical methods

2.5.1 UK-wide single-species analyses

The amount of data available varied markedly among species (see Table 2.1), and we chose the statistical approach accordingly to estimate the fishery/closure effect in a BACI design. For species monitored at several colonies in- and/or outside the closure zone, we used a nested two-way ANOVA, with zone and period as main effects, and colony and year as nested effects. For species monitored at only one colony in either or both zones, nesting was adjusted accordingly. Annual sample size (number of nests monitored) was used as a weighting factor. A significant interaction between zone and period would indicate an effect of the fishery/closure on breeding productivity, controlling for any overall differences between zones and periods (Stewart-Oaten et al. 1986, Underwood 1994).

For species only monitored in the closure zone, we used a nested one-way ANOVA, with period as main effect, year as nested effect and colony as random effect. This design is less powerful and only allows inference about whether the fishery period differed from the non-fishery period.

2.5.2 Isle of May single-species analyses

In order to estimate properly the effect of the fishery/closure on Isle of May seabirds, we attempted to control for environmental variables, which might also affect demographic parameters. As environmental variables we included a) local winter sea surface temperature lagged by one year, which has been shown to affect black-legged kittiwake breeding productivity on the Isle of May (Frederiksen et al. 2004); b) the mean length of 1 group lesser sandeels collected from chick-feeding Atlantic puffins, adjusted to 1 June (Wanless et al. 2004 and updated), which was correlated with common guillemot breeding productivity on the Isle of May (Frederiksen et al. submitted-a); and c) an index of the biomass of sandeel larvae (SBI) in the NW North Sea, based on Continuous Plankton Recorder data, which was correlated with seabird breeding productivity on the Isle of May 1986-2003 with a one-year lag (Frederiksen et al. submitted-a). We used standard multiple regression to fit all possible models including one or more of the four predictor variables (fishery/closure and three environmental variables). These sixteen models were then ranked using Akaike's Information Criterion (AIC), and the importance of each predictor was evaluated by summing the Akaike weights

(w_i) for all models where the effect appeared (Burnham & Anderson 1998). Model-averaged estimates and standard errors of the fishery/closure effect were calculated using Akaike weights according to Burnham & Anderson (2004). Significance of the fishery/closure effect was then evaluated with a t test with 20 – 5 = 15 degrees of freedom. In contrast to traditional hypothesis testing, this approach is not conditional on one specific 'best' model, and it provides a test of the fishery/closure effect controlled for environmental effects and adjusted for model selection uncertainty (Burnham & Anderson 2004). We thus combined traditional hypothesis testing with model selection based on information theoretical measures, rather than relying on only one of these approaches, as advocated by Stephens et al. (2005). The lagged SBI was not available for 2005, so the full analyses were carried out with data from 1986-2004; the degrees of freedom for the t test were adjusted accordingly to 14. To confirm the findings, we repeated the analyses for 1986-2005 using only two environmental variables; conclusions about the importance of the fishery/closure from these analyses did not differ from the full analyses of 1986-2004 data (results not shown).

2.5.2Multivariate methods for Isle of May across-species analyses

In addition to assessing the effect of the fishery/closure on individual species, we also attempted to evaluate whether it had an effect on the seabird community. To do this, we used multivariate statistical methods, which estimate the relationship between one or more predictor variables and a linear combination of several response variables, in our case seabird breeding productivity and chick fledging mass. This combination is termed a canonical variable or axis. Correlations between the original response variables and the canonical axis (often termed loadings) indicate how important each response variable is to the canonical axis, and thus how much influence they have on the relationship with predictor variables. McGarigal et al. (2000) provide a very useful overview of the most commonly used multivariate methods in ecology.

When only categorical predictors are included, multivariate analysis of variance (MANOVA) extracts the canonical variable most closely associated with the predictors, and tests the effect of these predictors on the canonical response variable. This approach allows testing of whether fishery/closure had an overall effect on breeding performance of our five seabird species.

In order to test for such an effect while controlling for environmental variables, we used partial canonical correspondence analysis (Lepš & Šmilauer 2003) in CANOCO 4.5 (ter Braak & Šmilauer 2002) with the same environmental variables as for the single-species analyses. This procedure can be viewed as a multivariate analogue of analysis of covariance. A canonical variable, maximally associated with categorical and/or continuous predictors, is extracted after controlling for other predictors, here termed 'covariables'.

We also carried out a principal component analysis of the seven response variables, and used multiple regression to relate the first two principal components (canonical variables extracting the strongest possible common signals) to the four predictors used in the single-species analyses.

3. Results

3.1 Distribution of the sandeel fishery

Fishery effort in the control zone was unequally distributed, but the spatial distribution was roughly constant throughout the study period, although the northernmost part was not exploited during the 1980s (Fig. 3.1). Within the closure zone, fishery effort was concentrated in one ICES statistical square (41E8), largely corresponding to the Wee Bankie. During the 1990s, mean effort in this square was similar to the most heavily exploited parts of the control zone (Fig 3.1). Both effort and landings in the closure zone rose quickly from 1990 and peaked in 1993, remaining high until 1998 (Fig. 3.2). Although the fishery was open and active in 1990 and 1999, effort and landings in these years were low and similar to the monitoring fishery in 2000-2005 (Fig. 3.2), justifying our definition of 1991-1998 as the fishery period.

Figure 3.1. Effort of the industrial sandeel fishery in the closure and control zones in the western North Sea, 1986-2005. Effort is shown for each ICES statistical square, averaged over all years in three periods, before, during and after the fishery operated in the closure zone. The sandeel fishery closure zone is indicated by the solid line, and the control zone by the dashed line.

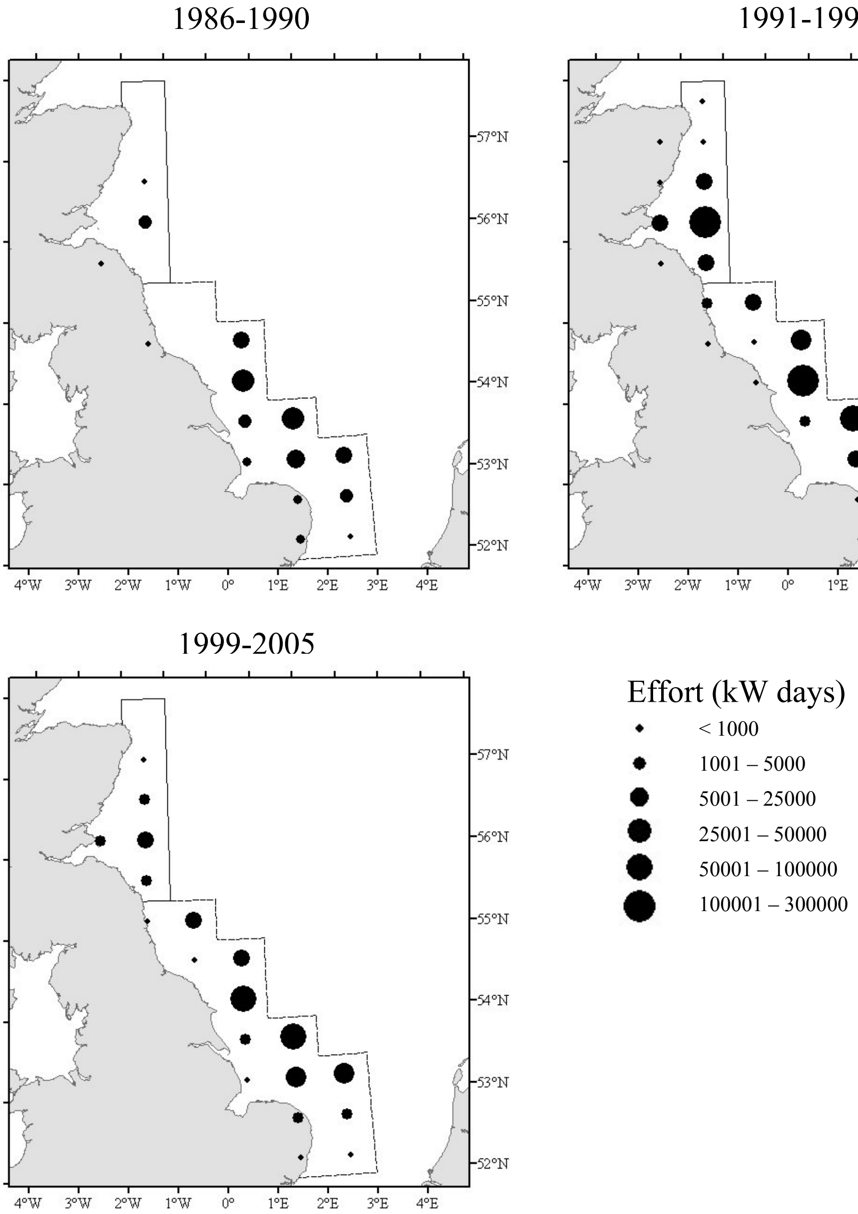
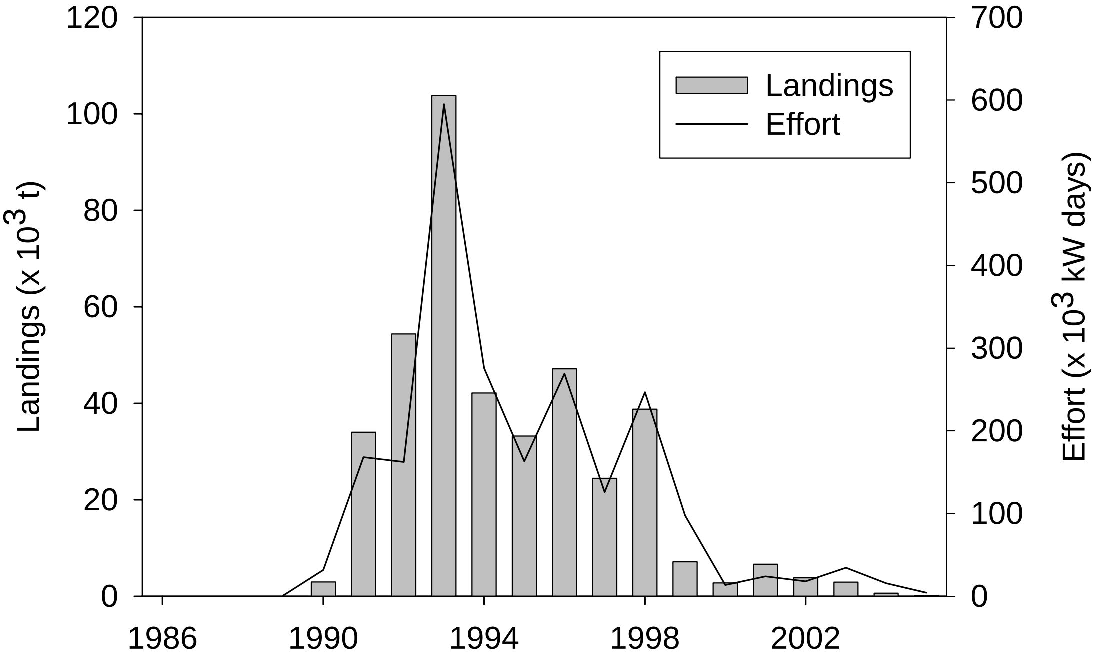


Figure 3.2. Landings and effort of the industrial sandeel fishery in the closure zone off E Scotland, 1986-2005, including the restricted-effort monitoring fishery in 2000-2005.



3.2 Effects of the sandeel fishery/closure on UK seabirds

3.2.1 Single-species analyses

Northern fulmar. Data were available from 5 colonies, 4 inside the closure zone and 1 outside it. Nineteen of the 100 data points were missing. We used a weighted nested two-way ANOVA to estimate the fishery/closure effect in a partly replicated BACI design. The interaction between period and zone was non-significant ($F_{1,56} = 0.03$, $P = 0.86$). Northern fulmars had a slightly higher breeding productivity outside the closure zone and during fishery years (Fig. 3.3), but only the zone effect was significant (zone: $F_{1,56} = 4.26$, $P = 0.044$; period: $F_{1,56} = 1.68$, $P = 0.20$).

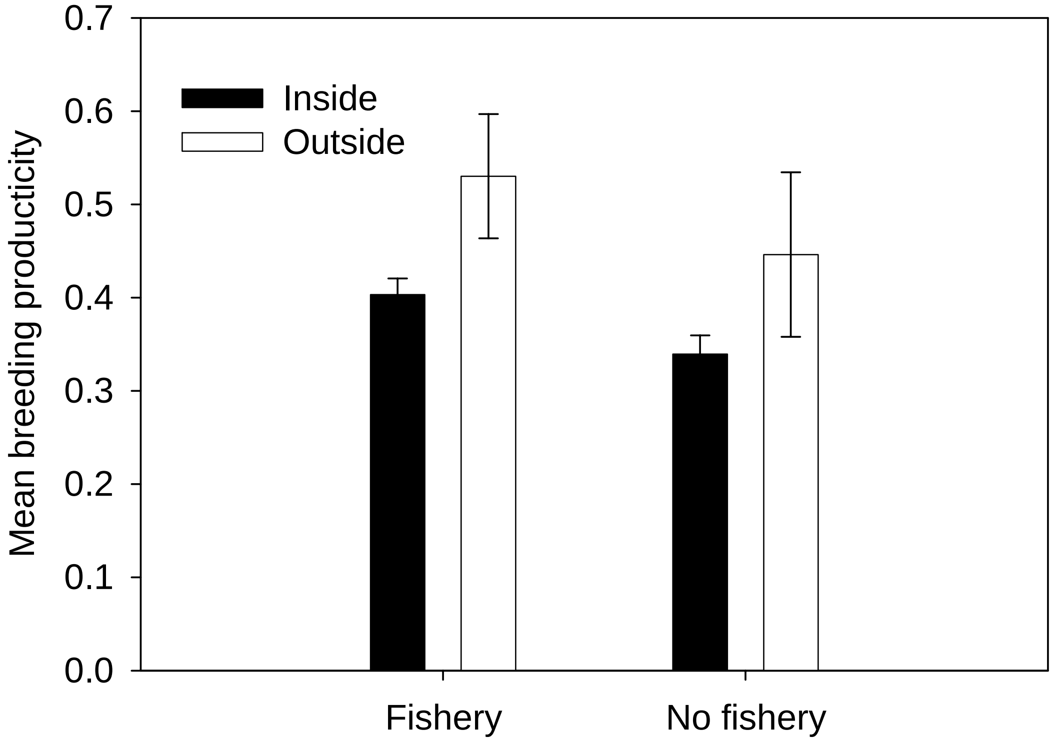


Figure 3.3. Mean breeding productivity (fledged chicks per nest) of northern fulmars inside and outside the closure zone, in fishery and non-fishery years. Error bars indicate ± 1 standard error.

European shag. Data were available from 3 colonies, all inside the closure zone. Five of the 60 data points were missing. We used a weighted nested one-way ANOVA to estimate the fishery/closure effect. The fishery/closure effect was not significant ($F_{1,33} = 0.03$, $P = 0.86$; Fig. 3.4).

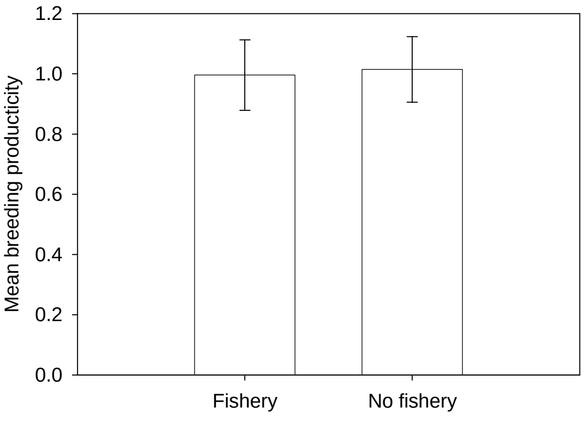


Figure 3.4. Mean breeding productivity (fledged chicks per nest) of European shags in fishery and non-fishery years at three colonies inside the closure zone. Error bars indicate ± 1 standard error.

Black-legged kittiwake. Data were available from 12 colonies, 7 inside the closure zone and 5 outside it. Twenty-six of the 240 data points were missing. We used a weighted nested two-way ANOVA to estimate the fishery/closure effect in a replicated BACI design. The interaction between period and zone was highly significant ($F_{1,182} = 10.03$, $P = 0.0018$). Breeding productivity did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (*post hoc* comparison, adjusted for multiple testing: difference = 0.275 chicks nest^{-1}, $t_{1,18} = 8.01$, $P < 0.0001$; Fig. 3.5). Both main effects were highly significant (zone: $F_{1,182} = 56.48$, $P < 0.0001$; period: $F_{1,182} = 19.25$, $P < 0.0001$), but this should be interpreted cautiously given the significant interaction. Breeding productivity was lower inside the closure zone than outside throughout the 1990s (Fig. 3.6), i.e. also in the two years not defined here as fishery years (1990 and 1999). The test for fishery/closure effect would have been even more significant if we had defined these two years as fishery years.

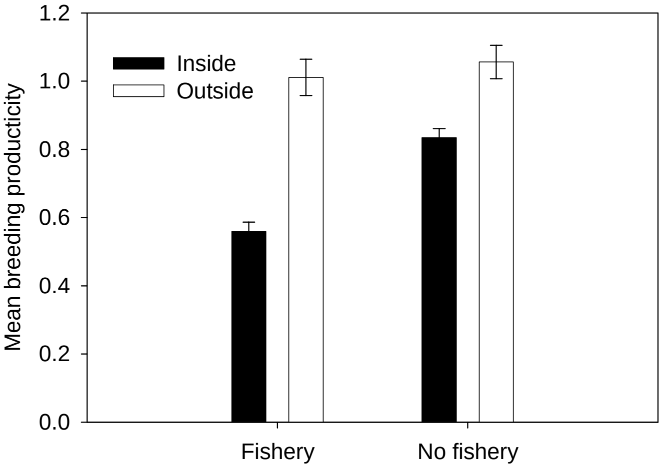


Figure 3.5. Mean breeding productivity (fledged chicks per nest) of black-legged kittiwakes inside and outside the closure zone, in fishery and non-fishery years. Error bars indicate ± 1 standard error.

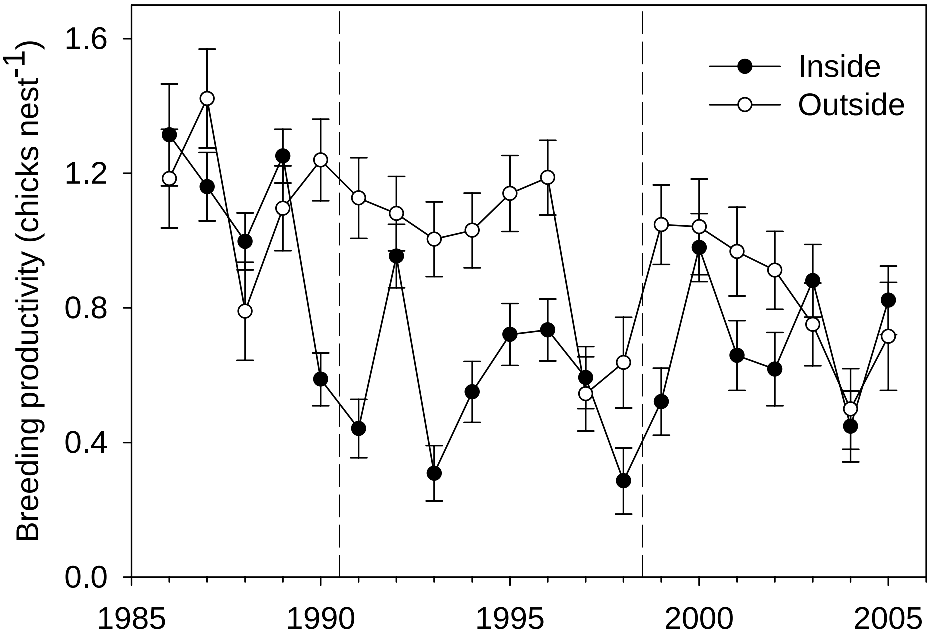


Figure 3.6. Mean breeding productivity (fledged chicks per nest) of black-legged kittiwakes inside and outside the closure zone from 1986 to 2005. The vertical dashed lines indicate the fishery period as defined here (1991-1998). Error bars indicate ± 1 standard error.

Sandwich tern. Data were available from 5 colonies, 2 inside the closure zone and 3 outside it. Twenty of the 100 data points were missing. We used a weighted nested two-way ANOVA to estimate the fishery/closure effect in a replicated BACI design. Although the differences in breeding productivity were in the expected direction if the fishery/closure had an effect (Fig. 3.7), the interaction between period and zone was non-significant ($F_{1,55}$ = 1.70, P = 0.20) due to large standard errors. Sandwich terns had a higher breeding productivity outside the closure zone and during non-fishery years (Fig. 3.7), but only the zone effect was significant (zone: $F_{1,55}$ = 7.98, P = 0.0066; period: $F_{1,55}$ = 1.07, P = 0.30).

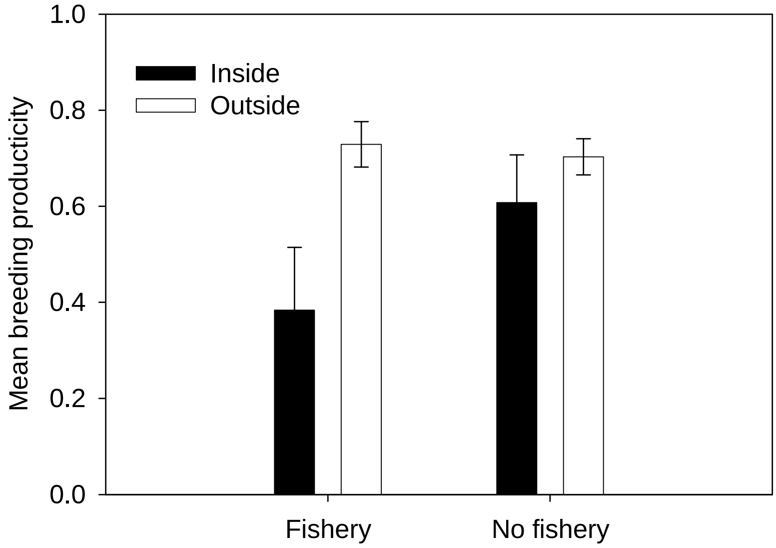


Figure 3.7. Mean breeding productivity (fledged chicks per nest) of Sandwich terns inside and outside the closure zone, in fishery and non-fishery years. Error bars indicate ± 1 standard error.

Arctic tern. Data were available from 4 colonies, 3 inside the closure zone and 1 outside it. Nineteen of the 80 data points were missing. We used a weighted nested two-way ANOVA to estimate the fishery/closure effect in a partly replicated BACI design. The interaction between period and zone was non-significant ($F_{1,37} = 0.06$, $P = 0.82$). Arctic terns had a higher breeding productivity outside the closure zone, whereas there was no apparent effect of period (Fig. 3.8; zone: $F_{1,37} = 19.6$, $P < 0.0001$; period: $F_{1,37} = 0.03$, $P = 0.85$).

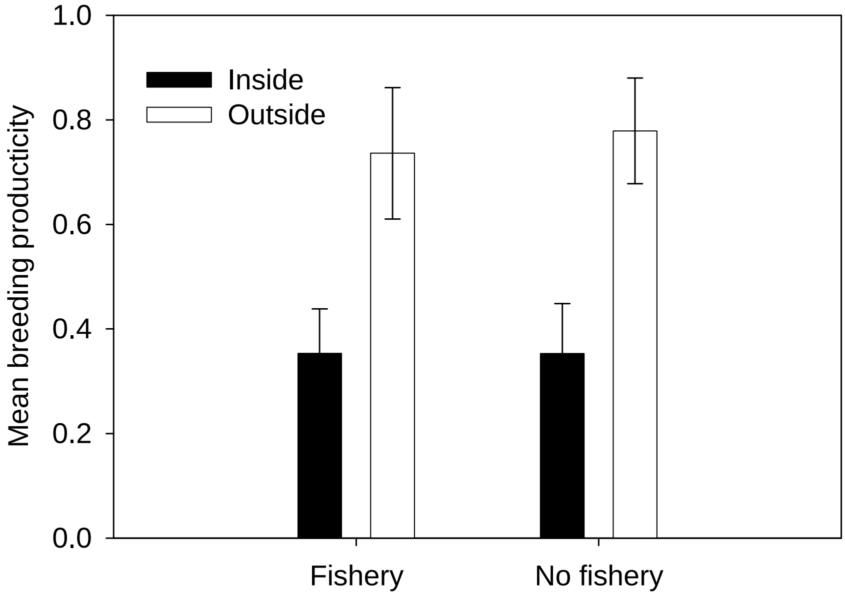


Figure 3.8. Mean breeding productivity (fledged chicks per nest) of Arctic terns inside and outside the closure zone, in fishery and non-fishery years. Error bars indicate ± 1 standard error.

3.3 Effects of the sandeel fishery on Isle of May seabirds

3.3.1 Single-species analyses

European shag. None of the predictors were included in the highest-ranking model for breeding productivity, and the summed Akaike weights were low (< 0.40) for all predictors. Controlling for environmental variables, the fishery/closure had little effect on breeding productivity (-0.023 (SE 0.063) chick nest^{-1}, $\Sigma w_i = 0.20$, $t_{14} = -0.37$, $P = 0.72$; Fig. 3.9).

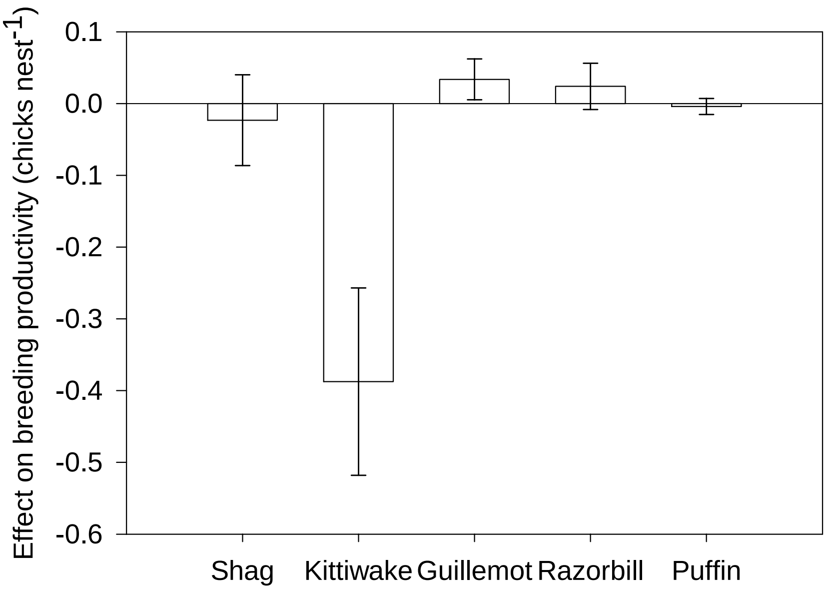


Figure 3.9. Model-averaged effects of the fishery/closure on seabird breeding productivity on the Isle of May. Error bars indicate ± 1 standard error.

Black-legged kittiwake. Lagged SST and the fishery/closure effect were included in the highest-ranking model for breeding productivity (R^2 = 67.2%), confirming the results of Frederiksen et al. (2004). The summed Akaike weights were very high (> 0.96) for these two predictors, and moderate (0.45 – 0.55) for lagged SBI and 1 group sandeel size. Controlling for environmental variables, the fishery/closure had a strong negative effect on breeding productivity (-0.387 (SE 0.131) chick nest^{-1}, $\Sigma w_i = 0.964$, $t_{14} = -2.97$, $P = 0.010$; Fig. 3.9).

Common guillemot. 1 group sandeel size and the fishery/closure effect were included in the highest-ranking model for breeding productivity (R^2 = 77.8%). The summed Akaike weight was very high (> 0.99) for 1 group sandeel size, moderate (0.70) for fishery/closure, and low (< 0.40) for lagged SBI and lagged SST. Controlling for environmental variables, the fishery/closure had at most a weak positive effect on breeding productivity (0.034 (SE 0.029) chick nest^{-1}, $\Sigma w_i = 0.70$, $t_{14} = 1.18$, $P = 0.26$; Fig. 3.9).

1 group sandeel size and lagged SST were included in the highest-ranking model for fledging mass (R^2 = 50.3%). The summed Akaike weight was high (> 0.90) for 1 group sandeel size, moderate (0.50) for lagged SST, and low (< 0.35) for lagged SBI and fishery/closure. Controlling for environmental variables, the fishery/closure had little effect on fledging mass (3.43 (SE 5.92) g, $\Sigma w_i =$ 0.30, $t_{14} = 0.58$, $P = 0.57$; Fig. 3.10).

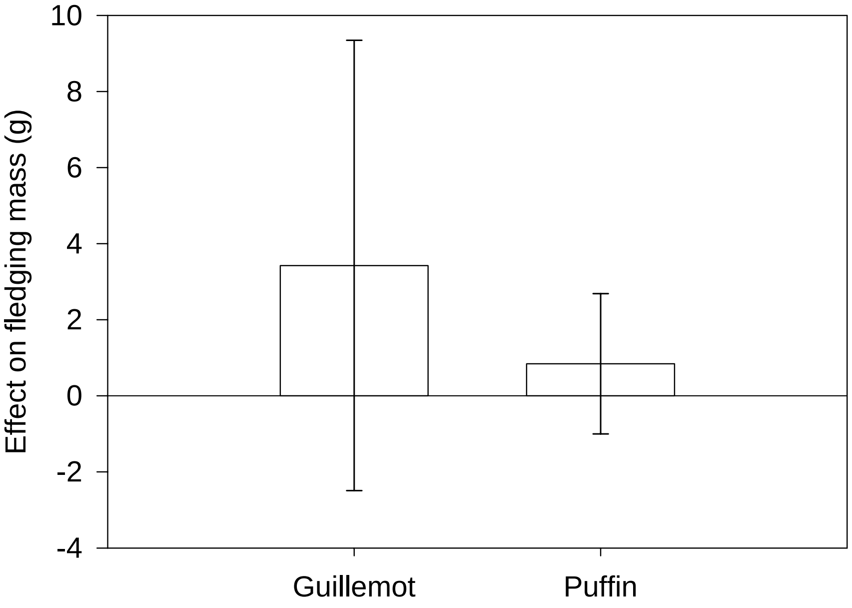


Figure 3.10. Model-averaged effects of the fishery/closure on mean fledging mass of seabird chicks on the Isle of May. Error bars indicate ± 1 standard error.

Razorbill. 1 group sandeel size, fishery/closure and lagged SBI were included in the highest-ranking model for breeding productivity (R^2 = 64.9%). The summed Akaike weight was very high (> 0.99) for 1 group sandeel size, high (0.78) for lagged SBI, moderate (0.46) for fishery/closure and low (0.18) for lagged SST. Controlling for environmental variables, the fishery/closure had little effect on breeding productivity (0.024 (SE 0.032) chick nest^{-1}, $\Sigma w_i = 0.46$, $t_{14} = 0.74$, $P = 0.47$; Fig. 3.9).

Atlantic puffin. Lagged SST was included in the highest-ranking model for breeding productivity (R^2 = 47.5%). The summed Akaike weight was very high (> 0.96) for lagged SST, moderate (0.50) for 1 group sandeel size, and low (< 0.25) for lagged SBI and fishery/closure. Controlling for environmental variables, the fishery/closure had little effect on breeding productivity (-0.0041 (SE 0.011) chick nest^{-1}, $\Sigma w_i = 0.17$, $t_{14} = -0.37$, $P = 0.72$; Fig. 3.9).

Lagged SST was included in the highest-ranking model for fledging mass (R^2 = 22.2%). The summed Akaike weight was moderate (0.52 – 0.55) for lagged SST and 1 group sandeel size, and low (< 0.30) for lagged SBI and fishery/closure. Controlling for environmental variables, the fishery/closure had little effect on fledging mass (0.84 (SE 1.84) g, $\Sigma w_i = 0.21$, $t_{14} = 0.46$, $P = 0.65$; Fig. 3.10).

3.3.2 Across-species analyses

We used three approaches to investigate the fishery/closure effect on the seabird community on the Isle of May. In the simplest approach, a one-way MANOVA with all seven response variables (breeding productivity of five species and mean fledging mass of two species) showed an almost significant fishery/closure effect ($F_{7,12} = 2.47$, $P = 0.08$). However, when only breeding productivity of the five species was included, the effect was significant ($F_{5,14} = 3.78$, $P = 0.02$). In both cases, the canonical MANOVA axis was most strongly associated with black-legged kittiwake breeding productivity (loadings 0.69 and 0.70, respectively), whereas loadings for other response variables were lower (< 0.54).

In order to estimate the community-level effect of fishery/closure controlled for environmental variables, we carried out a partial canonical correspondence analysis in CANOCO, using lagged SST, 1 group sandeel size and lagged SBI as covariables. Significance of the fishery/closure effect was evaluated in a Monte Carlo test, by comparing the observed correlation with the canonical variable to 9999 randomised correlations. With all seven response variables, fishery/closure was not significantly correlated with the canonical axis ($F_{1,15}$ = 2.13, P = 0.124). When only breeding productivity of the five species was included, the correlation was almost significant ($F_{1,15}$ = 3.03, P = 0.057). In both cases, the canonical axis was most strongly associated with black-legged kittiwake breeding productivity (loadings 0.42 and 0.44, respectively), whereas loadings for other response variables were low (< 0.26 and 0.30, respectively).

We also used multiple regression to relate the two first principal components of the seven response variables to the four predictors used in the single-species analyses. The first principal component explained 57% of the total variation and was positively associated with all original response variables (all loadings > 0.35). 1 group sandeel size, lagged SST and lagged SBI were included in the highest-ranking model (R^2 = 67.8%). The summed Akaike weight was very high (> 0.98) for 1 group sandeel size, high (0.70) for lagged SBI, moderate (0.57) for lagged SST and low (0.22) for fishery. Controlling for environmental variables, the fishery/closure had little effect on the first principal component (0.15 (SE 0.32) units, Σw_i = 0.22, t_{14} = 0.47, P = 0.65; Fig. 3.11). The second principal component explained 19% of the total variation and was positively associated with shag and kittiwake breeding productivity (loadings > 0.55), and negatively with guillemot and razorbill breeding productivity (loadings < -0.40). 1 group sandeel size, lagged SST and fishery/closure were included in the highest-ranking model (R^2 = 65.9%). The summed Akaike weight was high (> 0.88) for 1 group sandeel size, lagged SST and fishery, and low (0.15) for lagged SBI. Controlling for environmental variables, the fishery/closure had a negative effect on the second principal component (-1.10 (SE 0.44) units, Σw_i = 0.91, t_{14} = 2.30, P = 0.038; Fig. 3.11).

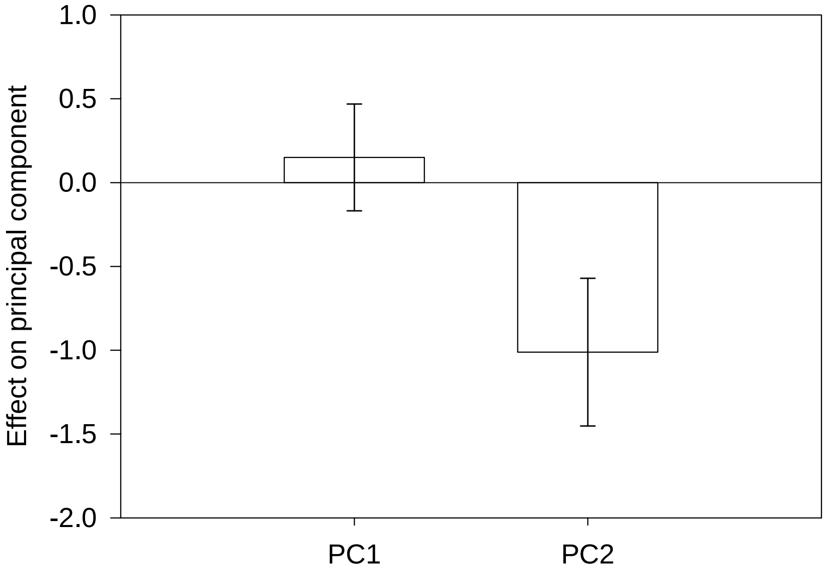


Figure 3.11. Model-averaged effects of the fishery/closure on the two first principal components of seven seabird reproductive performance variables on the Isle of May. Error bars indicate ± 1 standard error.

4. Discussion

4.1 Effects on black-legged kittiwakes

There was clear evidence that the breeding productivity of black-legged kittiwakes at local colonies was reduced during the period when the sandeel fishery was active in the Wee Bankie area (Figs. 3.5, 3.6, 3.9). This confirms and extends the findings of Frederiksen et al. (2004), who found that both breeding productivity and annual probability of survival of adult breeders at the Isle of May were reduced during 1991-98, when the fishery was operating. Here, we document that this reduction in breeding productivity occurred throughout the area affected by this fishery and subsequently closed from 2000. Frederiksen et al. (2004) also found a negative effect of high sea surface temperatures on breeding productivity, with a one-year lag, and concluded that the low breeding productivity in 1990 and 1999 was more likely linked to exceptionally warm winters in 1988/89 and 1997/98 than to the very limited fishery occurring in those years. The statistical relationship between breeding productivity, fishery and sea surface temperature was very similar for the seven colonies in the closure zone (Frederiksen et al. submitted-b).

Breeding productivity did not differ between the two periods in the control zone (Fig. 3.5), although colonies in this area did experience bad years in 1997 and 1998 (Fig. 3.6). Only parts of the control zone were affected by sandeel fisheries within seabird foraging range, and this exposure was relatively constant in the periods before, during and after the fishery operated in the closure zone (Fig. 3.1). Interestingly, Frederiksen et al. (2005) found that black-legged kittiwake breeding productivity was correlated among colonies within each zone, but not between the two zones, and inferred that birds in each zone depended on separate sandeel aggregations with non-synchronous dynamics. However, it is also possible that the different patterns of temporal variation were related to the sandeel fishery affecting the area off the closure zone in E Scotland, but only parts of the area further south off the control zone. The extremely poor breeding success noted for black-legged kittiwakes as well as other seabird species at the Isle of May and elsewhere in 2004 (Proffitt 2004, Mavor et al. 2005), which has been tentatively linked to poor food quality (Wanless et al. 2005), was apparent in both the closure and control zones (Fig. 3.6), indicating that some large-scale environmental processes were common to the two zones.

4.2 Effects on other seabirds

No consistent effects of the fishery/closure were found for any seabird species other than black-legged kittiwakes, whether on the Isle of May or across the UK. This conclusion remains tentative because coverage was much less extensive for these other species. In particular, no diving species were monitored regularly in the control zone, and we were therefore unable to test fully whether the fishery/closure affected any of these species. However, no effects on breeding performance were found for the four diving species monitored on the Isle of May (Figs. 3.9, 3.10). The statistical effects found in the across-species multivariate analyses of Isle of May data (section 3.3.2, Fig. 3.11) probably reflected the difference in black-legged kittiwake breeding productivity between periods with and without a fishery. The strong effect on black-legged kittiwakes, and the lack of effects on diving species, may indicate that the fishery mainly affected the availability of sandeels to surface feeders, and that any reductions in sandeel abundance were sufficiently small that diving seabirds were able to compensate for them and feed their chicks as normal (cf. Daunt et al. submitted). Availability might be affected e.g. if the activity of fishing vessels caused sandeel shoals to change the extent or timing of their diurnal vertical migration between sediment and surface waters.

If sandeel availability was affected by the fishery, effects would be expected not only for black-legged kittiwakes, but also for other surface-feeding seabirds. It was not surprising that no effect was found for northern fulmars (Fig 3.3), as they are generalist predators and sandeels usually only form a minor component of the diet (Phillips et al. 1999, Ojowski et al. 2001). In contrast, Sandwich and Arctic terns are both regarded as sandeel specialists (Furness & Tasker 2000), and these species would be expected to be affected in a similar way to black-legged kittiwakes. Arctic terns did indeed suffer large-scale breeding failures in Shetland during the period of locally reduced sandeel abundance/availability in the 1980s, along with black-legged kittiwakes (Monaghan et al. 1989). However, we found no significant effects of the fishery/closure on tern breeding productivity (Figs. 3.7, 3.8). One complicating factor affecting terns is that they are prone to desert the colony completely when conditions are bad, and breeding productivity is thus not necessarily a very useful metric for monitoring their performance, because data are likely to be missing during the worst years. Detailed examination of SMP data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie (included in the analysis) was more or less completely deserted from 1992-1998, and that this species did not breed on the Isle of May (not included in the analysis) until 1999 (cf. Harris et al. 2000). Combined with the observed non-significant reduction in breeding productivity in the closure zone during the fishery years (Fig. 3.7), this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes. However, there was no indication that Arctic terns were affected (Fig. 3.8).

4.3 Conclusions

Breeding productivity of black-legged kittiwakes in the closure zone was reduced during 1991-1998, when the sandeel fishery was operating, and there were some indications that Sandwich terns were affected as well. No effects were found for any diving seabirds. Black-legged kittiwakes and Sandwich terns are surface feeders with a high degree of dependence on sandeels, and these results imply that the availability of sandeels to surface feeders was reduced during the 1990s. However, because we have only one closure zone and one control zone, it is only possible to conclude that the two zones differed in some respect during this period (Underwood 1994). Although the most obvious difference between the zones was the operation and subsequent closure of the sandeel fishery, we cannot exclude the possibility that the low sandeel availability to surface feeders in the closure zone in the 1990s was caused by some unidentified environmental perturbation rather than by the sandeel fishery. However, a cautious interpretation consistent with the precautionary approach might conclude that it is probable that the fishery caused a decline in sandeel availability for surface feeders, leading to reduced breeding productivity of black-legged kittiwakes throughout the closure zone.

5. Acknowledgements

We thank Henrik Jensen, Danish Institute for Fisheries Research, for supplying data on the spatial distribution of sandeel fisheries, Roddy Mavor, Joint Nature Conservation Committee, for supplying seabird data (other than terns) from the UK Seabird Monitoring Programme, and Sabine Schmitt, Royal Society for the Protection of Birds, for supplying tern data. Thanks also to Francis Daunt, Centre for Ecology and Hydrology, for producing the maps.

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7. Appendix: Scientific names of organisms mentioned in the text

Arctic tern	*Sterna paradisaea*
Atlantic puffin	*Fratercula arctica*
Black-legged kittiwake	*Rissa tridactyla*
Common guillemot	*Uria aalge*
European shag	*Phalacrocorax aristotelis*
Lesser sandeel	*Ammodytes marinus*
Northern fulmar	*Fulmarus glacialis*
Razorbill	*Alca torda*
Sandwich tern	*Sterna sandvicensis*

Annex 8: A note on the ecological–economic modelling of marine reserves in fisheries. *Ecological Economics*

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Ecological Economics (in press)

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A note on the ecological–economic modelling of marine reserves in fisheries

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ABSTRACT

This paper gives an overview of the bioeconomic modelling of marine reserves, and illustrates how economists have responded to the modelling results found in the ecological literature. The economic analysis is shown to be far more pessimistic with regards to the potential of marine reserves as a fisheries management tool, than what one finds in the purely ecological analysis, the reason being the latter's neglect of issues such as discounting and economic incentive behaviour. However, economic analysis, despite some of it being relatively advanced with regards to spatiality, is still simplistic with regards to for instance ecosystem and habitat content. A simple expansion of the existing bioeconomic models with regards to positive habitat effects of area closures is presented and analysed, showing room for improved results from marine reserve implementation as compared to the existing analysis.

1. Introduction

Marine reserves[1] have become one of the hot topics in marine management with proponents in most camps. Biological research on the topic has exploded since it re-emerged in the late 1980s (Conover et al., 2000), after Beverton and Holt (1957) laid closed areas on the back burner, and instead directed the focus upon gear restrictions and fishing intensity. Economic research on marine reserves is, however, comparatively new and of a much more limited quantity, though increasing rapidly. In this note the economic critique of the biological approach to marine reserves is presented, while at the same time also adding to the standard bioeconomic model in order to open for broader ecosystem considerations as demanded by ecologists.

Fisheries economists have traditionally concentrated on issues concerning management of commercially important species, and have tended to focus on one or perhaps two-species models (for some exceptions to this see Flaaten, 1988; Eide and Flaaten, 1998), leaving the broader issue of biodiversity to more general ecological–economic research. In recent years, however, there has been a growing worry that marine resources are being rapidly decimated (Botsford et al., 1997; Myers and Worm, 2003; Jackson et al., 2003). Resulting concern for the biodiversity in the oceans has awakened policy-makers as well as researchers in many fields, and the fact that *fisheries management* now must move from a single-species focus to a biodiversity focus has been underlined (Hanna, 1999). However, *fisheries economics research* does not seem to have taken this step yet. One topic that has emerged where biodiversity and commercial fisheries could meet is in management options such as marine reserves. Fisheries economists started publishing work on marine reserves towards the end of the 90s, showing an increasing interest for the topic, and often a slightly different approach and attitude to that of the ecologists. Economists have usually been more critical to marine reserves as a fisheries management option than the ecologists (see Hannesson, 1998; Smith and Wilen, 2003). However, economic analysis is still to a large degree done by applying single-species systems (see however Bonceur et al., 2002; Reithe, 2006), with issues of biodiversity or habitat seldom being included (see however

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[1] In the literature a large number of different expressions are used to describe closures of areas in the oceans. In this work marine reserves and marine protected areas (MPAs) are used interchangeably to describe permanently closed areas to some or all of human activity.

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Rodwell et al., 2003; Schnier, 2005; Upton and Sutinen, 2003). This work attempts to add to the existing bioeconomic analysis by taking into account habitat effects of marine reserve implementation. A simple habitat improvement feature is presented in an accepted bioeconomic marine reserve model, showing improved results from marine reserve implementation as compared to existing models. This accentuates the need for further integration of ecology and economics in the study of ocean management.

The advantage of reserves from a biodiversity point of view is clear — an area undisturbed will presumably over time regain some form of natural equilibrium and secure biodiversity at least within the reserve. Positive biodiversity effects for the area outside the reserve could be through migration or density dependent dispersal, or the fact that the reserve may function as a buffer to shocks to the system (see Conrad, 1999; Sumaila, 1998; Hannesson's, 2002 more critical comment). The advantages of reserves for commercial fisheries are usually presented as the increased fish production migrating out of reserves. The norm has been to focus on some form of density dependent dispersal in the economic literature (Hannesson, 1998; Sanchirico and Wilen, 1999, 2001) but also to some degree in the biological literature (Hastings and Botsford, 1999). But, as Gell and Roberts (2003) point out, very little is known regarding density dependent dispersal. The limited amount of empirical work done does however not preclude the possibility that density dependent dispersal exists, and density dependent influences are still clearly important (MacCall, 1990), but presumably more relevant as regards spawning, recruitment and cannibalism. Though some work has been done on differing habitats and marine reserves (Schnier, 2005)[2], habitat effects of reserves have received scant attention (see however, Rodwell et al., 2003; Upton and Sutinen, 2003).[3] It is clear that commercial fisheries do impact upon marine habitats (see Auster and Langton, 1999 for an overview), and this needs to be accounted for in economic analysis. In this paper a standard bioeconomic marine reserve model is expanded upon by allowing habitat enhancing effects, depending on the size of the reserve (see Halpern, 2003 for a discussion of reserve size and biological effects). These effects come in the shape of changes in carrying capacity (as suggested in Roberts and Sargant, 2002), and could well be deleterious (as described by Skonhoft and Armstrong, 2005; Armstrong and Skonhoft, 2006). The intuition behind positive effects is that closing areas to fishing allows improved habitat for protection of young and spawning, as well as increased prey abundance (Auster and Malatesta, 1995; Garcia-Charton and Perez-Ruzafa, 1999; Roberts and Sargant, 2002).

This paper first presents the ecological–economic research on marine reserves, and shows the lack of ecosystems or habitats in these models. A bioeconomic habitat enhancing model of a marine reserve is presented in the second section, followed by the results and comparison between different management options. In conclusion the paper discusses the results and reflects more generally over the integration of ecological and economic knowledge in models of marine reserves.

2. Bioeconomic modelling and marine reserves

Ecological research on marine reserves is generally more positive to this management option than economic research is, with Bohnsack (1993) summing up much of the former groups' optimism, stating that marine reserves will protect resources, enhance fisheries and even solve conflicts. Economists question all these issues as well as others in different studies which will be reviewed below.

2.1. *Discounting the future*

One of the earliest problems that economists have pointed to with regards to marine reserves is the issue of time. Also biologists have underlined the trade-off between short term profits from fisheries versus possible long term gains from risk reduction of marine reserves (Dayton et al., 2000). This is however explicitly studied in Holland and Brazee's (1996) dynamic bioeconomic analysis where the discount rate of those affected by a reserve implementation, and their minimum production requirements, are critical with regards to policy decisions on marine reserves. If society discounts the future to a large degree, possible gains from marine reserves a long time ahead will not pay for the loss in economic activity today. The consequences for policy-making are well known from the global warming debate surrounding the Kyoto Protocol. Hence the "bounce-back" propensity of chosen marine reserve areas compared to society's rate of discount, will determine whether marine reserve policy as a fisheries management option will successfully win through.

2.2. *Uncertainty*

Natures' ability to "bounce back" naturally brings us to the issue of uncertainty, which is also of interest from an economic viewpoint. It has been hypothesised that marine reserves could insure against environmental shocks, or function as a "hedge against inevitable management limitations" (Lauck et al., 1998). Sumaila (1998) applied a Beverton–Holt bioeconomic simulation model and found that the implementation of a reserve may indeed protect discounted economic rent. This model assumes, however, that the shock only occurs in the fishable area. Conrad (1999), using a Gordon–Schaefer model assumes a more general shock to the system, and finds that marine reserves do succeed in reducing biomass variation, but also reduce harvests and thereby economic rents as compared to a private property management without a reserve.

2.3. *Management options outside the reserve*

Since Gordon's (1954) seminal work, economists have criticised open access fisheries, as the effect of profits to attract fishers in unmanaged fisheries leads to excessive effort investment, which decimates the profit, even in the presence of marine reserves. In the analysis of marine reserves, many economists

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[2] Much of the bioeconomic literature on marine reserves allows for differing growth and carrying capacity, but little attention is given explicitly to this issue prior to Schnier (2005).

[3] Studies of terrestrial reserves and increased carrying capacity have however been made (see Carroll et al., 2003, Song and Li, 1995; Warkentin et al., 1995).

have pointed to this fact (Holland and Brazee, 1996), and also shown that perfectly applied private property management without reserves will better take care of both stocks and harvests, unless marine reserves are exceedingly large (Hannesson, 1998). It is clearly not fair to assume that in fisheries with a high degree of sophisticated management, the implementation of marine reserves will eradicate all other management except perhaps access (see references to limited-entry management and marine reserves below). Much work still remains, therefore, with regards to the analysis of different management options than solely open and limited access outside marine reserves.

2.4. *Reduced management costs*

It has been claimed that marine reserves in industrialised fisheries may be cheaply monitored via satellite systems, thereby reducing management costs. Bioeconomic analysis including management costs (Armstrong and Reithe, 2001) has indeed shown that the introduction of marine reserves has the potential to reduce the reserve size that will give equal stock and harvest results as that of private property management without a reserve. However, existing reserves are most common in tropical waters with greater prevalence of subsistence fisheries, where unless there is community acceptance, the monitoring costs are prohibitive. There are however very few existing reserves in fisheries where one could expect reduced management costs, such as for instance in highly industrialised fisheries with for instance satellite monitoring. We do however see increasing pressure for the implementation of marine reserves in areas with high degree of industrial fishing activity, where remote sensing and closed areas could potentially reduce management costs.

2.5. *Reduced conflict*

It has been claimed that marine reserve implementation will somehow reduce conflicts at sea (Bohnsack, 1993), presumably by zoning, and separating different interest groups in the ocean. The many conflicting opinions that have arisen prior to reserve implementation in for instance California may not be an indication of the climate after a reserve implementation, but it definitely begs the question of how a shrunken area of use will reduce conflict among a given group of agents. Sumaila and Armstrong (2006), using a one-species cohort model with two harvesting groups fishing on different age groups within the stock in question, show how there may be clear disagreement between the two groups with regards to marine reserve size and fisheries management implementation. This result is underlined in Holland's (2000) applied model of multi-species and multi-agent fisheries on Georges Bank. Via simulations it is shown that some groups of fishers obtain increased yields from permanent marine reserves, while others obtain reduced yields.

2.6. *"Double payoff"*

So far we have mainly discussed work that focuses on simple two-patch models. Sanchirico and Wilen (1999, 2001) introduce greater degree of spatiality into bioeconomic models of marine reserves. They show that economic results are highly dependent upon the type of interaction between different patches, and which patch is closed. This due to complex spatial and intertemporal effort redistribution effects. Hence both economic and biological factors affect the results of reserve implementation. The authors (Sanchirico and Wilen (1999, 2001)) identify circumstances which would afford what they call a "double payoff", namely increased aggregate biomass and aggregate harvest by closing one or more areas to exploitation. They find that, given open access outside reserve patches, biological benefits may be forthcoming from reserve implementation, but few combinations of biological and economic parameters give both increased aggregate harvests and aggregate biomass.

2.7. *Human behaviour*

A central critique made by economists of biologists' work is how the latter include human behaviour. In most biological models of marine reserves, effort is assumed exogenously given and constant (Man et al., 1995; Carr and Reed, 1993). Economists see this as an overly strong assumption and underline the incentive effects that fishers react to. Smith and Wilen (2003), use an applied, spatially and behaviourally explicit, dynamic bioeconomic model of a specific fishery, the sea urchin fishery in northern California, which they compare to a pure biological model with standard simplifying assumptions regarding human behaviour, i.e. in effect ignoring economic incentive behaviour. They demonstrate that the optimistic conclusions regarding reserves found in ecological work may well be a result of the simplified assumptions ignoring economic behaviour. Furthermore, the preferred areas to close vary greatly for the two models analysed.

2.8. *Which areas should be closed?*

What does economic research say about which fisheries and areas to close in the case of reserve implementation? Holland and Brazee (1996) show that only fisheries with a high degree of effort already present will gain from reserves. This is also a result found in Holland's (2000) applied work, as well as biological modelling (see the review in Gerber et al., 2003). Hence fisheries that manage to control effort by other means are best left alone. Sanchirico and Wilen (2001) find that given open access conditions, closing the most profitable high productivity patch provides the greatest chance of increased aggregate harvests. This is due to the fact that under open access, the most profitable area is overexploited, and hence fits into Holland and Brazee's (1996) definition of high effort levels. Introducing limited-entry management changes these results. In this case Sanchirico and Wilen (2002) show that the closure of lower productivity areas gives higher values in the fishery, as the high productivity areas under limited-entry already without reserves give large returns. The opportunity cost of closing such areas is thus high. Hence closing low productivity and high cost areas may lead to increased profitability in a limited-entry fishery. This underlines the importance of the status quo in a fishery prior to reserve implementation, and also points a finger at one of the most usual determinants for reserve area choice, namely natural productivity. Once the economics of a managed fishery is taken into account, areas less attractive are more advantageously closed. This is often in direct conflict with what we see in actual reserve management as well as suggestions from biologists (Garcia-Charton and Perez-Ruzafa, 1999).

2.9. *Bycatch fisheries*

Bycatch fisheries are by many economists seen as the type of fishery that will best gain from marine reserve implementation. This is however a topic that has not been sufficiently studied[4], though Reithe (2006) shows numerically in a two-species model that the type of ecological interaction between the two species influence the possibility of actually protecting a bycatch species through the use of a reserve. The ecological interaction also affects the possibility of obtaining a win–win situation when implementing a reserve, and also determines the optimal patch to close. Ecosystem modelling is not prevalent in bioeconomic research as of yet, but marine reserves as a fisheries management tool seems like a natural place for such research to emerge. Bonceur et al. (2002) apply a two-species, two-area model of marine reserve implementation. One stock is a commercially interesting prey species while the other stock is a non-commercial predator. They show that interactions between species may reduce benefits to the fishing industry of reserve implementation, but on the positive side create opportunities for eco-tourism development.

Applied bioeconomic modelling work on marine reserves is limited, but one of the few examples that does exist, Holland's (2000) multi-species, multi-fishery bioeconomic model of Georges Bank brings out a critique of the perhaps most often supposed positive result of marine reserve implementation, namely increased stock sizes (see however Parrish (1999) for a similar argument from a biologist). In this work he shows that in a situation where fishers react to area closures in an economically sound fashion, resulting fishing may actually increase the danger of depletion for some fish stocks. This is clearly a warning regarding how we design marine reserves.

So far, however, almost all modelling of marine reserves in economics rests upon the driving force of density dependent migration. Density dependent migration is highly probable, but nonetheless not demonstrated (Gell and Roberts, 2003). This begs the question of possible alternative advantages of marine reserves, such as how possible habitat improvement as a result of closed areas will affect the harvests and profits. Issues such as the "shifting baseline syndrome" (Pauly, 1995), and the question of whether applied carrying capacities are large enough, or should be revised for marine reserves is clearly an issue that would affect some model results. Here is a clear ecological input which may affect the bioeconomic results with regards to marine reserves. This issue is discussed in the following.

3. Modeling habitat effects of marine reserves

In economic analysis of marine reserves so far, studies have had a single-species focus (see however Bonceur et al., 2002; Reithe, 2006), and issues of biodiversity have not been given economic weight. A model that takes inherent values of ecosystems into account could be one way to remedy this, for instance by giving species biomass or density economic weight and value (Skonhoft and Johannesen, 2000, used a similar model in a study of reindeer herding). The object would be to maximize the total value of habitat use:

$$\max \Pi(h,S) = \omega \Pi(h) + (1+\omega)\Pi(S)$$

where $\Pi(h)$ are profits from harvesting h, and $\Pi(S)$ are profits and valuations tied to the biodiversity or density S of production in the relevant area. $\Pi(S)$ could be profits connected to tourism, which would be dependent upon the biodiversity or the density of natural production in the area of interest. Alternatively, or additionally, $\Pi(S)$ could incorporate non-use values (Skonhoft, 1999). The harvesting h will often be dependent upon S. ω is a weighting parameter giving the relative weight attributed to the two forms of value emanating from the habitat in question. The issues of determining ω could be solved in a way similar to Munro's (1979) method of finding the weighting parameter between the profits of two countries harvesting a common fish stock, i.e. applying for instance the Nash bargaining solution.

A few bioeconomic papers have discussed habitat quality and reserves. Schnier (2005) models heterogeneity of habitats, but does not include how the implementation of reserves can affect this quality. Rodwell et al. (2003) study the positive and negative fisheries effects as a function of time with a reserve in place; the positive effects emanating from reduced natural mortality and the negative being reduction in spatial movement out of the reserve, both due to improved habitat within the reserve. Upton and Sutinen (2003) model the reduction in the habitat growth dependent upon fishing, which feeds into intrinsic growth and carrying capacity of the fish species in question. A different model that takes into account habitat effects of marine reserves could be analysed by modifying the carrying capacity K of the logistic growth function.[5]

In the following we will study a simple model of carrying capacity being affected by reserve size[6], in keeping with existing bioeconomic models on marine reserves, hence allowing comparison.

The model is based on Hannesson's (1998) marine reserve model, though in this paper stock entities are studied, rather than densities as in Hannesson.[7] To this basic model is added a possible habitat effect of marine reserve implementation. One type of habitat effect is opened for, namely that the

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[4] Also on the biological side, models with more than one species are scarce (Gerber et al., 2003).

[5] Alternatively the intrinsic growth rate of the logistic function could be modelled as being habitat dependent, as this will have a different effect upon long run stock size and optimal harvesting effort as compared to that of the carrying capacity.

[6] Armstrong et al. (2004) analyse a hypothetical marine reserve for the North–East Atlantic cod stock applying a more complex version of the carrying capacity habitat effect presented below. They show that given modest habitat effects, a marine reserve for a migratory species such as cod may well be economically advantageous.

[7] Anderson (2002) and Conrad (1999) use models similar to the basic model applied here.

carrying capacity in the reserve depends on marine reserve size. The following four management regimes are studied;

- Open access
- Private property
- Marine reserve
- Marine reserve with habitat effects.

We will start by defining the model for the marine reserve, assuming that it is always combined with open access outside the reserve, i.e. effort enters where allowed as long as there exists positive profits in the fishery. The private property regime ensures profit or rent maximization, without a reserve. On the other end of the scale is a pure open access regime. Hence these two standard management options are compared with marine reserve management with and without habitat effects. Habitat effects as the result of the implementation of a marine protected area will be described in the following.

The resource in question is uniformly distributed over a homogeneous area. K is the carrying capacity in the total area. A fraction m, $0<m<1$, of the total area determines the reserve size, making $(1-m)$ the fraction of the total area found outside the reserve. Hence the basic carrying capacity within the reserve is mK, and the carrying capacity outside the reserve is $(1-m)K$. The rate with which fish move between the two areas, or in this context the dispersal parameter, is defined as z. It is assumed that for a positive m, i.e. reserve, there is an addition to the basic carrying capacity within the reserve, as total carrying capacity in the reserve is affected by the carrying capacity habitat effect $g(m)$.[8] We assume that $g'>0$, and $g''\leq 0$. Thus, the total carrying capacity increases for increased reserve size, but at a constant or decreasing rate.[9]This makes the total carrying capacity in the reserve equal to;

$$mK+g(m)$$

The aggregate resource biomass, if we have a biodiversity perspective, or single stock size in the simpler context, is described by X_i, with $i=M,O$, denoting the marine reserve and the outside area respectively. The rate of change in biomass in the fishable area becomes:

$$\frac{dX_O}{dt}=rX_O\left(1-\frac{X_O}{(1-m)K}\right)+z\left(\frac{X_M}{mK+g(m)}-\frac{X_O}{(1-m)K}\right)-Y, \tag{1}$$

While the rate of change in biomass in the reserve area is:

$$\frac{dX_M}{dt}=rX_M\left(1-\frac{X_M}{mK+g(m)}\right)+z\left(\frac{X_0}{(1-m)K}-\frac{X_M}{mK+g(m)}\right) \tag{2}$$

The first term on the right hand side of Eq. (1) describes the growth in the fishable area, while the second term expresses the net migration to the fishable area, and Y is the harvest. The net migration expressions are determined by the relative densities of fish in the two areas, hence a density dependent migration between reserve and outside area. The growth in the reserve area is described in Eq. (2), and is equivalent to Eq. (1), except that in the reserve there is no harvest.

Assuming p is the unit price and c/X_O is the unit cost of harvest, the profit function becomes:

$$\Phi(X_0,X_M)=\left[rX_O\left(1-\frac{X_O}{(1-m)K}\right)+z\left(\frac{X_M}{mK+g(m)}-\frac{X_O}{(1-m)K}\right)\right]\times\left(p-\frac{c}{X_O}\right) \tag{3}$$

Since we assume there is open access outside the reserve, Eq. (3) is set equal to zero, giving the standard open access stock size outside the reserve.

In the following we will assume that $g(m)=\upsilon m$, with υ being a positive constant, i.e. the habitat effects of reserve size are directly proportional to the reserve share m of the total area. For the marine reserve case without habitat effects, υ is set equal to zero.

The open access and private property regimes can be modeled as follows. The biomass is here defined as X, as we only have one area, and the other parameters are as for the marine reserve case. This gives us the objective function for the private property case:

$$\Psi(X)=rX\left(1-\frac{X}{K}\right)\left(p-\frac{c}{X}\right) \tag{4}$$

which is maximized subject to

$$\frac{dX}{dt}=rX\left(1-\frac{X}{K}\right)-Y$$

giving the standard private property results for the whole area.

For the open access case, we set Eq. (4) equals to zero, giving the standard open access results for the whole area.

4. Data

Carrying capacity is normalized; i.e. $K=1$, as is unit price, $p=1$. Intrinsic growth rate r is set equal to 0.6, and cost c is equal to 0.05.

5. Results

The results show the open access and private property regimes to be lower and upper bounds, respectively, for yield in the marine reserve case. But, once habitat effects in reserves

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[8] We assume that only the carrying capacity in the reserve is affected by the size of the area closure. Hence there is an underlying assumption that the outside area's habitat is either in such a condition or of such a character that any change in effort that may occur from reserve implementation does not affect it to any greater degree. This seems like an acceptable assumption in cases where there is extensive prior fishing activity, or where the habitat outside the reserve is of a kind that is not affected substantially by fishing effort.

[9] Carrying capacity could also be dependent upon the reserve shape, in the sense that large dispersal z may describe a reserve shape that gives a large reserve boarder line (as described by Flaaten and Mjølhus, 2005), while low z defines a more compact reserve. This may clearly be an issue with regard to protection of genetic base, in the sense that a reserve shape that allows for large dispersal out of a reserve, irregardless of size, will not to the same degree protect attributes of individual fish that may make them desirable from a commercial point of view, or for sustaining large stock levels (Trexler and Travis, 2000).

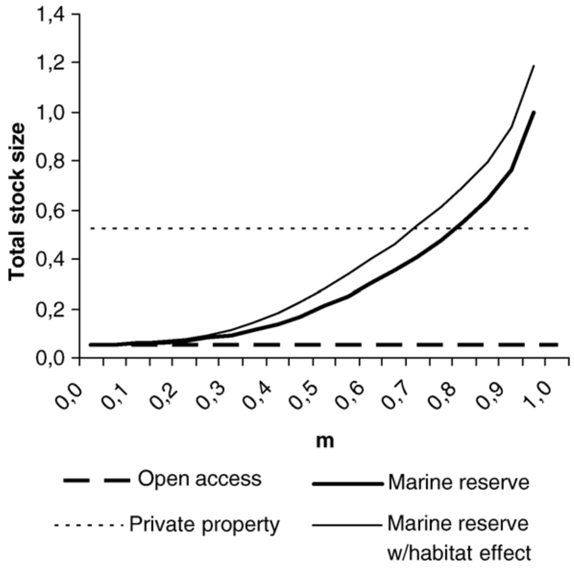

Fig. 1 – Effects on stock of varying reserve size m with carrying capacity positively affected by m ($z=0.3$, $\upsilon=0.2$).

are allowed, the private property case no longer functions as an upper bound for yield at all closure levels.

As we see in Fig. 1, the habitat effect is positive as one would expect, given that υ is positive. When compared to the marine reserve case without habitat effects, as the size of the marine reserve increases, the increasing habitat effect increases the stock. Figs. 1 and 2 are for a low habitat effect of $\upsilon=0.2$. This value is chosen as it gives the minimum value of υ for which the marine reserve can match the optimal management regime, when $z=0.3$. For $\upsilon=0.2$ the reserve must have a size of approximately 75% of the total area in order for the harvest to be as large as under the private property regime. This means that the increase in the carrying capacity in this case is 15%. By increasing υ, the minimum reserve size that matches the harvest in the optimal management case is reduced, as shown in Fig. 3. In this figure $\upsilon=0.4$ and the minimum reserve size that gives harvest matching the case of optimal management is approximately 60% of the total area, implying an increase in the carrying capacity of 24%.

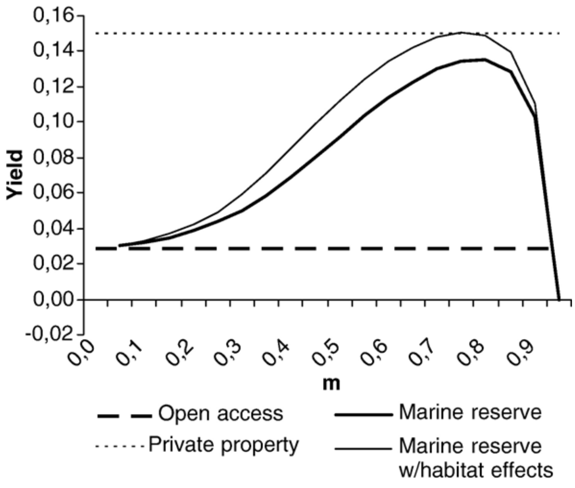

Fig. 2 – Effects on catch of varying reserve size m, with carrying capacity positively affected by m ($z=0.3$, $\upsilon=0.2$).

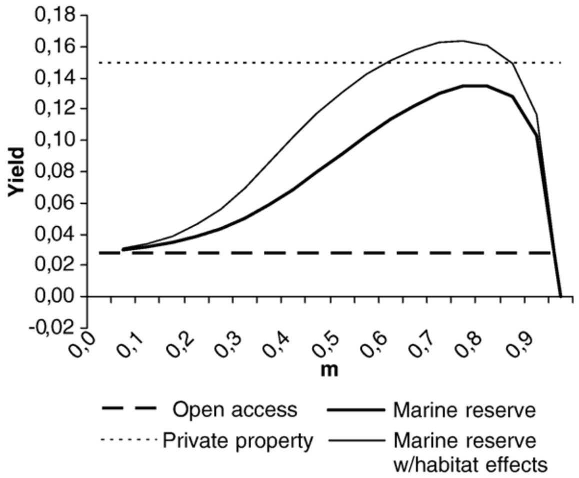

Fig. 3 – Effects on catch of varying reserve size m, with carrying capacity positively affected by m ($z=0.3$, $\upsilon=0.4$).

Applying $\upsilon=0.4$ and a 60% reserve, we observe in Fig. 4, as expected, that the stock situation is improved with carrying capacity improvement due to reserve size. This improvement wears off, however, as migration z increases. Looking at the equivalent for catch in Fig. 5, we see that harvest is not affected as drastically as the stock by increased migration, as migration allows the harvest quantity to remain high as the stock declines.

6. Conclusion

Comparing to conventional models with purely density dependent gains from marine reserves, it is shown that relatively limited carrying capacity improvements as a result of reserve size give increases in stock and harvest. Private property results are emulated for reserve sizes of approximately 75% of the total

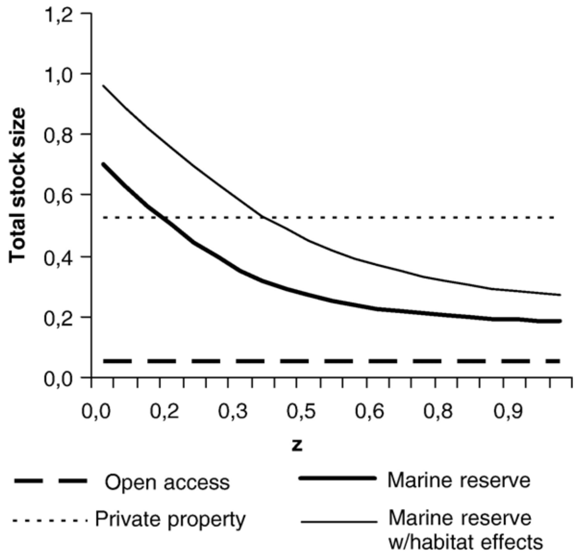

Fig. 4 – Effects on stock of varying migration rate z, with carrying capacity positively affected by m ($m=0.6$, $\upsilon=0.4$).

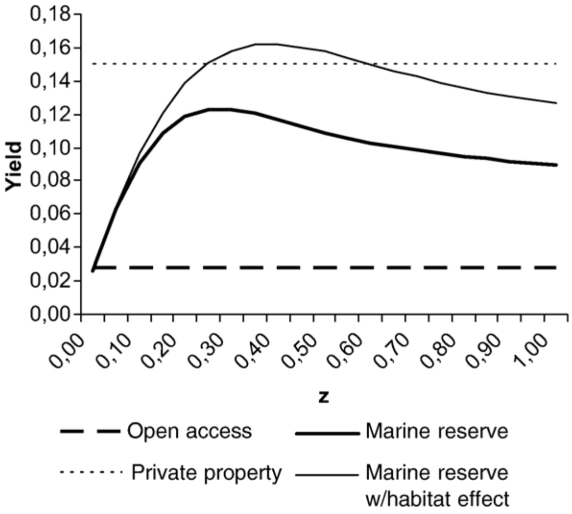

Fig. 5 – Effects on catch of varying migration rate z with carrying capacity positively affected by m ($m=0.65$, $v=0.4$).

area when carrying capacity is increased by 15%. For increasing habitat effects, the smaller is the needed reserve to match the private property harvest and stock levels.

Though it is unsure whether large carrying capacity increases as described above actually appear in marine environments, terrestrial reserves have been shown to have carrying capacities more than 20% greater per ha than outside their borders (Song and Li, 1995). Higher densities of fish within reserves than outside are found in many cases (see Attwood et al., 1997; Halpern, 2003 for overviews). It is however unclear whether this is due to reserve implementation or the fact that reserves are often established in more attractive habitats (Garcia-Charton and Perez-Ruzafa, 1999), as data has to a large degree not been available. Furthermore, when closing heavily harvested areas, the consequences for the areas outside may be detrimental, hence unless managed, reducing the carrying capacities outside the reserve. And even when increases in density within reserves are shown to follow reserve implementation, this does not necessarily imply that carrying capacity has increased, as this would depend on how large the population was originally relative to carrying capacity. Nonetheless ecological research increasingly stresses ecosystem effects on single species, and underlines the fact that these issues should be taken into account. The results given here support this, as positive habitat effects as a result of marine reserve implementation are shown to have a potential to match private property harvest. This underlines the need to take into account broader ecological effects in the ecological–economic modelling of marine reserves.

Summarising economic research so far, it is safe to say that economists in general underline the dangers of assuming marine reserves to be a panacea for fisheries management. Marine reserves must be seen as one of the many tools of fisheries management, and the how, where and for what fisheries, reserves are implemented is of great concern. Economists underline the political problems of reserve implementation, due to diverging economic interests in the relevant areas, and illustrate the sometimes conflicting economic and biological goals in the use of marine reserves in fisheries management. However, the economic analysis of marine reserves is still in its infancy, demanding much further study in a wide range of topics, where economic analysis may illustrate the shortcomings of the purely ecological modelling, and ecological modelling can illustrate where more encompassing ecological models, compared to those applied by economists, may affect the results with regard to marine reserve implementation.

Looking at fisheries management, we can see an evolutionary path leading from input controls such as gear and time restrictions to output controls such as TAC and more disaggregated quota limitations. Economists have for years (starting with Gordon, 1954) pointed to the fact that management focus purely on the biological and technological side of fisheries will result in inefficient fisheries, where the potential economic rent is eaten up by increased fishing capacity. Economists have traditionally espoused economic incentive systems for management, such as output taxation, or more commonly individual transferable quotas (ITQs). However, experience shows that such management options have not eliminated stock decline (see Iceland's cod stock decline a few years back, and the recent experiences with New Zealand's Hoki as well as other species). Indeed this was not the aim of ITQs as such, as efficiency was the main focus, and in this ITQs may well have succeeded (Hannesson, 2004). The next evolutionary step, however, seems to be a return to input control in managing the stock, but this time in a more perfect form; permanent area closures, or marine reserves. There is however a danger of marine reserves faring as poorly as ITQs. Not because either management option necessarily is detrimental, but that knowledge, control and degree of implementation is imperfect, hence leading to poor results which colour perceptions of the management option negatively (Dayton et al., 2000). Hence there is a great need for increased knowledge within a broad range of academic disciplines in order to better understand the many consequences of marine reserves.

Here it seems clear that research that is truly multidisciplinary, i.e. also with regards to the researchers collaborating would be advantageous. Economists working in the field of bioeconomics accept criticism from ecologists for their use of simplistic ecological models, such as the one applied here for expositional purposes. The economist defence is however that with the additional complexity of economics on top of for instance cohort models, the creation of a "big black box" is imminent, rendering constructive analysis minimal.[10] Hence most bioeconomic models of marine reserves are so-called Gordon–Schaefer biomass models (with some exceptions such as Holland and Brazee, 1996; Sumaila, 1998) and also most often one-species models (here Bonceur et al., 2002; Reithe, 2006 are the only exceptions as of yet), and are usually designed to analyse specific general issues. In fisheries, multi-species management issues brought to a fore that economics was needed in conjunction with biology, in order to somehow weight the different interacting species. The complexity of the biology involved, increased by the addition of economics may be why there is so little multi-species bioeconomic analysis carried out in fisheries. It is however clear, in the same way that multi-

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[10] This defence does however only seem to be valid for economists, as ecologists have to a greater degree accepted that model construction and analysis cannot be done using simple Gordon–Schaefer type models.

species approaches require greater interaction between the disciplines, ecosystem approaches have the same requirement. When bringing several, sometimes competing species, as well as habitats, into the analysis, some form of valuation is required for determining optimal human utilisation, hence economics is required. However, a limited presentation of the ecology may mask the full effects of human interactions with the ecosystem, hence requiring greater ecological involvement. It remains to be seen whether the multi-disciplinary research which one would have expected to emanate from multi-species management issues coming to a fore, will now appear with the increasing focus upon ecosystem management.

Acknowledgements

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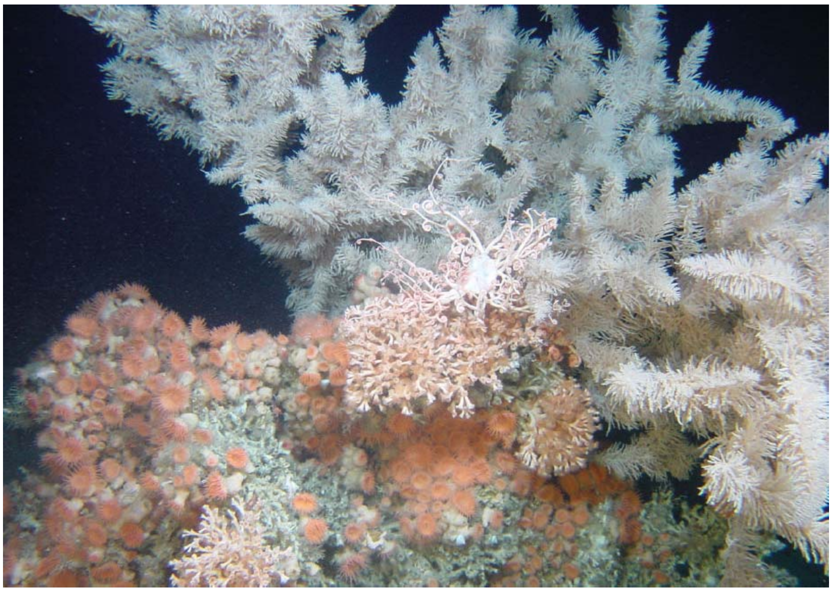
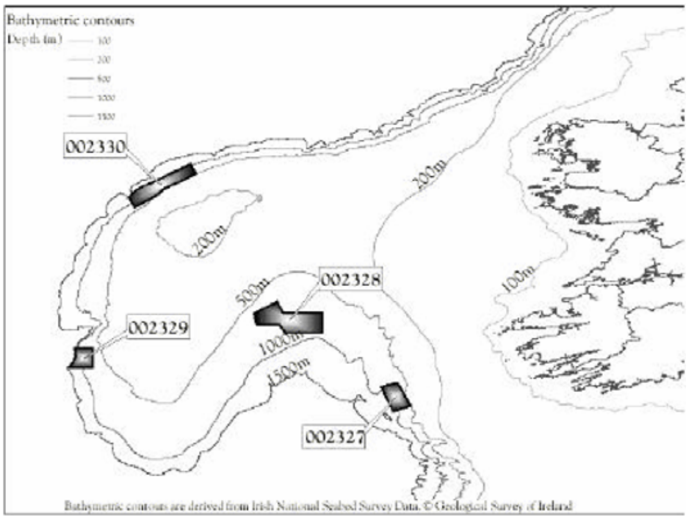
Annex 9A: Draft Environmental valuation questionnaire on Cold Water Corals



Spectacular Irish Cold Water Corals (c. IFREMER 2001)

Public Preferences for the Protection of Irish Cold Water Corals.

CEMARE and NUI, Galway



The locations of proposed Special Areas of Conservation (Habitats Directive) to protect cold water coral reefs off the west coast of Ireland. The total area to be protected is approximately 2500 sq.km.

Irish Cold Water Coral Reefs: Background

What is a deep water coral?

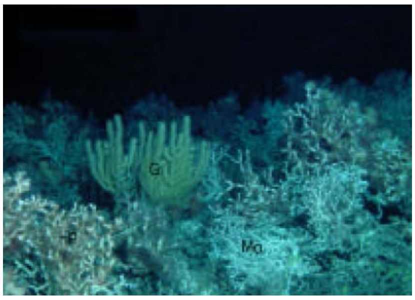


Fig.1: Deep water coral reef

Cold water corals come in many shapes and sizes from the stone corals (with external bony skeletons) to flexible soft corals. They are animals that look like plants, delicately branched or with thick tree-like stems, but they are related to sea anemones and jellyfish. They live fixed in place and feed by catching passing prey with their stinging tentacles. *Lophelia pertusa* is the best known Irish example. *Lophelia* grows slowly (less than 2 cm per annum), and reefs can take 100s to 1000s of years to grow. Cold water reefs grow much more slowly than their shallow, tropical relatives (e.g. Great Barrier Reef). Although constructed by relatively few coral species (unlike tropical reefs), these deep reefs are home to an enormous variety of associated creatures - sponges, worms, crustaceans (e.g. crabs), molluscs (e.g. clams) and fish species. The diversity rivals that of some tropical reefs.

What is the habitat like?

Irish cold water coral reefs are among the most extensive in Europe and are probably second only to Norwegian reefs in number. They typically occur along the upper part of the continental slope (see map on cover) and are associated with underwater hills called 'giant carbonate mounds'. These mounds tend to occur in clusters that extend over several thousand square kilometers. They are not formed from rock but 'grow' over 100s of thousands of years through slow accumlation of dead coral skeleton and mud with the living coral colonies on top. Coral colonies resemble bushes, with each 'bud' housing a polyp (miniature anemone). Colonies grow together to form continuous reefs.

Recent studies indicate that some mounds may be 3 million years old. Individual mounds can be 1 km or more at their base and rise up to 350m above the seafloor (about half the size of Croagh Patrick in County Mayo). They serve as a carbon sink and analysis of cores through the mounds and analysis of coral skeletons contained therein, can provide a window on past climate conditions which may have relevance for our understanding of the climate change we are experiencing today.

What species do cold water coral reefs support in the marine environment

The coral framework provides a unique habitat for many different animals. Large organisms (e.g. other corals, sponges, anemones, clams, starfish, sea urchins) live attached to the dead coral framework or rubble. Others (e.g. sponges, worms) burrow or live in cavities inside the dead coral branches or in the sediments associated with the reef. Large mobile predators (e.g. fish, crabs and lobsters) also live amongst the coral branches. In the northeastern Atlantic over 1300 species have been found to be associated with *Lophelia*. Various commercial fish species targeted by fishing are found around corals, including monkfish, blue-ling, tusk, hake, orange roughy and red crab.

What is their function (and why is it important to keep them around)?

Cold water coral reefs are unique and important biodiversity 'hotspots' in the deep-sea. They make an important contribution to Irish marine natural heritage. We know little about the potential of reefs as a source of bio-compounds for use in the biomedical and pharmaceutical industry in developing new anti-viral/bacterial drugs. In this regard, reefs are often considered the marine equivalent of rainforests. Also, reefs provide refuge for juveniles of actively fished fish species.

What are the activities that impact on the corals?

Deep-sea trawling is the activity with greatest potential to damage cold water coral - a fishery for Orange Roughy has developed since 2001 and has been responsible for destruction of reefs at a number of carbonate mounds. Other forms of fishing using gear that is not actively dragged along the bottom can have a medium to low impact. Also, oil and gas exploration/exploitation involving drilling and accidental oil spillage; laying of pipelines and communications cables and even scientific research have potential to damage reef areas.

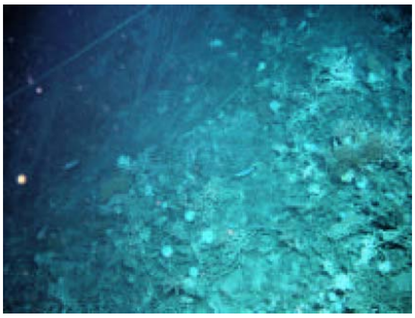


Fig.2: Lost nets on a fished reef

The Aims and Structure of this Questionnaire

In this questionnaire, we are aiming to investigate the opinions of the Irish public towards the protection of Irish cold water corals. In order to monitor and manage activities that take place in these areas, a cost will have to be incurred. This is because currently the deep water corals considered here are not monitored routinely. As the corals are in Irish waters, the initiative for monitoring and managing must come from the Irish authorities.

As explained on the previous page, various activities takes place in deep water coral reef areas (e.g. fishing, oil and gas exploration/exploitation, cable laying and pharmaceutical searches). However, in this survey we concentrate on fishing and its interaction with the coral reef areas.

What type of fishing takes place in the proposed protected areas?
Currently, fishing in the proposed areas is mainly confined to the use of static gears such as gill-netting, long-lining and crab potting. Trawling for orange roughy occurs in coral areas but this is outside of the proposed protected areas.

Is there a need to protect the fish (and will this be achieved by protecting corals)?
There are already fisheries conservation measures agreed under the Common Fisheries Policy. These measures include temporary area closures, and effort and quota restrictions. The aim of area closures to protect corals would be to protect some spawning aggregations or orange roughy.

Does scientific research cause impacts?
Little is known about the functioning of the coral ecosystem and scientific research is on-going. Some sampling of the coral is inevitable but in general non-destructive techniques (e.g. video and photos) are preferred. A Code of Scientific Research exists to guide scientific research.

Key elements of the survey:

Please note the following definitions before reviewing the example provided below

- Impact of activity – The impact of fishing activity on the habitat (i.e. the deep water corals)
- Area closed to fishing – Amount of an MPA that is closed to fishing
- Management and monitoring cost – Cost for managing and monitoring the protection of the deep water corals. This payment would be a yearly additional tax contribution per person
- Deep water coral research – Adding knowledge to the scientific understanding of the deep water corals

It is estimated that €10 per person per year would provide a very good level of management and monitoring. For other important factors not mentioned here (i.e. other environmental quality), assume that they are the same across all options.

Please remember that there are no right or wrong answers, only your personal preferences.

Example question of a Choice Set:

So, in the choices presented on the following pages, we will present to you options (or choice sets). These are made up of the elements described above. From these, we ask you to identify your preferred option.

The Questionnaire

‘Survey of preferences for the protection of deep water corals’

Section 1

In all questions presented in this section, we would like to ask you to consider,

> “which option do you prefer the most in order to protect deep water coral reefs in Irish seas”.

Attributes and levels used in the stated preference discrete choice model.

Attribute	Description	Level
Impact of activity (ACTIVITY)	What type of fishing would you permit in cold water coral MPAs?	1. Trawling 2. Less destructive fishing techniques 3. No fishing
Area closed to fishing (AREA)	Where cold water corals occur outside of existing MPAs, should there be:	1. No further protection of cold water coral areas 2. Protection of cold water coral areas not damaged 3. Protection of all cold water coral areas
Management and monitoring cost (COST)	Are you willing to pay for managing and monitoring the cold water corals? This payment would be a yearly additional tax contribution per person.	1. €0 – no additional tax 2. €1 – additional tax 3. €5 – additional tax 4. €10 – additional tax

Section 2

In the following questions, please CIRCLE the number that best describes your level of agreement.

Question #.	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
a) Before filling in this survey I was unaware of the Irish cold water corals	1	2	3	4	5
b) I feel that I am well-informed about the benefits of cold water corals	1	2	3	4	5
c) I have an interest in commercial fishing	1	2	3	4	5
d) I have an interest in recreational fishing	1	2	3	4	5
e) I have little or no interest in the marine environment	1	2	3	4	5
f) I think that the Irish public have a responsibility for the protection of the marine environment in Irish waters	1	2	3	4	5
g) I have never come across deep sea fish such as Orange Roughy, Grenadier and Black Scabbard	1	2	3	4	5
h) The Government should do more to protect the interests of Irish fishermen and fishing communities	1	2	3	4	5
i) The Government should do more to inform the public of the importance of cold water coral	1	2	3	4	5
j) I think that trawling can be a sustainable method of fishing	1	2	3	4	5
k) The Government should do more to protect the Irish marine environment	1	2	3	4	5
l) I do not understand the difference between cold water & tropical corals	1	2	3	4	5

Question #. I eat fish…

1. Never	2. Once a month or less	3. Once a week
4. Twice a week	5. Three to six times a week	6. Every day

Question #. Are you a member of any environmental or marine conservation organisations?
1. YES 2. NO

If YES, please list the environmental and marine conservation organisations to which you belong:
____

Question #. Do you or any member of your family work in the (marine) fishing industry?
1. YES 2. NO

Question #. Where did you hear about cold water corals?
TV/Radio
Newspapers
Internet

Section 3 (Information you provide in this section will remain strictly CONFIDENTIAL)

Question #. What is your age (Please circle)?
1. 18-25 years 2. 26-40 years 3. 41-60 years 4. 60+ years

Question #. Are you?
1. FEMALE 2. MALE

Question #. Do you read Playboy?
1. YES 2. NO

Question #. Do you have children?
1. YES 2. NO

Question #. What is your approximate annual household income before taxes?

1. UNDER €20,000 2. €20,001 - €40,000 3. €40,001 - €60,000
4. €60,001 - €80,000 5. €80,001 - €100,000 6. OVER €100,000

Question #. Was this survey completed by the person to whom it was addressed?

1. YES 2. NO

Please return your completed questionnaire in the reply-paid envelope as soon as possible.
(Please include any comments that you would like to make on a separate sheet?)

A summary of the results of this survey will be provided at http://www.port.ac.uk/cemare/protect

We hope you have enjoyed completing this survey, and we thank you very much for your time and interest in this study – it is greatly appreciated.

Annex 9B. Questionnaire – Seabirds, fish and MPAs

Seabirds, fish and marine protected areas

Centre for the Economics and Management of Aquatic Resources (CEMARE)
University of Portsmouth

Some ideas for the layout (trying to keep it as small as possible, double sided):

To keep:

Front page – maybe with picture(s) of birds in Scottish coasts and map of Sand-eel box (only page in colour)

Back page – Introduction to sandeel and birds, with (B&W?) pictures of activities

To return: (Four pages)

First page – outline of the aims of the survey and an example choice set

Second and third pages – the choice set questions and WTP question

Fourth page – general and socio-economic questions

Introduction to Sandeel in North Sea

Fish farming is the fastest growing sector in the world food economy, and to keep up with global demand, **industrial fisheries** (produce fish meal and oil to feed livestock) may be seriously straining the UK's marine fish stocks, and the birds that rely on them. Surveys carried out several yeas ago have revealed that North Sea sandeel numbers were just half of the 300,000 million fish required to allow fishing - a threshold set by the European Commission.

The Danish industrial fleet catches more than 90 per cent of the total allowable catch for sandeels, which was set at 660,960 tonnes in 2005.

The warming of North Sea waters is thought to be partly responsible for the decline in sandeels. Sea temperatures have risen by 1°C over the last 25 years, causing a major change in the North Sea ecosystem and delaying the appearance of the plankton (small organisms) that young sandeels feed on.

Sandeels form a key component of the marine food web, providing food for seabirds, porpoises and fish such as cod and mackerel among others. The sandeel shortage was thought to be a major factor in seabird breeding failures along the UK's North Sea coast in 2004, which turned out to be the worst breeding season on record.

Investigations into the effects of the fishery on the seabird populations and the effect of creating a marine protected area (MPA) and closing the area for sandeel fishing are presently being carried out. The idea is that overall sustainability of the populations of both sandeel and its predators can be achieved through the maintenance of a dynamic matrix of permanent or rotating protected local spawning aggregations, which ensure sources of larval recruitment to nearby areas.

The PROTECT Project

A very brief description.

The aims of this Questionnaire

In this questionnaire, we are trying to identify the opinions of the Scottish public towards the protection of birds (Kittiwake) along Scottish coasts by creating MPAs for Sandeels. The creation of monitoring and management activities that take place in these areas will inevitably mean that costs will be incurred with the cost necessarily met by Scottish authorities. A maximum of £10 per person per year is envisaged although it could possibly be less. In the questionnaire we ask you to help identify the preferred management and monitoring combination and level of cost you regard as acceptable for each combination. You will be given a number of options (or “choice sets”) from which we would like you to identify your preferred option. We give an example at the bottom of this page.

Please note the following definitions before reviewing the example provided below (Remember that sandeel scenarios deal with the protection of birds in Scottish coasts)

BIRD – The impact of fishing activities on birds.
FISH – The level of fishing allowed in the sandeel box
COST – Cost for managing and monitoring the protection of the bird population. This payment could be a yearly additional tax contribution per person

There are no right or wrong answers, only your personal preferences.

Example Question of a Choice Set:

T.B.A.

Section 1: Part A: Survey of preferences for the protection of sandeel for birds

Please select your preferred option from these below.
Attributes and levels used in the stated preference discrete choice model.

Attribute	Description	Level
Reduction of breeding success (BIRD)	Breeding success; number of chicks	1. lots 2. some 3. few
Threshold level of fishing (FISH)	Fishing boats allowed in the sandeel box	1. Unrestricted 2. Ten boats 3. One boat 4. None
Management and monitoring cost (COST)	Cost for keeping this area closed per person per year	1. £25– additional tax 2. £10 – additional tax 3. £0 – no additional tax
Sandeels use in fishmeal	Is it important that Northsea sandeels are available to be fed to farmed Scottish salmon	1. Yes 2. No

Section 1: Part B: Survey of preferences for the protection of sandeel for birds

Please also answer the following direct question on your willingness to pay (WTP) for management and monitoring of sandeels through a proposed MPA:

a). Would you be willing to pay **£5** as additional tax for management and monitoring of birds?
Yes / No (Please circle)

If the answer is **Yes** to a):
Would you be willing to pay **£10** as additional tax for management and monitoring of birds?
Yes / No (Please circle)

If the answer is **No** to a):
Would you be willing to pay **£1** as additional tax for management and monitoring of birds?
Yes / No (Please circle)

If **No**, Why?

____

____

Section 2: Socio-economic factors and attitudes

To help us analyse the results from the previous questions, please could you also answer the following question for us. For each of the following statements, CIRCLE the number that best describes your level of agreement

	Strongly disagree	Disagree	Neutral	Agree	Strongly agree
a) I would like to see a growing sea bird population along Scottish coasts	1	2	3	4	5
b) I feel that industrial fisheries negatively impact the bird population	1	2	3	4	5
c) I have an interest in commercial fishing	1	2	3	4	5
d) I have an interest in recreational fishing	1	2	3	4	5
e) I have little or no interest in the marine environment	1	2	3	4	5
f) I think that the Scottish public have a responsibility for the protection of the marine environment around Scottish waters	1	2	3	4	5
g) I had never heard of sandeels or kittiwake birds before this survey	1	2	3	4	5
h) The Government should do more to protect the interests of Industrial fishermen and fishing communities	1	2	3	4	5
i) I believe the fishing sandeels affects the marine food web	1	2	3	4	5
j) The Government should do more to protect the marine environment in Scottish waters	1	2	3	4	5
k) I believe that the warming of the north sea is a major cause for decline in sandeels	1	2	3	4	5

Question #. I eat fish… (Please circle only one)
1. Once a week
2. Once a month
3. Occasionally
4. Never

Question #. I eat farmed salmon… (Please circle only one)
1. Once a week
2. Once a month
3. Occasionally
4. Never

Question #. Are you a member of any environmental or marine conservation organisations (Please circle)?

1 YES
2 NO

If YES, please list the environmental and marine conservation organisations to which you belong:

____

____

Question #. Do you or any member of your family work in the (marine) fishing industry (Please circle)?

1 YES
2 NO

Question #. Are you a regular bird watcher (Please circle)?

1 YES
2 NO

The information that you provide from the following questions will remain strictly CONFIDENTIAL and you will not be identified from your answers.

Question #. Please indicate your age (Please circle)?
18-25 years
26-40 years
41-60 years
60+ years

Question #. Are you (Please circle)?
1 FEMALE
2 MALE

Question #. What is your approximate annual household income before taxes (Please circle)?
1. UNDER £10,000
2. £10,000 - £20,000
3. £20,001 - £30,000
4. £30,001 - £40,000
5. £40,001 - £50,000
6. OVER £50,000

Question #. Postcode where you live?
____

Do you have any comments that you would like to make?

A summary of the results of this survey will be provided at the following website: http://www.port.ac.uk/cemare/protect

Please return your completed questionnaire in the reply-paid envelope as soon as possible.

Thank you very much for your time and interest in this study – it is greatly appreciated.

CEMARE.