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Discarding Discards: Identification of influential factors and possible mitigation tools in demersal trawl fisheries

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PhD Thesis

Written by Jordan Paul Feekings
Defended 1 October 2012

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For it is a universal law that the sea and its use is common to all . . . For everyone admits that if a great many persons hunt on the land or fish in a river the forest is easily exhausted of wild animals and the river of fish, but such a contingency is impossible in the case of the sea.

Hugo Grotius

Mare Liberum

1609

Preface

The present thesis is submitted in partial fulfilment of the requirements for obtaining a Doctor of Philosophy (Ph.D.) degree. The thesis consists of a review and four supporting papers. Three papers are published and one is a manuscript ready for publication.

I wish to express my sincere gratitude to my supervisor: Dr. Niels Madsen from the Technical University of Denmark – National Institute of Aquatic Resources (DTU Aqua). I would also like to express a special thanks to Valerio Bartolino, Peter Lewy, Tom Catchpole, Jan Jaap Poos, and Geert Aarts for their contributions to the thesis. Further, I would like to thank all my highly appreciated colleagues at DTU Aqua for many inspiring talk and discussions.

Fisheries observers, too numerous to mention individually, collected almost all of the data used in this dissertation as part of the Danish discard observer programme. I owe a debt of gratitude to Helle Andersen for helping me extract the data, and spending countless painstaking hours with me correcting and validating the data. Without those data this dissertation would not exist.

Last but not least, I would like to thank my dear partner Karoline for her support, especially during the final months.

Hirtshals, July 2012

Jordan Paul Feekings

"The trawler, of course, has long been the recognised piscatorial scapegoat, reviled by the inshore line-fisherman with an energy which is usually in inverse ratio to that with which he pursues his own calling, and condemned with scant ceremony by the amateur " naturalist " and the public at large" (Holt, 1895).

"A brilliant suggestion that the capture of undersized fish should be prohibited need not detain us long, since it is obviously impossible to avoid catching some undersized fish if one fishes at all, and what benefit could be expected from a legal prohibition of this sort I am at a loss to conjecture, since the law could not possibly be enforced as long as a fisherman was allowed to go to sea. There are, of course, methods by which the capture of a very large proportion of undersized fish can be prevented, but prohibition of capture, per se, is not one of these" (Holt. 1895).

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List of papers

Paper I: Feekings, J., Bartolino, V., Madsen, N., and Catchpole, T. 2012. Fishery Discards: Factors Affecting their Variability within a Demersal Trawl Fishery. PLoS ONE, 7(4): e36409. http://dx.doi.org/10.1371/journal.pone.0036409

Paper II: Feekings, J., Madsen, N., and Lewy, P. 2012. The effect of regulation changes and influential factors on cod discards in the Baltic Sea demersal trawl fishery. Canadian Journal of Fisheries and Aquatic Science, 70(4): 534−542. http://dx.doi.org/10.1139/cjfas-2012-0273

Paper III: Feekings, J., Aarts, G., Poos, JJ., Madsen, N., van Helmond, A., Catchpole, T., Rochet, M-J., Pout, A., Ulleweit, J., Vandemaele, S., Ulrich, C., and Kempf, A. 2012. Spatio-temporal variability of North Sea cod discards. Manuscript.

Paper IV: Madsen, N., Feekings, J., and Lewy, P. 2012. Discarding of plaice (*Pleuronectes platessa*) in the Danish North Sea trawl fishery, Journal of Sea Research, 75: 129−134. http://dx.doi.org/10.1016/j.seares.2012.05.005

Dansk resumé (abstract in Danish)

Forståelsen af hvad der fører til discard (udsmid) af fisk kan bidrage til at forbedrede forvaltningsmetoderne i fremtiden. Ændring af forvaltningsmetoder for at reducere discarden kan vise sig at være gavnlig ikke kun for fiskeriets levedygtighed, men også for økosystemets biologiske funktion. Nærværende Ph.D. afhandling med titlen "Forhindring af discard: Identifikation af betydningsfulde faktorer og mulige forbedrede værktøjer indenfor fiskeri med bundtrawl", består af en sammenfatning samt 4 artikler, der beskriver problemer og foreslår forbedrede forvaltningsmetoder, som kan reducere discard.

Artikel I var en undersøgelse af de faktorer, der potentielt kan påvirke discard af kommercielt vigtige arter i Kattegat. Tidligere studier, som har undersøgt de faktorer, der påvirker discard, har fortrinsvis fokuseret på den bortkastede mængde som en helhed uden hensyntagen til, at discard over og under mindstemål (MLS) fremkommer af forskellige årsager. Undersøgelsen dokumenterede, at de faktorer, der har betydning for discard, var forskellige for de to undergrupper (over og under mindstemål) samt for de forskellige arter.

Artikel II fokuserede på discard i Østersøens torskefiskeri. I løbet af de seneste 15 år er der udført et omfattende arbejde for at forbedre selektiviteten med redskaber og dermed mindske discard. Dette studie undersøgte: i) De virkninger, som tekniske foranstaltninger, nemlig selektivitet med udstyr og mindstemål, havde på discard og; ii) En lang række faktorer, som kan påvirke discard og kan sløre en potentiel virkning af forbedret selektivitet. Resultaterne viste, at når lovmæssige ændringer af udstyr er gennemført korrekt, er de effektive forvaltningstiltag. Det skal dog nævnes, at effektiviteten påvirkes af en bred vifte af faktorer, som, hvis disse ikke er indregnet, kan fordreje en potentiel effekt på forbedret/hæmmet selektivitet.

Artikel III samlede discard data fra Nordsøen i perioden 2003-2010 fra i alt 11.370 fiskerier indsamlet i syv EU medlemslande. Viden om spatio-temporal discardmønstre er bydende nødvendige for forskere og myndigheder, men mangler ofte. Her blev den rumlige og tidsmæssige fordeling af torskediscard i hele Nordsøen analyseret og sammenholdt med hovedårsagerne bag forekomsten af discard. Det diskuteres, hvordan sådan viden kan anvendes for a forbedre fremtidige fiskeriaktiviteter og efterfølgende fangstsammensætning under et discard forbud.

Artikel IV beskriver discard af rødspætter (*Pleuronectes platessa*) i Nordsøen. Rødspætter spiller en vigtig rolle i Nordsøens havbunds økosystem, idet de er en af de mest talrige arter inden for fladfisk og en af de vigtigst arter inden for fiskerierhvervet. Ikke desto mindre er rødspættefiskeriet i Nordsøen karakteriseret ved en stor grad af discard, hvor ca. 50 % (efter vægt) rødspætter discardes. Her beskrives de generelle mønstre i disse data med særlig fokus på faktorer, der kan være vigtige for forvaltningsstrategier i fremtiden.

Abstract

Discarding of aquatic organisms is a global problem in the world's fisheries, where more than 7 million tonnes are caught and subsequently discarded each year. The understanding of what drives discarding can help provide mitigation measures in the future. Altering management measures which result in high discard rates/ratios may prove beneficial not only to the economic viability of the fishery but also to the biological functioning of the ecosystem. The present Ph.D. thesis, titled "Discarding Discards: Identification of influential factors and possible mitigation tools in demersal trawl fisheries", investigates discarding practices in demersal trawl fisheries and identifies possibilities for reducing discards. In focus, the factors that determine discards, including environmental factors, fishing methods, management regulations, and biological factors have been analysed. This includes an examination of the efficiency of technical regulations currently in force and retrospective analyses of the efficiency of such measures in the past. The thesis consists of a review and 4 papers.

Paper I is an investigation of the factors that can potentially influence the discarding of commercial species in the Kattegat. Previous studies that have investigated the factors that influence discarding have typically focused on the discarded portion as a whole, without considering that discards above and below minimum landing size (MLS) occur for different reasons. The study documented that the factors influential to discarding were different for the two subgroups (under and over MLS) and also for the different species.

Paper II focuses on discarding in the Baltic Sea cod (*Gadus morhua*) trawl fishery. Over the past 15 years extensive work has been conducted to improve the selectivity of the gears and subsequently reduce discards. This study investigated: i) the effects that technical measures, namely gear selectivity and minimum landing size (MLS), had on discards and; ii) a wide range of factors that can influence discards and may blur a potential effect of improved selectivity. The results showed that when gear regulations are implemented correctly they are an effective management measure. However, their effectiveness is influenced by a diverse range of factors that if unaccounted for may distort a potential effect of improved/hampered selectivity.

Paper III compiles discard data from 11370 fishing events collected across seven European Union (EU) Member States for the North Sea over the period 2003-2010. Knowledge about the spatio-temporal nature of discards is imperative to researchers and regulators but is often lacking. Here we analysed the spatial and temporal distribution of cod discards throughout the entire North Sea together with the main driving factors behind its occurrence. We discuss how such information can be used to improve future fishing activities and their subsequent catch compositions under a discard ban.

Paper IV describes the discarding of plaice (*Pleuronectes platessa*) in the North Sea. Plaice play an important role in the North Sea benthic ecosystem, being one of the most abundant flatfish species and one of the most important species for the fishery. Nevertheless, the plaice fishery in the North Sea is characterised by a high discard ratio, where approximately 50% (by weight) of plaice are discarded. Here we describe the general patterns in these data with particular focus on factors that could be important for management strategies in the future.

1. Introduction

Discarding is a global problem. Throughout the present thesis, discards are defined as the part of the catch that is brought on deck only to be returned to the sea. The practice of discarding occurs in almost all developed fisheries worldwide (Kelleher, 2005). In the past, total estimates have been as high as 27 million tonnes annually (Alverson et al., 1994). The most recent assessment was made for the period 1992-2001, where the average yearly estimate of discards in the world's marine fisheries was estimated to be 7.3 million tonnes, approximately one tenth of total recorded landings (Kelleher, 2005). Demersal trawling is the most problematic form of fishing with respect to discards (Hall and Mainprize, 2005); accounting for approximately 22% of the world's total landings while 50% of the total estimated discards (Kelleher, 2005). Within Europe, 60–70% of discarded resources are roundfish and flatfish species which arise mostly from the roundfish, flatfish and Norway lobster (*Nephrops norvegicus*) directed demersal trawl fisheries (Catchpole et al., 2005a). Consequently, the present thesis focuses on discards within demersal trawl fisheries.

The issue of discarding would not be of great concern if discard mortality was low. However, this is often not the case (Evans et al., 1994). Discarding can contribute a substantial component of fishing mortality (Borges et al., 2005a). Subsequently, the issue of discarding is of concern to the industry and sustainable exploitation of the stock. There are many sources of additional mortality that are associated with the capture process, however, these are generally not as high as in discards. Furthermore, discarding has wider implications whereby ecosystem functioning and its biodiversity are negatively affected (EC, 2007). There are indications that discarding has altered the ecosystem functioning of some seabird communities (Votier et al., 2004; Votier et al., 2010) and has negative effects on a variety of endangered, threatened, protected and charismatic non-target species such as seabirds, sea turtles and marine mammals (Alverson et al., 1994; Suuronen et al., 2012). The European Commission (EC) considers the practice of discarding to be negative, both in terms of ecosystem functioning and economic viability, and is committed to eradicating the problem (EC, 2007).

High discard rates have been reported in many trawl fisheries worldwide, with Danish fisheries being no exception. The proportion of discards within Danish demersal trawl fisheries has been estimated to be approximately 46 % for Norway lobster, 52 % for cod (*Gadus morhua*), and 64 % for plaice (*Pleuronectes platessa*) (in numbers; Andersen et al, 2005). Discards of low valued species are often much higher than what is observed for the main commercial species (e.g. 99 % of whiting (*Merlangius merlangus*) and dab (*Limanda limanda*), and 92 % of haddock (*Melanogrammus aeglefinus*), in numbers, were discarded in the Kattegat Danish demersal trawl fisheries in 2002; Andersen et al., 2005). The perceived loss to the stocks is generally less when discarded weights are concerned. This is because it is often juveniles under minimum landing size (MLS) that are discarded. Due to the complexity of the system, factors that drive the practice of discarding are numerous and occur for many different reasons.

Discarding is influenced by various economic, sociological, technical, legislative, environmental and biological factors. The myriad of factors driving discards form a complex network of often interwoven causes and effects, e.g. regulations that govern fisheries, and markets, often create a complex web of incentives and disincentives that drive the discarding practice of fishers (Jennings and Kaiser, 1998; Viana et al., 2011). In order to be able to propose solutions, the driving factors need to be identified (**Paper I; Paper II; Paper III; Paper IV**). The effect and relative importance of these factors will vary for different species, vessels, and fleets, and will fluctuate over time and space. Subsequently, it is important to define such driving factors for each individual fishery and area.

The most common means of addressing discards is either through improvements to gear selectivity or temporal and spatial closures (Davis, 2002; Cook, 2003; Valdemarsen and Suuronen, 2003; Broadhurst et al., 2006). Improving the species or size selectivity can not only reduce discards but also improve the yield in a fishery. Subsequently, large efforts have been spent developing and testing gears with improved selectivity parameters. The Baltic Sea demersal trawl cod fishery is probably one of the fisheries where the highest number of selective devices have been implemented (Madsen, 2007). Despite this, the changes to gear selectivity which have been implemented in the Baltic Sea demersal trawl cod fishery have never been analysed to determine whether they have had the intended effect on the stock (**Paper II**).

Knowledge about the spatio-temporal nature of discards is imperative to researchers and regulators (Dunn et al., 2011). However, such information is often lacking (Viana et al., 2011). This is largely due to the sparse nature of the data, often covering less than 1% of total fishing effort. Consequently, little knowledge exists regarding the spatial and temporal dispersal of discards (**Paper III**). **Paper III** focuses on strengthening knowledge about the spatio-temporal distribution of discards (cod) throughout the North Sea by combining national sampling programmes. The methods presented here can be adapted and applied to other areas, fleets and species.

In the present review, the data and collection methods are described, the magnitude of the problem is highlighted, factors that describe why discarding occurs are outlined and different methodologies used to reduce discards (rates and totals) are defined. Finally, the current issues of discards within the European Union are discussed.

2. Definitions

Throughout the present thesis, discards are defined according to the FAO definition:

"Discards, or discarded catch, is that portion of the total organic material of animal origin in the catch, which is thrown away, or dumped at sea for whatever reason. It does not include plant materials and post-harvest waste such as offal. The discards may be dead or alive" (Kelleher, 2005).

Bycatch is referred to as organisms that are not the primary fishing target. This includes small individuals of the target species, or other species with little or no commercial value.

Discard rate. Many publications refer to discard rates as the ratio of discards to total landings (Kelleher, 2005; Rochet and Trenkel, 2005). Herein we refer to discard rates as either the number or weight per unit effort (DPUE).

$$Discard\ rate\ (DPUE) = \frac{discards}{effort}$$

Discard ratio throughout this thesis refers to the proportion (percentage) of the total catch that is discarded.

$$Discard\ ratio\ (\%) = \frac{discards}{discards + landings} \times 100$$

Total discards are referred to as the total weight of individuals or a species discarded within a fishery or area.

High-grading is defined as the practice of preferentially discarding lower valued (generally smaller) individuals over the MLS in order to maximize the value of a quota.

3. Effects of discarding (The issue of discarding)

Awareness of the extent of the problem and the need for mitigation was initially provoked by the bycatch of charismatic and endangered species such as dolphins, porpoises and sea turtles (i.e. the vaquita porpoise; Rojas-Bracho and Taylor, 1999). Awareness spread rapidly to other species, especially juvenile fish caught in shrimp trawls (Kennelly and Broadhurst, 2002). Subsequently, large efforts have been spent on ways of reducing discards and mortalities of fish escaping from fishing gear (Goncalves et al., 2008).

3.1.   Mortality of discards and escapees

If the mortality of discarded individuals is low the issue of discarding becomes less of a concern (Mesnil, 1996). However, in many circumstances this is not the case and the mortality of discarded individuals can represent a significant portion of total fishing mortality (van Beek et al., 1990; Evans et al., 1994; Kaiser and Spencer, 1995; Borges et al., 2005a; Yergey et al., 2012). Therefore, the issue of discarding is of important concern to the industry and the sustainable exploitation of the stock (Alverson et al., 1994; Crowder and Murawski, 1998; Rijnsdorp et al., 2007; Aarts and Poos, 2009). Accounting for the survival of discards translates into a reduction in the estimated fishing mortalities (Mesnil, 1996). Therefore, assessment methods must be expanded to account for survival of discards, or at least to test how significant its effects are in view of the generally large noise and variability in the discard data (Mesnil, 1996). There are many sources of additional mortality that are associated with the capture process, however, these are generally not as high as discard mortality. The mortality of discarded individuals is an important issue in fisheries management and, because it is generally unmeasured, represents a large source of uncertainty in estimates of fishing mortality worldwide (Davis, 2002). To achieve reductions in discard mortalities the key influential factors of why discarded fish die need to be identified at a species and fishery level (Davis, 2002; Broadhurst et al., 2006).

The damage and mortality of discarded organisms from fisheries using towed gears is rarely attributed to a single cause but more often to a combination of the numerous interacting factors (Broadhurst et al., 2006; Benoit et al., 2010) which can be grouped into several classes. These classes include technical factors (gear type, catch volume and composition, towing speed, haul time and duration, time on deck, handling procedures), environmental conditions (water and air temperatures, light conditions, anoxia, sea conditions, depth of capture), and biological attributes (fish size and species, behaviour, and physiology) (van Beek et al., 1990; Wassenberg and Hill, 1989; Chopin and Arimoto, 1995; Richards et al., 1995; Mesnil, 1996; Davis and Olla, 2001; Davis, 2002; Broadhurst et al., 2006; Benoit et al., 2010; Yergey et al., 2012). Despite the numerous factors that can influence discard mortality the largest sources are from predation by seabirds and midwater/ bottom-dwelling scavengers (Wassenberg and Hill, 1990; Hill and Wassenberg, 1990). Approximately 57-70% of discarded animals are taken by seabirds (Blaber and Wassenberg, 1989; Berghard and Rosner, 1992; Evans et al., 1994; Catchpole et al., 2006). For sedentary animal, such as Norway lobster, the distance from fishing ground can also

have an effect on survival due to the potentially unsuitable habitat (Evans et al., 1994). An additional source of mortality for crustaceans is associated with their shell durability, and subsequently the stage of moult (Stevens, 1990; Broadhurst et al., 2006). Further factors that have been recognized to potentially influence discard mortality include inherent biological differences between sexes, the presence or absence of a closed swim bladder, and ongoing mortalities caused by infection, predation or the ability to feed (Broadhurst et al., 2006; Yergey et al., 2012).

Solutions to mitigating discards have focused on increasing the escape of live, unwanted organisms during fishing through measures such as avoiding areas containing potential discard, modifying fishing gears in ways that reduce discard capture, and allowing for potential discards to escape through grids, panels, or increased mesh sizes (e.g., Kennelly and Broadhurst, 1995; Broadhurst, 2000; Davis, 2002). However, through changes to operational and/or on-board handling techniques, it may be possible to further mitigate the mortality of discards. Two of the simplest changes to on-board handling procedures include reduced air exposure and regulated temperature (Broadhurst et al., 2006). For management of fisheries resources, measures that improve survival of discards or escapees are likely to be more acceptable to fishers than traditional technical measures such as increasing mesh size (Mesnil, 1996; Goncalves et al., 2008). Irrespective of the actual changes to operational and/or on-board handling techniques, like all modifications to gears, these need to be practical, easily regulated and demonstrated to clearly mitigate unwanted fishing mortalities. It should be possible to reduce some component of discard mortality, although, because of the cumulative stress on an organism during catch and discarding processes, wherever possible, the escape of unwanted organisms should be promoted during fishing (Broadhurst et al., 2006). This is because the mortalities of discards are considerably greater than escapees for a majority of species (Broadhurst et al., 2006).

Many of the biological, environmental and technical factors affecting escapee mortality include components of those already described above for discards. As with discards, various interacting factors contribute towards escape mortalities (Davis, 2002) and these factors may differ across species (Bjordal, 1999). However, escapees are not subjected to considerable additional cumulative stress associated with being brought to the surface, exposed to air, thrown from the vessel and then sinking or swimming back to their habitats (Broadhurst et al., 2006). Despite this, an additional suit of factors influence escape mortality and include the size of the catch and its composition, water temperature and its effects on physiological and behavioural responses, availability of light (and/or diurnal effects), sea state, and mesh size and shape (Suuronen et al., 2005; Broadhurst et al., 2006; Suuronen and Sardà, 2007). The mortality of escapees can also differ depending on when during the tow the individual escapes. Escape during hauling causes additional stress and physical damage; therefore, the mortality is expected to be higher (Madsen et al., 2008a; Madsen et al., 2008b). To facilitate escapement at depth, selective devices, such as grids and windows, appear to be more appropriate than changes in mesh size or mesh configuration if escapee mortality is wanting to be reduced (Madsen et al., 2008a, Madsen et al., 2008b; Grimaldo et al., 2009). For some species, escape from selective devices can cause less damage and mortality

than escape through meshes (Suuronen et al., 1996). To ultimately validate gear selectivity improvements, the mortality of organisms escaping various selective devices during fishing needs to be quantified. Unless escape mortality is low, technical solutions intended to improve selection may not be justified (Broadhurst et al., 2006).

3.2. Ecological effects of discarding

Apart from the ethical issues, the practice of discarding is known to threaten endangered species, damage habitats, impact the food web, and affect ecosystem function and biodiversity (Alverson et al., 1994; Votier et al., 2004; EC, 2007; Votier et al., 2010; Zhou, 2008; Suuronen et al., 2012). There are indications that discarding has altered the ecosystem functioning of some seabird communities (Votier et al., 2010; Votier et al., 2004) and has negative effects on charismatic and endangered species (Alverson et al., 1994). An additional concern is that trawling inflicts major damage to the ecology of the seabed. It not only causes physical damage to the substrate (e.g. de Groot, 1984; Hutchings, 1990; Daan, 1991; Bergmann and Heep, 1992), but also removes large quantities of organic matter from the seafloor to the surface where a majority of it is either removed as human food or, in the case of most discards, as food for surface scavengers. Relatively little of it returns to the seafloor, dead or alive (Evans et al., 1994). Including discards in assessments is crucial for accurate evaluations regarding the ecosystem effects of fishing (Anon, 1992; Mesnil, 1996).

3.3. Economic effects of discarding

The biological and environmental impacts of discarding are of most concern to fishery managers, NGO's and the general public. However, discarding also generates direct and indirect economic problems for the fishing industry. Pascoe (1997) suggests that the economic impacts of discarding can be classified into four categories:

- Forgone income associated with discarding juvenile and adult target species,
- Inter-fishery costs associated with discarding juvenile bycatch species,
- Costs associated with discarding non-commercial species; and
- Costs associated with measuring/ estimating the level of discards.

4. Discard data

4.1. Discard sampling

The present thesis is based on data collected as part of the Danish at-sea sampling programme that began in 1995 and continues to the present day (Fig. 1). Since 1995, Denmark has collected data on catches and discards with the purpose of estimating the quantity discarded. The programme aims to collect information on discards from all demersal fisheries except the ones with very limited fishing effort and discard. In 2002 the EU identified the need to describe and quantify discards as part of the European Data Directive (1639/2001 and 199/2008). The data collected are stratified with regards to ICES area, quarter, and discard pattern of the relevant fisheries (e.g. fisheries with low discards are seldom sampled). According to the DCF (949/2008), from 2008 onwards sampling is done by metier (a combination of fishing ground, gear, target assemblage, and mesh size). Participation in the discard sampling programme is opportunistic, i.e. permission by the skipper is required, and as the observer has no relation to the control unit, the fishing practice is assumed to be unaffected by the observers presence. In order for the sampling programme to be representative of the fisheries in question, vessels of all sizes are sampled from all the main fishing harbours during the entire period of activity of a given fishery. Biological information (i.e., lengths, weights and otolith samples) are collected from the catch, together with vessel, gear, geographical position and environmental attributes (depth, bottom type).
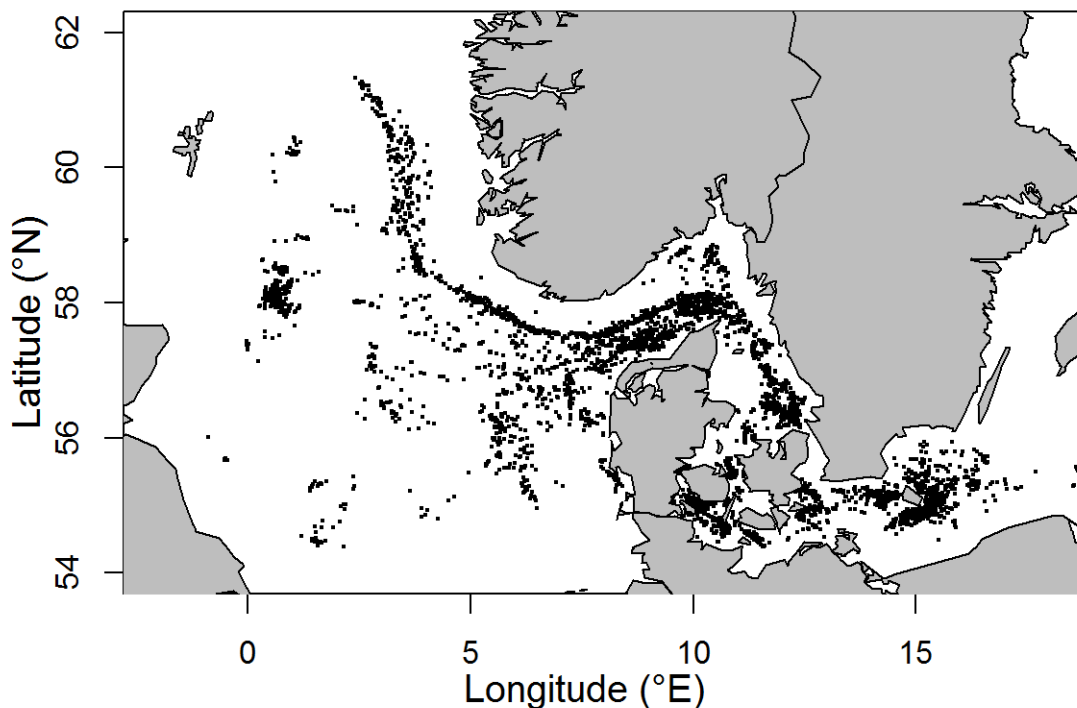

Fig. 1. Locations of hauls (demersal trawls) in the Danish at-sea-sampling programme 1997 – 2010.

For each observed haul, an approximation of the total catch weight is made by the fisher and the observer in collaboration. The total catch is then sorted into the retained and discarded components by the commercial fishers. The total weights of each individual species retained are recorded. If the abundance of a species is small, total numbers and lengths are recorded, otherwise a subsample is taken, numbers and lengths recorded and raised accordingly. The total weight of the discarded portion is approximated, a subsample taken, and then sorted by the observer into species. Total weights and numbers of each discarded species in the subsample are determined and raised based on the total approximated discarded weight (**Paper I**).

The data that are collected as part of the Danish at-sea sampling programme are the only data that provide a complete overview of the total catch; discards and landings. The demersal trawl fisheries have the most comprehensive discard sampling in relation to other fisheries, namely Danish seines and static gears (principally gillnets) due to the high discard quantities that occur in these fisheries. Observer programmes are often associated with high operating costs. Due to the high costs associated with collection of data through the use of on-board observers, coverage is often low (~ 1 %). In 2010, approximately 8% (67 of 853) of vessels active within Danish fisheries were sampled in the observer programme (Storr-Paulsen, 2011).

Despite the best efforts to create a statistically sound sampling programme, there still exists bias in the data collected. Observer programmes are considered biased if the data are not representative of the fleets and their fishing operations. To avoid a systematic bias in the selection procedure, random selection should be applied. This is not the case in many sampling programmes, as they are often stratified with regards to area, season and harbour. However, random selection of vessels does not entirely eliminate systematic bias. If observers cannot be deployed on selected vessels by a representative method, or if some of the selected vessels change fishing behaviour when observers are participating, the sample is biased. Additionally, bias resulting from logistical problems and lack of compliance is particularly difficult to quantify and control, and is not likely to be reduced by increasing sample sizes (Storr-Paulsen, 2011).

4.1.1. Merging international discard data

The European Data Collection Regulation and the Data Collection Framework (DCR EC no. 1639/2001 and DCF EC no. 199/2008) do not outline the methodology for data collection. Consequently, differences can exist in each Member State's national data collection programme. This limits the methods that can be used to analyse these data when merging national discard data. Therefore, **paper III** focuses on the numbers discarded and not the whole catch. The analysis in **paper III** is based on data from 2003-2010 and comprises 11370 hauls in 1189 trips sampled by 7 Member States within the North Sea demersal fisheries.

4.2. Measuring discards

When assessing ways to reduce discards, there are two control mechanisms that can be used to achieve this (Hall, 1996). Either reduce the discards per unit effort (DPUE) or reduce the level of effort (Hall et al., 2000). Most of the research into ways to reduce discards has focused on DPUE as this results in the greatest benefit to the fisheries. Furthermore, reducing the level of effort in a fishery is frequently a costly solution (Hall et al., 2000).

4.2.1. Discard rates/ ratios

As defined earlier (section 2), discard rates are either the number or weight per unit effort (DPUE) while discard ratios refers to the proportion (percentage) of the total catch that is discarded. When analysing discard rates specific changes to improve the fishery can be assessed. For example, gear modifications, changes to management regulations etc. The factors that potentially drive the discarding process can also be examined. Discard ratios have been calculated for a variety of different species across numerous fleets and fisheries (i.e. Stratoudakis, 1997; Stratoudakis et al., 1999; Stratoudakis et al., 2001; Borges et al., 2001; Rochet et al., 2002; Borges et al., 2005a; Goncalves et al., 2008).

4.2.2. Total discards

When raising discard samples to population levels there are numerous auxiliary variables (i.e. total landings in weight, effort in hours fished or in numbers of fishing trips) which can be used, often giving significantly different results (Stratoudakis et al., 2001; Trenkel and Rochet, 2001; Borges et al., 2005b). This huge variability makes any attempt to estimate total discards and discard proportions futile (Rochet and Trenkel, 2005). The most widely used assumptions for estimating and raising discards is that they are proportional to catch and fishing effort (Rochet and Trenkel, 2005; Borges et al., 2005b). Unfortunately, there is some evidence that these auxiliary variables might not be proportional to discards (Stratoudakis et al., 1999; Tamsett et al., 1999; Trenkel and Rochet, 2001; Borges et al., 2005b). Furthermore, variation in the results is exacerbated due to the low observer coverage that is often found in most discard data collection programmes. Borges et al. (2005b) analysed a range of sampling units (e.g. haul, trip) and auxiliary variables commonly used in discard estimations and found effort in hours fished provided the highest discard estimates while total fishing trips was the best auxiliary variable to raise discard samples to population levels. Stratoudakis et al. (1999) found that total demersal and gadoid landings were the best auxiliary variables tested. Grouping trips with the same gear, fishing ground and targeted species, was found to reduce the variability between trips (Borges et al., 2005b). However, a single optimum variable cannot be determined or recommended for application to studies across areas and fleet segments (Borges et al., 2005b). Until more conclusive knowledge has been acquired, great care should be taken when interpreting discard estimates obtained using assumptions of proportionality to catch or fishing time.

Measuring fishing effort in number of tows rather than time spent fishing and using simple sampling theory, i.e., raising by sampling units rather than using any auxiliary variable, are the recommended methods in the present state of knowledge (Rochet and Trenkel, 2005).

There is considerable literature devoted to estimating total discards within specific fleets or fisheries (see Rochet and Trenkel, 2005 Table 2). While such studies help to highlight the extent of the problem, absolute volumes alone have little meaning for assessing the impact of fishing. It is the relative mortality rate that gauges the impact on sustainability (Alverson and Hughes, 1996; Zhou, 2008).

4.2.3. Discard Indicators

Indices of discards are used to illustrate changes in overall discard patterns over time (Fig. 2). A discard quantity index provides annual changes in total discard quantity, while discard rate and discard ratio indices demonstrate how discarding behaviour during fishing operations changes with time (Catchpole et al., 2011). Indicators are often too unspecific and observed changes in state indicators can be hardly attributed to a single cause (Daan, 2005). Subsequently, indicators need to be used with caution. For example, an increase in mean length of discards can be caused by an increase in selectivity, but also by a decrease in the abundance of small species as a result of low recruitment. Furthermore, knowledge of the consistency in the reporting of fishing effort and landings as well as in the sampling programme is important in identifying genuine trends (Catchpole et al., 2011).
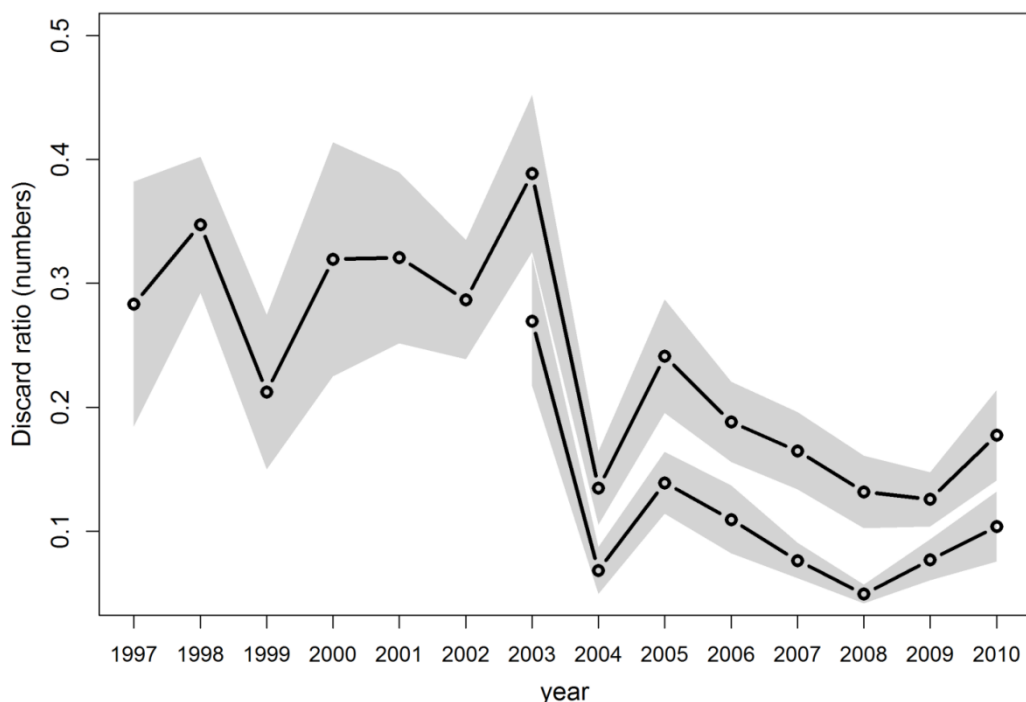

Fig. 2. Discard ratio, in numbers, of cod per year. Mean (black line) and standard deviation (shaded area). The lower line represents the discard ratio assuming the MLS had remained at 35 cm for years 2003 – 2010 (**Paper II**).

4.3. Modelling Discards

The application of additive models to fisheries data has been described in great detail by Swartzman et al. (1992). The use of modelling approaches to discard data provides the possibility to answer a range of different questions. Borges et al. (2006) suggest that additive models can be used to:

- evaluate the impact of changes or new management measures on discards
- investigate the effectiveness of (and changes in) technical conservation measures (such as mesh size increases, changes in MLS) between gears and areas (**Paper II**)
- detect the effect of landing restrictions on discards, for instance the impact of quota reductions and changes between quota systems (e.g. monthly to trip-based) (**Paper I**)
- allow for the inclusion of other explanatory variables (such as area, gear, depth, species catch composition and abundance, or season) (**Paper I; Paper II; Paper III; Paper IV**)
- compare/calibrate other discard data estimation methods, for example based on selectivity data (Casey, 1996) or in population simulation studies (van Keeken et al., 2003)
- detecting trends in species distributions (**Paper III**) and improving abundance estimates by including such trends

Additive models (Generalized Additive Models (GAMs), Generalized Additive Mixed Models (GAMMs); Hastie and Tibshirani, 1990; Zuur, 2009) are nonparametric or semi-parametric generalizations of multiple linear regressions. Both methods allow for non-linear relationships between the response variable and multiple explanatory variables. The difference between the two is that GAMMs can model the smooth terms as random effects. Smooths are tools for summarizing the trend of a response variable as a function of one or more predictor variables. Scatterplot smooths (Fig. 3) in additive models replace least square fits in regressions. The non-parameteric nature of smooths means that it does not assume a rigid form for the dependence of the response variable on the predictor variable. Whereas linear models assume that the response is linear in each predictor, additive models assume only that each predictor affects the response in a smooth way.

Generalized Additive Models (GAM) can be written as

$$Y = \alpha + \sum_{j=1}^{n} f_j\left(X_j\right) + \varepsilon$$

where the usual linear function of a covariate, $\beta_j X_j$ is replaced with f_j, an unspecified smooth function. The amount of smoothing is determined by the number of degrees of freedom applied to the smoothing spline function of each covariate. More information on non-parametric GAM regression models can be found in Hastie

and Tibshirani (1990) and Zuur et al. (2009). Their practicality is evident when the relationship between the variables is expected to be of a complex form, not easily fitted by standard linear or non-linear models, there is no a priori reason for using a particular model, and when we would like the data to suggest the appropriate functional form. Despite the usefulness of additive models, the outputs can sometimes be difficult to interpret.

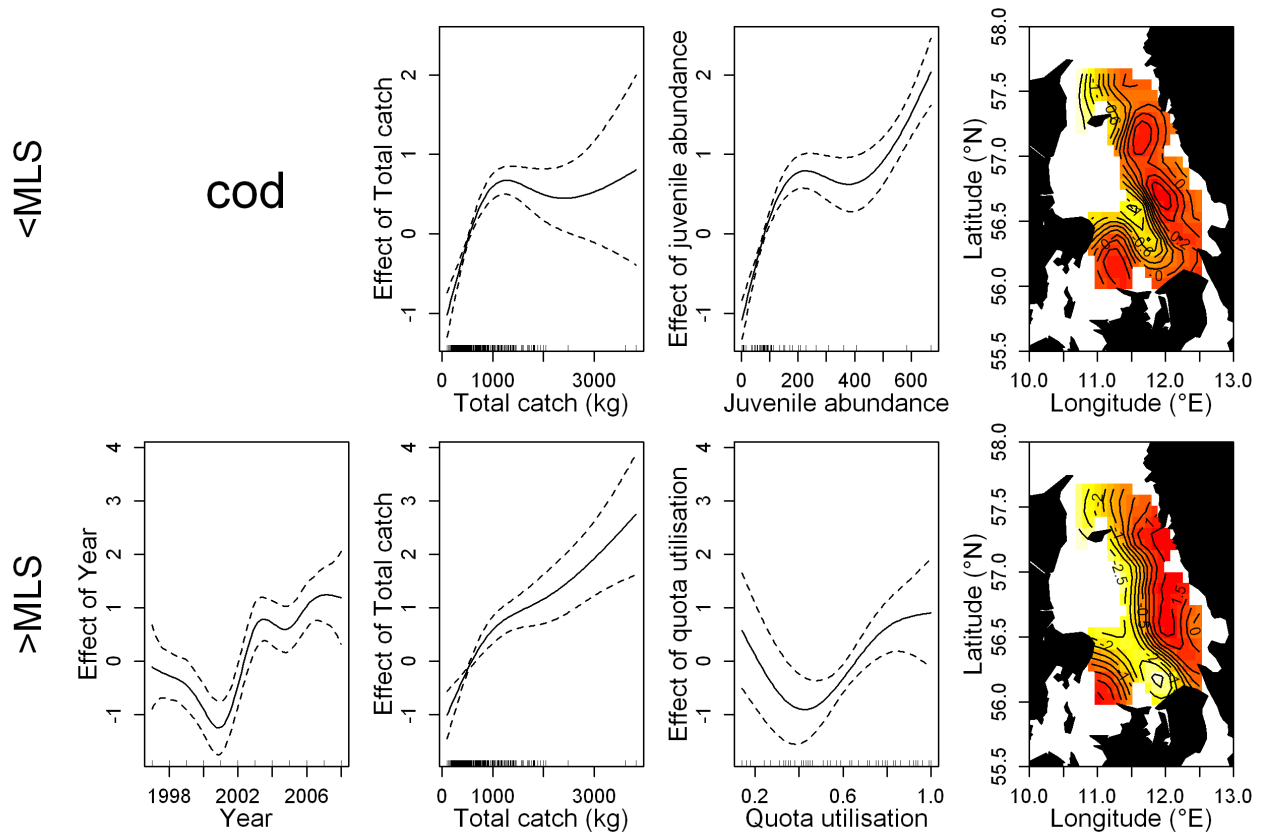Fig. 3. Effect of the significant smoothing functions (solid line) on the discard rate of cod in the Kattegat demersal trawl fishery. Cod <MLS (top row) and ≥MLS (bottom row). Dotted lines represent the 95% confidence limits. Vertical bars along the x-axis indicate observational values. The surface and contour lines describe the effect of 2-d smoothing function on the geographical coordinates (**Paper I**).

5. Causes of discarding

Knowledge about the various factors influential to discarding is essential when designing management strategies to maximise landings and minimise discards (**Paper I; Paper II; Paper III; Paper IV**; Murawski, 1996). In addition, it is a key element in the progress towards a theory of discarding (Rochet and Trenkel, 2005). The process of discarding is a consequence of a combination of different complex factors (Jennings and Kaiser, 1998). The relative importance of each factor is often highly species and length specific (i.e. discards under MLS are affected by a suit of factors that are different from those that are influential to discards over MLS). The same is valid for different fisheries, gears, and areas. Therefore, extrapolating results from one study to the next is often not feasible. The factors influential to discarding include; (i) biological and environmental, (ii) economic, (iii) social, (iv) legislative, and (v) technical factors. Disentangling the influential factors can often be difficult and can vary for numerous reasons (Machias et.al., 2004). However, the root cause is the lack of selectivity of fishing gears or operations, both within and among species, notably in trawl fisheries (Murawski, 1996; Mesnil, 1996).

5.1. Technical (The catching process)

The fishing method plays an important role in determining what is discarded. Subsequently, regulations of gear design and mesh size are commonly used to limit discards (Kulka, 1998) and have been examined in several studies (Rochet and Trenkel, 2005). However, various other technical parameters are also known to be influential (Morizur et al., 1996; Murawski, 1996; Perkins and Edwards, 1996; Blasdale and Newton, 1998; de Silva and Condrey, 1998). As each fishery generally has specific technical regulations, discards need to be studied on a fishery basis and cannot be extrapolated from one fleet to another (Rochet et al., 2002).

5.1.1. Gear selectivity

Improving gear selectivity is one of the most common ways to address the issue of discarding (Broadhurst et al., 2006; Madsen, 2007). Improvements generally involve either exploiting the various behavioural and morphological differences between species or sorting the catch mechanically based on size. The main issue is the need to decouple catches of different species and/or sizes. E.g. in the Kattegat the main issue is the need to decouple cod catches from those of Norway lobster (*Nephrops norvegicus*), plaice (*Pleuronectes platessa*), and sole (*Solea solea*) in a mixed demersal trawl fishery (**Paper I**; Madsen and Valentinsson, 2010). On the other hand, selective improvements in the Baltic Sea demersal cod trawl fishery were needed to reduce catches of undersized cod (**Paper II**; Fig. 4).
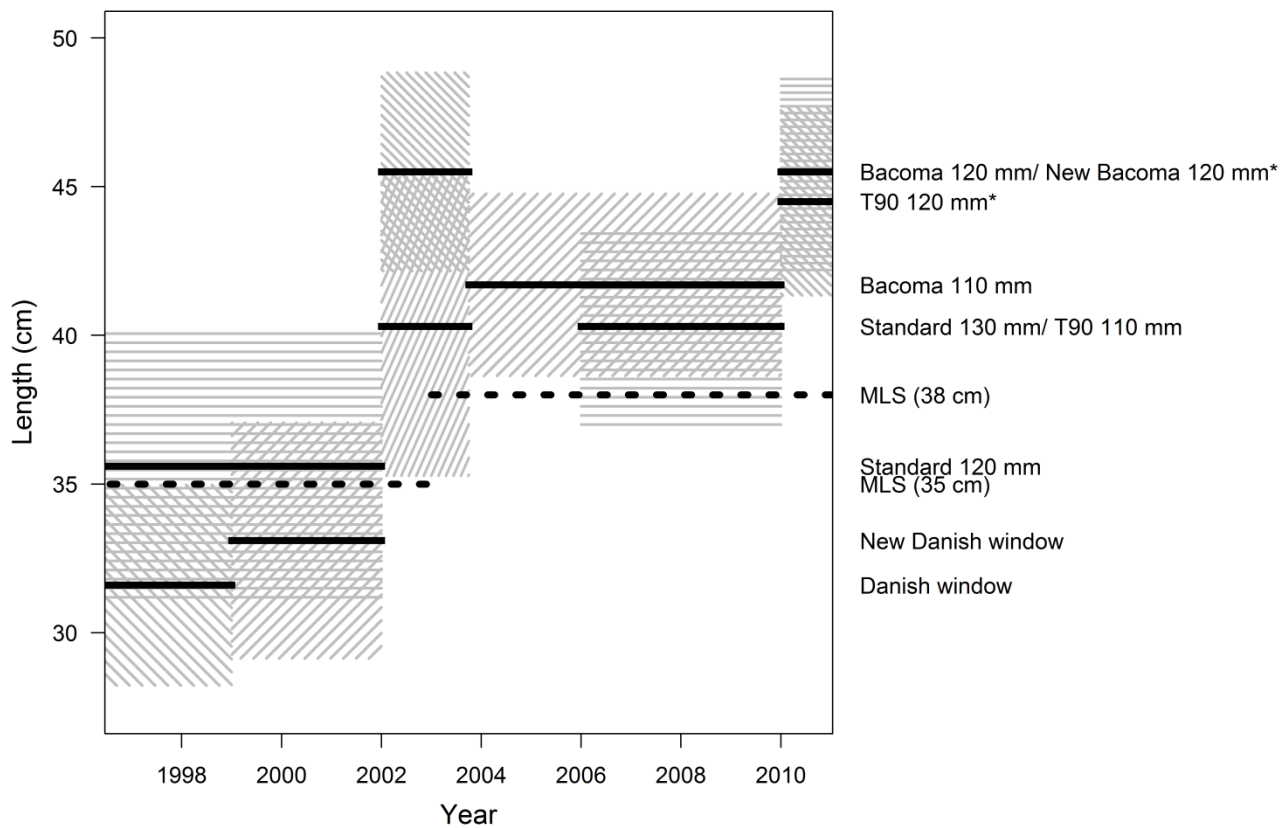
Fig. 4. Changes in codend selectivity (L50 and selection range, SR) for the regulated towed gears and changes in minimum landings size (MLS). *minimum mesh opening of 120 mm from 1 January 2010 in subdivisions 22–24 and from 1 March in subdivisions 25–32. The selectivity parameters for the T90 120 mm codend are taken from Wienbeck et al. (2011). All other selectivity values are taken from Madsen (2007). The selectivity estimates used for the Bacoma 120 mm and the New Bacoma 120 mm are unchanged, since the only difference is the window length, which is not expected to make any difference in relation to the used selectivity estimates obtained with relatively low catch rates (**Paper II**).

There are many additional factors that affect gear selectivity and consequently discards. These include mesh size (Madsen, 2007; Madsen and Valentinsson, 2010), twine thickness, single twine vs. double twine (Lowry and Robertson, 1995; Tokac et al., 2004; Herrmann and O'Neill, 2006; Sala et al., 2007), the use of attachments such as round straps (Herrmann et al., 2006), haul back (Madsen et al., 2008a; Madsen et al., 2008b; Grimaldo et al., 2009), length of the selvedge ropes or codend circumference (Herrmann et al., 2009), mesh shape (Herrmann et al., 2007; Madsen, 2007; Herrmann et al., 2009), location of the selective window (Graham and Kynoch, 2001; Graham et al., 2003), distance between doors, wing length, headline height (Madsen and Valentinsson, 2010), and raising the footrope of the trawl off the bottom (Krag et al., 2010).

In addition, most trawl selectivity experiments are conducted with newly constructed fishing gears. The materials used are known to change over time and the selective devices might be rigged and fished in other ways when used by commercial fishers, than during the original scientific experiments (Tschernij and Holst, 1999;

Madsen, 2007; Suuronen et al., 2007). Consequently, when introduced into a commercial setting the gear may not perform as expected. Furthermore, because a mixture of species is usually caught, and fishing practices are not standardized, the relations between mesh size, discarding and total catch are complex and confounded (Murawski, 1996). Therefore, assessing their effectiveness under commercial settings is necessary (**Paper II**).

5.1.2. Catch weight

Catch weight can influence the performance and selectivity of trawls (Wileman et al., 1996; Madsen, 2007; Madsen and Valentinsson, 2010). Selectivity of trawls has been shown to increase and discards decrease as catch weight increases (Rochet and Trenkel, 2005; Herrmann et al., 2006; **Paper I**). This is because the majority of selection occurs directly in front of the catch. As the catch accumulates the meshes begin to open up, making it easier for smaller individuals to escape. In other cases, the discarded fraction was shown to increase with catch at both the haul and trip level (Evans et al., 1994; Machias et al., 2001; Rochet and Trenkel, 2005). However, this was believed to be related to technical constraints (Rochet and Trenkel, 2005).

5.1.3. Haul duration and speed

Haul duration has been found to influence discards; however, no clear trend exists. Several studies have shown that longer hauls result in lower discard rates (**Paper II**; Murawski, 1996; Machias et al., 2001). Conversely, **Paper II** and Rochet and Trenkel (2005) found discarding increased nonlinearly and may be due to fish being damaged by long hauls or clogged nets preventing escapement (Rochet and Trenkel, 2005). The speed of the haul has also been found to influence discards (Hall et al., 2000; Broadhurst et al., 2006). Since a fishes' swimming speed and endurance is dependent on its body length (Bainbridge, 1958; Beamish, 1978), the speed of the haul can affect the sizes and quantities of fish retained (Broadhurst et al., 2006). Apart from restricting the spatial and temporal distribution of towed gears, regulating haul duration and speed are the most simple operational changes that might improve species and size selection and thus reduce discard (Broadhurst et al., 2006). It has been noted that haul duration may act to integrate patchy distributions of more-or-less segregated resources into what seem to be a mixture of species (Murawski, 1991). Thus, the implication is that shorter tow times may result in less diverse catches, and perhaps a higher proportion of target species (Murawski, 1996).

5.1.4. Trip duration

Trip duration has been found to affect discard rates, however, only for longer trips. Several circumstances could lead to this scenario: conservation problems on vessels without freezing facilities, or fish species with mainly a market for fresh fish (Rochet and Trenkel, 2005). Borges et al. (2006) found that haddock of all sizes caught

early on a long trip would be expected to have a greater discard rate than those caught later in the trip. Rochet et al. (2002) also found the duration of the trip to have a positive effect on discards of the most perishable species (cuckoo ray, hake and red gurnard), while a negative effect for megrim and *Nephrops*.

5.1.5. Vessel

Vessels can have large differences in discard rates. This is often a result of a combination of factors including; skipper effect, sorting behaviour of the fishers, vessel type, vessel power, storage capacity and hauling procedure (Evans et al., 1994; Tschernij and Holst, 1999; Machias et al., 2001; Rochet et al., 2002; Machias et al., 2004; Rochet and Trenkel, 2005; Poos et al., 2012). Vessel as well as crew discard behaviour are probably the key-factors explaining differences among trips (Rochet et al., 2002). The type of vessel can also influence the selectivity and subsequently the discards. This is largely due to the time the trawl spends floating beside the vessel before it is hauled on-board, allowing the meshes to become more open and subsequently providing smaller fish the chance to escape. It has been shown that the hauling procedure of, for example, a stern trawler and a side trawler have different selectivity curves (Tschernij and Holst, 1999). The power of a vessel can be considered a proxy for the size of the net which a vessel is able to tow. Therefore, a bigger vessel can tow a bigger net and catch/ discard more fish. **Papers I and III** examined this assumption and found it to be influential in only one of ten cases. The vessel factor can also be largely influenced by the skipper/crew effect and their sorting procedures. The captains' vigilance over the crew sorting the catch has also been mentioned as possibly influencing the discarding procedure in the Mediterranean (Machias et al., 2004). At the end of the fishing period all vessel owners were not on board. In this case, the owners of the vessels engage someone from the crew as a captain, which can result in rather loose sorting (Machias et al., 2004). The storage capacity of vessels has also been found to influence discards in several cases (e.g., Evans et al., 1994; Vestergaard, 1996; Machias et al., 2001; Rochet and Trenkel, 2005). The corresponding assumption is that the proportion of animals discarded increases as storage capacity becomes limiting (Rochet and Trenkel, 2005). In **Papers I, II, and III** the vessel effect was treated as a random variable and found to have a significant effect on discards in a majority of cases. Despite this, Borges et al. (2001) found the vessel characteristics not to be influential.

5.2.   Biological and Environmental

The impact of biological and environmental variables on discards is implicitly assumed in many studies that stratify their sampling design according to area, season, or both and generally proves to be true when examined (Rochet and Trenkel, 2005). Variations in discards are also highly probable according to recruitment, depth, bottom type and other abiotic and biotic factors (Murawski, 1996; Allain et al., 2003).

5.2.1. Recruitment

For commercial species, a closely related assumption is that recruits comprise a large portion of the discards; hence, year-class strength should be reflected in the amounts of fish under MLS discarded. The effects of year-class strength on discard rate has been well documented and generally found to be true (**Paper I; Paper II**; Reeves, 1990; Weber, 1995; Murawski, 1996; Rochet et al., 2002; Rochet and Trenkel, 2005; Borges et al., 2006). However, there are a couple of cases where this assumption has not held, e.g. for North Sea cod (*Gadus morhua*) discards by shrimp trawlers (Revill, 1997) and megrim, plaice and whiting discards in Irish demersal fisheries (Borges et al., 2006). The lack of an effect in the Irish demersal fisheries is believed to be because the recruitment estimates used as inputs in the study were drawn from stock assessments that generally consider landings only (rather than total catches) (Borges et al., 2006). An alternative approach to using recruitment estimates from such assessments would be to use survey estimates of recruitment directly (Borges et al., 2006). As what were used in **papers I and II**. Time series of both discards and recruitment estimates are needed to explore the theory (Rochet et al., 2002). If fluctuations in recruitment are unaccounted for when assessing the factors influential to discards, improvements made to selectivity may appear non-significant.

5.2.2. Depth

Depth-related variations in discard rates and quantities are linked to differences in the species compositions of the fish communities and in the length–frequency distributions of some species. Species replace each other according to their bathymetric and geographical preferences (Allain et al., 2003). When examined, depth was found to be a major factor in determining discards for different areas and species (**Paper I, Paper III**; Stratoudakis et al., 1998; Blasdale and Newton, 1998; Kennelly, 1999; Moranta et al., 2000; Machias et al., 2001; D'Onghia et al., 2001; Allain et al., 2003; D'Onghia et al., 2003; Sánchez et al., 2004; Machias et.al., 2004; Rochet and Trenkel, 2005; Poos et al., 2012).

5.2.3. Spatial and temporal variability

The spatial and temporal heterogeneity in species distributions (Beaugrand et al., 2003) results in discarding being highly variable in space and time (Fig. 5; **Paper I; Paper II; Paper III; Paper IV**; Andrew and Pepperell, 1992; Alverson et al., 1994; Kennelly, 1995; Liggins and Kennelly, 1996; Liggins et al., 1996; Kennelly, 1999; Machias et al., 2001; Murawski, 1996; Stobutzki et al., 2001; Bergmann et al., 2002; Catchpole et al., 2005b; Rochet and Trenkel, 2005; Tsagarakis et al., 2008; Poos et al., 2012). Furthermore, temporal variability can take place on yearly, seasonal and diurnal time scales and can differ for different age classes. Younger individuals will not necessarily have the same spatiotemporal distribution as adults. Seasonal differences in discard rates can occur as a result of differences in market prices, quota restrictions or recruitment (Helser et al., 2002; Machias et

al., 2004; Viana et al., 2011; Fernandes et al., 2011). Machias et al. (2004) observed discards to be lower in winter because market prices increased due to a decrease in catches as a result of bad weather. Several studies have found discards to be higher during recruitment periods (Stergiou et al., 1997; Machias et al., 2004; Viana et al., 2011). **Papers I and Paper III** found discards to be higher later in the year as a result of quotas becoming exhausted. Similar trends were also found by Helser et al. (2002) and Fernandes et al. (2011). Knowledge about the spatiotemporal nature of discards is imperative to researchers and regulators (Dunn et al., 2011) but is often lacking (Viana et al., 2011). Information on the spatiotemporal distribution of discards can be used to limit directed fishing to times and places where resources are segregated, subsequently reducing the quantity of unintended catch (Murawski, 1996). If areas of persistently high fishing efficiency and selectivity are to remain open to fisheries, researchers and regulators first need to understand the spatiotemporal nature of discards within their systems (Dunn et al., 2011).
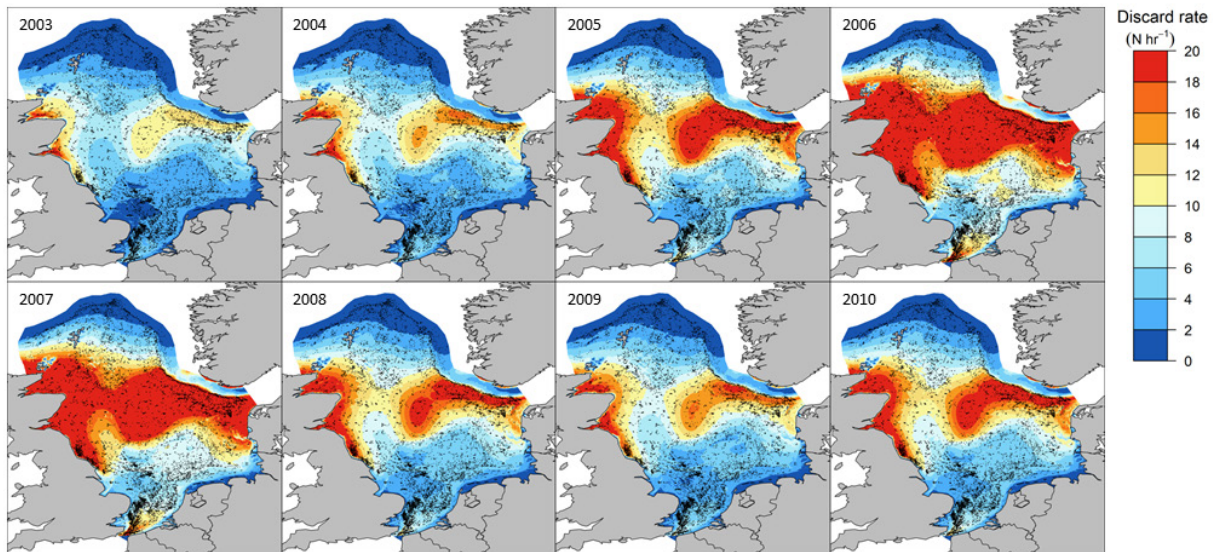
Fig. 5. Spatiotemporal distribution of young cod discards throughout the North Sea 2003 – 2010. Predicted from Generalised Additive Mixed Model (**Paper III**).

5.3. Legislative

The management framework has a strong influence on discard rates (Crean and Symes, 1994). In particular, output controls such as those that limit landings (quotas) and/or catch compositions, or technical measures such as minimum landing size will increase the incentives to discard (Pascoe, 1997; Stockhausen et al., 2012). Fisheries managed by such regulations are often characterised by high discard rates (Graham et al., 2007). A fundamental paradox when considering the regulatory system as a means for reducing discards is that it is the system itself that can often be responsible for generating discards in the first place. With respect to target species, for example, there are many cases where regulations enacted to try and ensure that they are not over-

exploited lead to discarding of the very species they are trying to protect. Thus, the mixture of incentives and disincentives that are put in place with particular legislation must be carefully evaluated and may not be easily foreseen (Hall and Mainprize, 2005).

5.3.1. TAC/ quotas

Within the European Union, Total Allowable Catches (TAC) are defined for most commercial species. It should be noted that in practice the TACs are actually Total Allowable Landings (TALs) since discarding is legal in most EU waters. Quotas, whether common pool quotas, individual transferable quotas (ITQs), or trip quotas all instil discarding (Gillis et al., 1995; Graham et al., 2007; Gray et al., 2011). Common pool quotas, such as what were active in Danish fisheries until 2007, often create a "race to fish" which can lead to wasted target quotas, high discard costs, and shortened seasons, all of which reduce rents and may lead to losses in the product market from reduced product quality and skewed product mixes (Abbott and Wilen, 2009). TACs and quotas are, in general, set to achieve a specific mortality for a single stock, independent of the status of other stocks. In multispecies fisheries, e.g. the North Sea demersal fisheries, this is often not the case. When the TAC/quota for one species is exhausted but opportunities remain for others, fishers often continue fishing for other species and discard catches of valuable species for which they have no quota (Brown et al., 1979; Graham et al., 2007; Poos et al., 2010; Kempf, 2010; Viana et al., 2011; Ulrich et al., 2011). Such is the case for North Sea cod (**Paper III**; Ulrich et al., 2011). Furthermore, increasingly stringent restrictions on landings alter behaviour towards fishing practices that seek to increase the value of a limited weight of catch, i.e. high-grading (Stratoudakis et al., 1998; Borges et al., 2006).

In 2007, Danish fisheries changed to an ITQ system, with the intention to improve the profitability in the demersal fisheries and to obtain a more suitable exploration of the stocks, with particular focus on reducing discards (Andersen et al., 2010). However, incentives to high-grade in ITQ fisheries are greater compared to open assess fisheries if limits are imposed on landings and not on catches (Vestergaard, 1996; Squires et al., 1998; Abbott and Wilen, 2009; Branch, 2009). This is because fishers will attempt to maximise their quota value. High-grading in ITQ systems occurs due to relatively low costs of discarding, a large price differential between classes of fish, and low costs of catching fish to replace those that were discarded (Kingsley, 2002). Despite the potential increase in high-grading under an ITQ system with landing limits, there are also incentives to reduce discards. Since ITQs provide an improved resource stewardship, through the increased security of harvesting rights, fishers are more willing to fish selectively, share information about which areas to avoid, increase self-enforcement, and lease or buy quota to reduce mismatches between quota and catch mixtures (Squires et al., 1998; Branch, 2009).

5.3.2. MLS

"A brilliant suggestion that the capture of undersized fish should be prohibited need not detain us long, since it is obviously impossible to avoid catching some undersized fish if one fishes at all, and what benefit could be expected from a legal prohibition of this sort I am at a loss to conjecture, since the law could not possibly be enforced as long as a fisherman was allowed to go to sea. There are, of course, methods by which the capture of a very large proportion of undersized fish can be prevented, but prohibition of capture, per se, is not one of these" (Holt, 1895).

Minimum landing sizes (MLS) are applied in many fisheries to protect smaller fish. MLSs are generally thought to be the key to the sorting process: fish smaller than the MLS should be discarded, whereas those larger than the MLS will be retained (Rochet and Trenkel, 2005). However, the effect of MLS regulations on discards is more noticeable for species of higher commercial value (Fig. 6). In the case of species with lower commercial value, animals much larger than the MLS have been found to be discarded (Evans et al., 1994; Rochet et al., 2002; Borges et al., 2005a; Rochet and Trenkel, 2005), suggesting that for lower valued species, the MLS regulation is not effective and that other mechanisms (e.g., market incentives) determine sorting behaviours. Alternatively, MLSs in some fisheries are not complied with and animals much smaller than the MLS are retained (Machias et al., 2004; Rochet and Trenkel, 2005).

Increasing the MLS should result in an increase in discards. Stratoudakis et al. (1998) reported an immediate increase in discarding with an increase in MLS for haddock and whiting. This was also the case for cod discards in the eastern Baltic Sea (**Paper II**). However, changes in MLS in some fisheries does not seem to cause a change in discarding practices, since discarding occurs at lengths higher than the established MLS (Borges et al., 2005a).

The mismatch between gear selectivity and MLS is a significant contributor to discards, especially in mixed fisheries (**Paper I and IV**; Graham et al., 2007; Frandsen et al., 2009; Madsen and Valentinsson, 2010). In fisheries targeting several species with different morphological characteristics, adjusting the legal mesh size and MLSs might not always be possible as the optimal mesh size for one species may not be suitable for other species (Rochet et al., 2002).
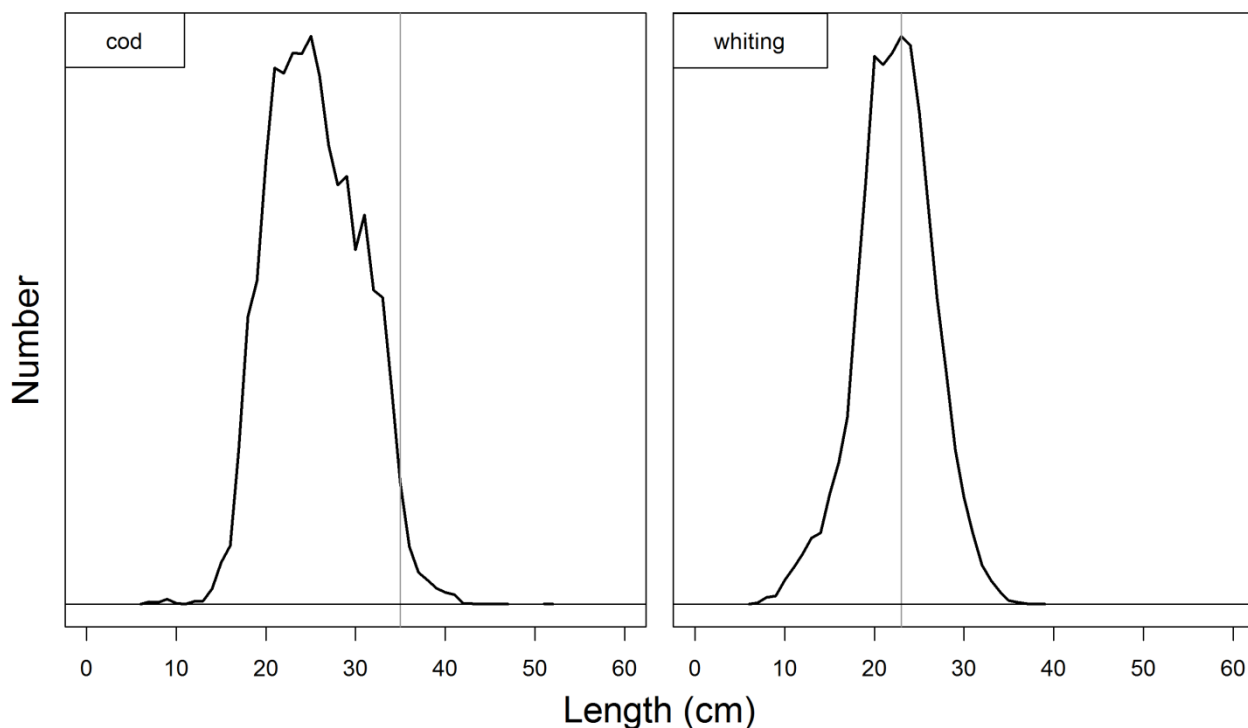
Fig 6. An example of the length compositions of the discarded portion of the catch for commercial (cod) and non-commercial (whiting) species in the Kattegat. Vertical grey lines represent the MLS for cod (35 cm) and whiting (23 cm).

5.3.3. Catch composition

The species and size distribution in the catch is largely determined by environmental variables (area, depth, season, recruitment, time of day) as well as the efficiency and selectivity of the gear. While knowledge of environmental variables may assist in reducing discards, species and size compositions cannot be precisely determined beforehand. This can lead to large catches of undesired species and/or undersized fish that are subsequently discarded (Goncalves et al., 2008). Furthermore, catch composition regulations may force fishers to discard excess catches of certain species (Graham et al., 2007).

In the EU, catch composition requirements are defined in Council Regulation (EC) No 850/98. The rules specify the minimum percentages of the target species that can be landed (as a proportion of the quota species) and are designed to limit catches of non-target quota species.

5.4.   Economical

‘‘Why is it that conservation is so rarely practiced by those who must extract a living from the land? It is said to boil down, in the last analysis, to economic obstacles’’ (Leopold, 1966).

Fishing is an economic activity where all marketable individuals (i.e. those over MLS) caught during a fishing operation do not have the same value. Hence, fishers aiming at increasing their revenue will discard the least valuable part of their catch (Rochet and Trenket, 2005). Economic influences are considered to be one of the main reasons for discarding and occur for a variety of reasons, including;

i)   high-grading (Machias et al., 2004; Stanley et al., 2011; Stockhausen et al., 2012),

ii)   species are of low market value (Borges et al., 2001; Catchpole et al., 2005b; Goncalves et al., 2008),

iii) the species is non-marketable, or

iv)   processor/market limits on the acceptable species and sizes of fish (Helser et al., 2002).

Furthermore, market price can fluctuate throughout the year and even differ considerably between ports (Fig. 7) and can potentially be correlated with the amount landed. Usually the total catch of nonmarketable species is discarded and the whole catch of high-value species is retained, whereas low-value species are partially discarded (Perez et al., 1995; Rochet et al., 2002; Rochet and Trenket, 2005). Economic influences are paramount, and efforts to reduce discarding that fail to take these influences into account are unlikely to be successful (**Paper II**; Graham et al., 2007).
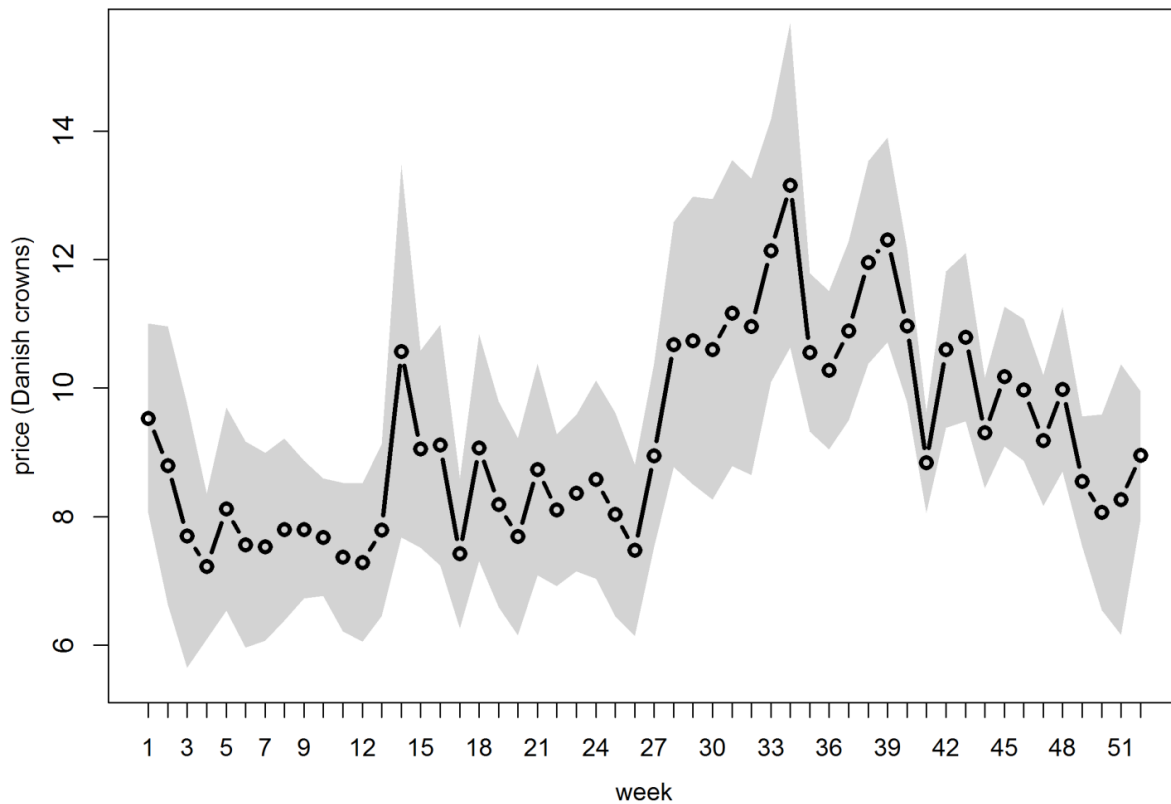

Fig 7. Variations in market price for cod in the Baltic Sea throughout the year. Grey band represents the variation (standard deviation) in price across ports.

5.5. Social

If there is no understanding and agreement from fishers with regards to the regulations enforced, compliance may be low. Also, if regulations result in large economic losses, as what was the case in the Baltic demersal cod fishery when the Bacoma window was first introduced, fishers will attempt to circumvent the regulation (**Paper II**). Fishers are aware that regulatory discarding of marketable dead fish serves no conservation purpose. This undermines their faith in the management system and can lead to non-compliance and illegal landings (Graham et al., 2007). Additionally, there is the challenge to alter the attitudes and values of fishers and ensure that economic incentives are aligned with those for conserving marine ecosystems and communities. Without such an alignment and shift in values to drive changes in fishers' behaviour, the effectiveness of the technical and legislative systems will be diminished (Hall and Mainprize, 2005).

5.6. Others

There may be additional factors that are often difficult to account for post data collection. For example, it is difficult to determine which proportions of animals of marketable size were discarded because they were damaged, or due to low market prices, or for other unknown reasons (Clucas, 1997; Rochet et al., 2002).

6. Methods for reducing discards

Methods for reducing discards are often fishery, fleet or area specific. What works for one fishery or area may not necessarily work for another. Subsequently, the range of methods available for reducing discards is numerous. The following chapter provides an overview of methods available for reducing discards. The first two sections are directed at methods that have been used to reduce discard rates and total discards within demersal trawl fisheries while the third section is devoted to additional strategies that are being discussed within EU to develop discard free fisheries.

6.1.   Discard rates

Most methods for reducing discards focus on discard rates and/or ratios. Reducing discards requires striking a balance between sustaining economic returns and minimising discards. Improvements to gear selectivity, together with spatial and temporal closures are the most common ways of achieving this. Improvements to gear selectivity are numerous and very often fishery specific. This is because of differences in target species, species compositions and the objectives of the selectivity improvements. Gear selectivity improvements are often required to either remove a species from the catch, such as cod in *Nephrops* trawls, or to remove a size class of a species, such as juvenile cod in the Baltic Sea cod trawl fishery (**Paper II**).

The composition of trawl catches is determined by the distribution of fish populations in relation to locations where the gear is deployed, and by the physical characteristics of the gear itself (Murawski et al., 1983). Therefore, the spatial aspect of discarding is extremely important in reducing discards (**Paper I; Paper II; Paper III; Paper IV**). Spatial and temporal closures are implemented for a variety of different reasons. Protection of spawning stocks or nursery grounds and controlling discard mortality are the main reasons for spatial and temporal fishery closures. Protection of a species or size class through the use of fishing closures has taken on a new form within the EU recently, with the emergence of real-time closures (RTC). RTCs are enforced when the catch rate of a species or size class exceeds a certain threshold. Such closures have been successfully implemented in Norway, Iceland, Faroe Islands, United States of America and Scotland, and there emergence with EU fisheries is increasing.

6.1.1. Gear selectivity (Trawls)

Excluding temporal and spatial closures, the most common way of addressing the issue of bycatch in towed gears has been to improve gear selectivity (Davis, 2002; Cook, 2003; Valdemarsen and Suuronen, 2003; Broadhurst et al., 2006). Substantial effort has been devoted to investigating the effectiveness of various modifications to trawls as a means for improving species and size selection and, therefore, reducing unwanted

bycatch (see Broadhurst et al., 2000; Madsen, 2007; Madsen and Valentinsson, 2010 for reviews). The effectiveness of gear selectivity improvements largely depends on differences in the behaviour and size of the target species compared to the organisms that will be discarded (Catchpole., 2005b; Ferro et al., 2007; Madsen, 2007; Krag et al., 2009a; Krag et al., 2009b). Furthermore, the type of modification appropriate for any given fishery will also depend on the nature of the trawling grounds, and the fishing practices and vessels employed in the fishery (Kennelly, 1995). Additionally, the approaches to improving selectivity are different for single-species fisheries compared to multispecies fisheries. Consequently, no one technical solution works universally.

The advantages of improving selectivity, apart from reduced discards, include: cleaner catches which in turn reduces sort time (Bjordal, 1999; Goncalves et al., 2008), improved catch quality (Bjordal, 1999), increased storage capacity devoted to commercial species (Goncalves et al., 2008), and the possibility to operate in otherwise closed areas. There are also environmental and ecological advantages associated with improving selectivity, such as reduced impacts at the population, species, community and even ecosystem levels (Goncalves et al., 2008). Despite the many positives associated with improving selectivity, many selective designs also reduce a portion of the marketable catch. The short-term economic losses often associated with improving selectivity are considered to be the most common reason that discourages their uptake by fishers (Catchpole et al., 2005a; Hall and Mainprize, 2005; Suuronen and Sardà, 2007). Additional factors that reduce the uptake of new selective designs include: the economic costs associated with new technologies (Hall et al., 2000; Catchpole et al., 2005a; Suuronen and Sardà, 2007), and the perceived increase in risk when operating more complex gear (Catchpole et al., 2005a; Suuronen and Sardà, 2007; Madsen and Valentinsson, 2010). Furthermore, when losses of marketable catch occur, effort may increase to compensate for the loss, thereby negating the benefits of bycatch reduction (Hall and Mainprize, 2005). In the following paragraphs several approaches that are used to improve selectivity and reduce discards in Danish waters are discussed.

Minimum Mesh Size (MMS) regulations have traditionally been the main legal measure to prevent catching juveniles and small individuals. They are one of the simplest and most commonly used measures available to improve the size selection in codends. In general, the larger the mesh size the larger the individuals are that escape and the lower are the discard rates and ratios (**Paper IV**; Glass, 2000; Krag et al., 2008). However, increasing the MMS also leads to a reduction in the catch of marketable sized fish and subsequently economic losses. Therefore, substantial efforts have been devoted to developing gears that minimise the loss of marketable individuals while increasing the escape of unwanted individuals. One simple method to emerge is to change the mesh shape in the codend.

Square-mesh and T90 codends are designed to reduce the capture of small roundfish and other animals by providing a greater number of open meshes along the entire codend through which the fish can escape (Glass, 2000; Frandsen et al., 2010a). The optimal mesh configuration for selection of a species is determined by its cross sectional shape (Herrmann et al., 2009). Therefore, square-mesh codends have good selective properties

for roundfish species such as cod, haddock and whiting but not for some flatfish species (**Paper IV**; Glass, 2000; Madsen et al., 2006; He, 2007; Frandsen et al., 2010b).

Selective grids have been developed and successfully used in Norway for more than 40 years (Karlsen, 1976). They are designed to mechanically sort the catch according to size, excluding those individuals that are larger than the openings in the separating panel (Broadburst, 2000). The effectiveness of grids in excluding large quantities of unwanted bycatch, while maintaining catches of target species, has led to voluntary and enforced application within fisheries around the world (e.g. Norway, Denmark, Sweden, Canada, and Australia) (Broadhurst et al., 2000; Madsen and Hansen, 2001; Valentinsson and Ulmestrand, 2008). The use of grids can effectively make *Nephrops* fisheries into single-species fisheries (Catchpole et al., 2006). Therefore, in fisheries where bycatch makes up a considerable portion of a fisher's income, such as in the Danish trawl fisheries, selective grids may not be the most viable option. Square-mesh panels may possibly be an alternative when bycatch is of an economical interest. In the Kattegat, almost all Swedish *Nephrops* fishers utilise the grid system (Catchpole and Gray, 2010) while in the Danish fisheries the economic losses from its use were considered substantial and its uptake was consequently minimal. Therefore, the use of selective panels is preferred in Danish fisheries as they are able to reduce cod bycatch while maintaining bycatches of commercially important flatfish species, namely sole and plaice (Madsen et al., 2006; Madsen et al., 2010).

Square mesh panels (SMPs), otherwise known as escape windows, were first tested almost 100 years ago in the Kattegat and Baltic Sea (Ridderstad, 1915). They are commonly used in single species fisheries such as in the Baltic Sea (**Paper II**; Madsen, 2007) and in multispecies fisheries where the target species and the species wanting to be removed have relatively different morphological characteristics, such as reducing discards of roundfish in Norway lobster (*Nephrops norvegicus*) trawl fisheries (Briggs, 1992; Madsen et al., 1999; Revill et al., 2007; Krag et al., 2008; Frandsen et al., 2009; Madsen et al., 2010). However, their effectiveness is reduced in multispecies fisheries where the target and bycatch species are of similar morphological characteristics, like in the multispecies fisheries in the North Sea. Utilising behavioural differences between species in such cases has shown to be effective (Krag et al., 2010)

The above mentioned methods all relate to changes in the codend of trawls. This is because it is where most of the selection within the trawl takes place. Additional alterations to trawls that can reduce the catch of unwanted species or size classes include raising the footrope (Krag et al., 2010), lowering the headline height (Sangster and Breen, 1998), modifying the herding effect (Ryer, 2008; Winger et al., 2010), using separator panels (Engås et al., 1998; Rihan and McDonnell, 2003; Ferro et al., 2007), and removing the top part of the trawl (Revill et al., 2006; He et al., 2007). The use of deterrents, such as physical, acoustic and electronic modifications, could also reduce components of unwanted catch and discards (Broadhurst et al., 2006). Finally, it might be feasible to consider completely different fishing methods to catch the target species that have lower discards, such as longlines, gillnets, and pots (Broadhurst et al., 2006; Catchpole and Gray, 2010).

6.1.2. Spatial and temporal closures

Temporal and/or spatial closures are a common approach that has widespread acceptance for protecting species at certain stages of their life history, for example, protection of juvenile nursery areas or adult spawning grounds (Hall and Mainprize, 2005) and controlling discard mortality (Alverson et al., 1994; Machias et al., 2004). The main objectives of employing spatial and/or temporal strategies to reduce discards are to avoid areas of high juvenile abundance or to utilise the variations in the degree of co-occurrence between target and bycatch species (**Paper I; Paper II; Paper III; Paper IV**; Murawski, 1992). While closures are considered an effective means for stopping bycatch problems within a desired area or time, their effectiveness can potentially decrease because trawling effort may increase in areas and times outside particular closures, effectively negating some or all of the desired effects of the management strategy (Kennelly, 1995; Suuronen et al., 2010). Consequently, the use of closures is becoming less prescriptive and more incentive based (Graham et al., 2007). Closures can be used as an incentive to improve selectivity by providing better fishing opportunities to fishers who are using more selective fishing methods (Hall and Mainprize 2005; Catchpole et al. 2005b; Dunn et al. 2011). Such is the case in the Kattegat where trawling is allowed to continue within an area closed to protect spawning cod on the condition that only gears with minimum catches of cod are used (e.g. Swedish grid/ SELTRA panel).

Incentives, such as those mentioned above, can reduce discards and decrease the shift in effort to other areas which may have higher discard rates/ratios. In the Baltic Sea, the introduction and enlargement of a spatial closure caused substantial effort displacement towards areas dominated by smaller sized cod. This contributed to an increase in the capture and discarding of undersized cod (Suuronen et al., 2010). Hence, when designing closures, there are many additional factors that need to be considered before their implementation. These include economic considerations such as reduced profitability, increased additional costs through wear and tear of fishing gears and increased competition between different fleet segments, which may also increase the number of lost nets (ghost nets) (Suuronen et al., 2010). Furthermore, the use of spatially restricted closures may not affect all fishers equally due to their locations being closer to some fishers' home ports or preferred fishing grounds than for other fishers. This can lead to fishers feeling unfairly treated and potentially circumventing the regulation. For example, the enlargement of the Bornholm Basin closure in the Baltic Sea only displaced Swedish effort, which Swedish fishers considered an unfair management action (Suuronen et al., 2010). Because of this, Swedish fishers favoured a seasonal ban rather than spatially restricted closures. The seasonal ban protects spawning individuals without the risk of redirecting fishing effort towards potentially sensitive nursery areas of juvenile cod (Suuronen et al., 2010). Viana et al. (2011) also proposed the use of seasonal closures as a potential mitigation tool to reduce discards during peak periods (Viana et al., 2011).

Regulations also need to be applied to the whole fishery active within a specific area. If not, belief in the system will be lost. Such is the case in the Kattegat. In 2008 a closed area was introduced to protect cod. However, this closure only applies to Danish and Swedish fishers and not German fishers. Subsequently, fishing activity was still observed within the closed area. If the regulation is not uniform for the whole population it will be perceived

as being unfair and more than likely circumvented. Finally, the response of fishers to the imposition of closed areas, while poorly known, can be critically important to their effectiveness, as it is to any management objective (Suuronen et al., 2007; Suuronen et al., 2010).

The variability in the timing and location of large bycatches of juveniles of important species precludes the establishment of fixed seasonal or localised spatial closures (Kennelly, 1995). Subsequently, the use of more flexible closures is becoming common.

6.1.3. Real time closures

A more dynamic approach than closing areas seasonally or permanently is the use of real-time closures (RTC). RTCs are areas closed to fishing for a limited period, triggered by information gained by managers in "real time", often in cooperation with the industry, such as on-board sampling of catch compositions, Vessel Monitoring System (VMS) data, analysis of catch rates or skippers declarations (Bailey et al., 2010). RTCs can be used to protect areas of high abundance, areas where young fish and juveniles comprise a higher than average proportion of the catch, or areas where catch composition is likely to result in high levels of discards. They can also be used to improve quota uptake in multi-species fisheries (Bailey et al., 2010). The use of RTCs has been shown to influence fisher's behaviour and the uptake of more-selective fishing technologies (Graham et al., 2007). The use of more selective gears can also be used as an incentive to gain access to otherwise closed areas. RTCs can also be used to incentivise fishers by rewarding participation with additional days at sea or extra quota (Catchpole and Gray, 2010; Holmes et al., 2011). Additionally, compliance with RTCs may potentially increase under a catch quota management system where all individuals are counted against quotas. This is because fishers will not want to fill their quota with individuals for which they receive little economic gain. As the use of real-time closures becomes more common, better scientific knowledge regarding their implementation, size, shape and duration will become available and their effectiveness may increase further (Gilman et al., 2006). Analysis of the spatial and temporal movements of cod from tagging studies has provided for a better understanding of their short-term movements. Incorporation of this knowledge into the Scottish system led to real-time closures increasing fourfold in size (Holmes et al., 2011). However, the success of RTCs is highly dependent on the compliance by fishers (Kempf, 2010) and the fast and reliable dissemination of information to fishers (Holmes et al., 2011). It will be finally up to the fishers to report high catches of juveniles or unwanted bycatch species even if this may imply economic loss for them (Kempf, 2010). Furthermore, like spatial and temporal closures, RTCs displace fishing effort rather than reducing it. Finally, analysing the effectiveness of RTC is particularly difficult as they represent an "uncontrolled experiment" where it is not possible to compare their outcomes against a hypothetical situation where they have not been deployed (Bailey et al., 2010).

In September 2009, the EU and Norway agreed to implement a RTC scheme in the North Sea and Skagerrak, with the aim of protecting juvenile and undersized fish (cod, haddock, saithe and whiting (*Merlangius*

merlangius), and to reduce discards (Commission Regulation (EU) No 724/2010; Commission Implementing Regulation (EU) No 783/2011; Holmes et al., 2011). A closure is implemented if 15 % of the catch consists of juveniles of these four species. However, if the quantity of cod exceeds 75% of the total, the trigger level is set at 10%. Reopening occurs automatically after 21 days (Bailey et al., 2010).

6.1.4. Adjusting the MLS

Minimum landings sizes (MLS) are applied in many fisheries to protect smaller individuals. In principle, MLSs should be based upon the size at first maturity of each species, rather than as a function of the gear selectivity. Therefore, MLSs regularly promote discards since they are often difficult to harmonize with the selectivity of the fishing gear, particularly in multispecies fisheries (Kelleher, 2005; Suuronen and Sardà, 2007). In multispecies fisheries, species of different sizes and morphological characteristics are caught, resulting in a range of different MLSs. Increasing the MLS without increasing selectivity can result in an increase in discards (**Paper II**; Kelleher, 2005) Therefore, lowering the MLS is warranted if discards want to be instantaneously reduced. However, lowering the MLS would result in the increased retention of pre-spawning individuals, which is considered to violate the precautionary approach (Myers and Mertz, 1998). This is of little concern as the individuals are already dead. Their initial capture is what is of most concern. Hence, further increasing selectivity may be a better alternative. As most species are still caught at a relatively young age, improving selectivity further will give great long term benefits. However, this will more than likely result in short-term economic losses.

6.2. Total discards

6.2.1. Effort reduction

While the objective of reducing fishing effort is to reduce fishing mortality, reductions in fishing effort can also prevent the exhaustion of quotas, subsequently reducing discards of over quota fish. However, reducing the level of effort in a fishery is normally an expensive solution (Hall et al., 2000). The EU fishing industry has been subsidized by the European Fisheries Fund to the tune of 3.8 billion Euros over the period 2007 to 2013 (Kempf, 2010). Limitations on effort (e.g. days at sea) also encourage fishers to increase their catching efficiency (more engine power, larger nets) as well as increase their actual fishing effort (longer hauls, more hauls per day) in order to maximise landings. Restrictions in effort can also result in restructuring among fleet segments. Following the introduction of effort regulations (days at sea) in the North Sea, Skagerrak, and Eastern Channel in 2003, there was a substantial switch from the larger mesh (>100 mm) gear targeting primarily roundfish to the smaller mesh (70–99 mm) gear targeting Norway lobster (ICES, 2011). This was because vessels using the larger mesh gear were restricted to 9 days at sea per month, while the smaller mesh gear to 25 days (Horwood et

al., 2006). Discards may have increased as a result of the restructuring. Effort restrictions can also be used to incentivise fishers to use more selective gears. In the Kattegat, days at sea have been unlimited if the more selective option "Swedish grid" was used.

6.3.   A move towards discard free fisheries

As mentioned earlier, there are many ways to reduce discarding. Here we discuss the issue of a discard ban within EU and Danish fisheries and the complimentary management measures which can help achieve a sounder utilisation of fish resources.

6.3.1. Discard ban

A number of countries (e.g. Norway, Iceland, New Zealand, and Canada) already manage discards by banning the practice through legislation (Hall and Mainprize, 2005; Diamond and Beukers-Stewart, 2011). The European commission is planning to follow suit and has agreed on introducing a discard ban, entailing that no fish, be it above or below the MLS, can be thrown overboard. It is important to highlight that a discard ban only applies to species that have commercial value and are either undersized or for which a fisher does not possess quota (Hall and Mainprize, 2005). A discard ban will encourage fishers to develop technical modifications to enhance gear selectivity (Hall and Mainprize, 2005; Graham et al., 2007). This is because the retention of discards can induce additional costs for the industry, relating to sorting, storage and landing of unwanted and unexpected fish (Hall et al., 2000; Hall and Mainprize, 2005; EC, 2011). A ban on discards may also encourage fishers to improve their selectivity by avoiding periods, areas, or times of the day with high bycatches (Hall et al., 2000). Furthermore, a discard ban ensures that more accurate information on total catches is recorded (rather than landings), and subsequently more accurate total allowable catches being set (Hall and Mainprize, 2005; Graham et al., 2007; Kempf, 2010). A decrease in overall fishing pressure may result from such a ban if currently discarded size classes would be counted against the available species quota (Kempf, 2010). Despite the many positives associated with a discard ban, this type of programme will only be effective with extensive monitoring to ensure compliance, which may not be economically viable (Hall et al., 2000). An additional danger with discard bans is that, if not carefully set up, one might develop a new or expanded market for the discards and thereby establish incentives for their capture (Hall and Mainprize, 2005; Stockhausen et al., 2012). Nevertheless, new markets and industries may need to be established to utilise the additional fish that will be landed under a discard ban. Thus, successful implementation of a discard ban requires striking a delicate balance between the incentives to discard and incentives to retain unmarketable catches (Gezelius, 2008). Incentives to retain currently unwanted catches can generate incentives to pursue such catch intentionally. Removing incentives to pursue such catch can create incentives to discard and misreport.

The proposed European discard ban includes a change, whereby all catches, not just landings, will be deducted from a quota, and fishing operations will stop once the catch quotas are met. To maximize their revenue, fishers would avoid catching immature fish, which yield no return, and avoid an early end to their fishing season by adopting more selective fishing practices. A ban on discards will likely face economic, regulatory, and political hurdles. Under a discard ban several issues emerge including: i) How to minimise the capture of juveniles and large individuals for which there is no quota under a discard ban; ii) How to ensure discarding does not take place. The success of a discard ban will depend critically on complementary management measures addressing these.

6.3.2. Catch Quota Management

One method which has been proposed by the EC to solve the discarding issue is the introduction of a Catch Quota Management (CQM) system. The primary objectives of CQM are to ensure total catch mortality of a given stock is accounted for and create incentives to fish selectively and avoid juvenile catches (Kindt-Larsen et al., 2011). Under a CQM system all fish are counted against quota, regardless of size and marketability. Therefore, the introduction of CQM immediately disposes of discarding. Because catches of undersized or unmarketable fish will reduce a fisher's income, a CQM system presents fishers with an incentive to optimize the catch selectivity of their fishing operations. The experiences gained from a Danish trial indicated that fishers did in fact change their behaviour to avoid fishing grounds where large proportions of small cod were being caught (Kindt-Larsen et al., 2011).

As CQM pertains to all commercial species caught within a given fishery, fishing must cease when the least plentiful quota – the "choke species" – is exhausted. While CQM stops excess fishing mortality due to discarding, it presents a new issue for fishers of how to fully utilize the quotas they have been allocated. In mixed fisheries this may result in the underutilisation of certain species. Failing to utilize the plentiful quota because of exhaustion of a choke species will result in a loss of income. Therefore, fishers may use their expertise (knowledge and fishing gear) to influence catch compositions within a certain range, thus optimising their catch in order to maximise their income. Fishers can also lease quotas from other vessels in order to obtain a more desirable quota portfolio (Holm and Schou, 2012). Therefore, vessels fishing under CQM have several ways to optimise their catch and income, while also reducing discards (Holm and Schou, 2012).

6.3.3. Electronic monitoring

The EU Commission has proposed a CQM system as part of the CPF reform. The success of CQM requires appropriate documentation to verify the total catch, the validity of scientific advice, and the implementation of the TACs through national catch quotas (Kindt-Larsen et al., 2011). Electronic monitoring systems (EMS) are

one method which has been proposed to monitor catches under a CQM system (Holm and Schou, 2012). EMSs consist of a sensor, imagery, and control unit (Kindt-Larsen et al., 2011; Stanley et al., 2011; Ames et al., 2007). EMSs can be used to verify that all catches taken on board a vessel are accounted for. This gives a greater confidence in the levels of fishing mortality documented and minimises discarding. By defining and being able to record exactly how much of a species is caught, there should be no need for any other restrictions, including effort restrictions (Dalskov et al., 2011). The effectiveness of EMSs varies among fisheries, but the technique has been successfully applied in monitoring a range of issues including fishing locations and times, catches (discarded and retained), fishing effort, protected-species interactions, and mitigation measures (Stanley et al., 2011). EMSs deliver much the same data as on-board observers, except for the accuracy of discard weights (Kindt-Larsen et al., 2011). They can potentially be used in a number of fisheries management applications such as monitoring for protected species bycatch, monitoring incidental capture of seabirds, monitoring activity in and around closed areas and the provision of enhanced scientific data for improved stock assessments and the determination of trigger levels for RTCs (Kindt-Larsen et al., 2011). The development of measuring software may also provide the possibility to collect length frequency data for scientific use. EMSs can also assist in achieving an ecosystem approach to fisheries management by monitoring the catch of all species within fishing fleets (Ames et al., 2007). EMSs provide a cost-effective means of achieving a wide range of monitoring functions (Ames et al., 2007; Kindt-Larsen et al., 2011). Electronic monitoring also provides a means for fishers to be able to demonstrate good practice, particularly in respect to demonstrating discard reduction or elimination, improved selectivity, and avoiding juveniles.

6.3.4. Incentive based management

One of the most important factors associated with successful implementation of new regulations is the introduction of appropriate incentives. Incentives available to management include increased quota share, unrestricted effort, and access to commercially important fishing grounds that are otherwise closed (Madsen and Valentinsson, 2010). Such incentives can be used to facilitate a faster shift in gear use and greater acceptance of selective gears (Graham et al., 2007; Krag et al., 2008; Valentinsson and Ulmestrand, 2008; Catchpole and Gray, 2010). The introduction of certification and eco-labelling schemes, such as Marine Stewardship Council (MSC), KRAV, and Friend of the Sea (FOS) in fisheries also provides an incentive to fish sustainably. In response, fishers attain higher prices for their eco-labeled products compared to other products (Kaiser and Edwards-Jones, 2005). It is becoming increasingly recognised that clean catches and an environmentally friendly image can have economic benefits (Hall and Mainprize, 2005). Eco-labelling can also help transform the management system from a top down approach where managers implement strict regulations to a more bottom up approach where fishers are actively participating in fishing sustainably. Finally, a necessary condition for any successful regulation is industry support.

6.3.5. Stakeholder involvement

In recent years, the Danish management system has undergone substantial changes, which to a large extent have been undertaken with stakeholders (Andersen et al., 2010). Fishers are the most qualified people to develop and improve discard mitigation techniques (Gilman et al., 2006). Their incorporation into the decision making process can help the industry to be seen as taking an active role in improving their activities, as well as providing the possibility for scientists and managers to fully utilise industry's unique practical knowledge to finding effective and practical solutions (Kennelly, 1995; Gilman et al., 2006). The earlier fishers are involved in all facets of the decision making process, the sooner and more complete will be the voluntary acceptance of bycatch reducing fishing technology, and the smoother the implementation of the relevant legislation (Kennelly, 1999; Hall and Mainprize, 2005). Kennelly and Broadhurst (1995) argue that one of the most important tasks is the promotion of industry acceptance and adoption of the recommended decisions. Further benefits of stakeholder involvement include:

i)   increased sense of ownership, encouraging responsible fishing;

ii)   enhanced management through use of local knowledge;

iii) increased compliance with regulations through peer pressure;

iv) improved monitoring, control and surveillance by fishers;

v)   collective ownership by users in decision making; and

vi) greater sensitivity to local socioeconomic and ecological restraints (Gutierrez et al., 2011).

The inflexibility of most regulatory systems provides fishers with little possibility to develop and test more-selective fishing practices. To alleviate this problem, Madsen and Valentinsson (2010) suggest a form of legislation should be considered that makes it possible to test selective devices during certain periods without further commitment. This would help to overcome the technical problems that often arise during the initial stage of commercial operations. They propose a system of temporal derogations for vessels willing to test new gears (Madsen and Valentinsson, 2010). The data that could potentially come from such a system would help to understand whether difference exist between selectivity trials on-board commercial vessels and actual commercial fishing. It would also help to better understand the variability in selectivity that occurs under commercial settings. Pilot programmes are commonly used to test new more-selective fishing practices. There use also provides the possibility for fishers and other stakeholders to be actively involved in the development of management strategies (Catchpole and Gray, 2010). Pilots provide a framework for industry to develop solutions acceptable to them, and therefore increase the likelihood of uptake and compliance with new measures (Catchpole and Gray, 2010). Participation from stakeholders in all stages of pilots; initiation, validation, completion and implementation is essential (Kennelly and Broadhurst, 2002).

6.3.6. Unselective fishing

From an economical and technical point of view, selective fishing makes sense and is often desirable. It reduces sorting times, reduces the complexities of handling and processing the mixture of species and sizes, and provides more space on-board for higher valued commercial species, just to name a few (Hall et al., 2000). However, from an ecological point of view, there is no experimental or theoretical evidence showing that highly selective fishing is the best or least harmful way to extract a sustainable harvest from an ecosystem (Hall et al., 2000). It is argued that selective fishing alters the existing community structure, spectrum of biodiversity, and species and size diversity (Zhou, 2008; Rochet et al., 2009; Zhou et al., 2010). Consequently, reducing discards as much as possible may not only be unnecessary but may even result in failure to achieve some of the objectives associated with ecosystem-based fisheries management (EBFM) (Zhou, 2008).

7. Final remarks and future work

In most fields, scientific and technological advances are often patented and taken up almost directly by the industry. In fisheries this is often not the case. Scientific and technological advancements within most industries result in improved performance at either a lower economical cost or the cost of the improvements are offset by an increase in economic efficiency. In fisheries, scientific and technological advancements often result in economic losses in the short-term which deters fishers from implementation. This is due to the developments either focusing on improving the sustainability of the ecosystem, the long-term economic gains or a combination of both. For alternative discard management approaches to be successful, the economic benefits to stakeholders, especially over the short-term need to be considered.

The present work (**Papers I-IV**) indicates that discards are influenced by a multivariate and complex range of factors that differ for each species. These factors are unique for a given fishery, area, and can even be unique from vessel to vessel. Such complexity means that unique solutions need to be defined for each fishery.

Paper II demonstrated that there are many factors that contribute to discards and the effectiveness of gear-based technical measures. If catch losses are great, fishers will attempt to improve their economic situation. Therefore, the successful implementation of gear improvements, as indicated by this study, depends on overcoming the short-term economic losses associated with their use and the fishers' acceptance. When short-term catch losses are too large and go unaccounted for, gears will be manipulated, rules circumvented and the potential long-term gains may never materialise. This clearly highlights the importance of incorporating stakeholders into the management and decision making process.

The significant spatial and temporal variations in discard rates that have been observed in **papers I-IV** demonstrate the importance of incorporating the spatio-temporal aspect of discarding into management. Spatio-temporal management of discards has largely been neglected in the past. However, the emergence of real-time closures within EU fisheries is one such method that attempts to solve this issue. It also provides the possibility to introduce economic incentives for fishers to adopt more-selective fishing techniques. Through the introduction of a discard ban and a CQM system, the importance of the spatio-temporal aspect of discarding will only increase. However, this will hopefully provide for a more proactive bottom up approach where fishers will take an active role in reducing discards.

The modelling approaches used throughout the thesis provide the possibility to answer a range of additional questions.

- What effect did the shift from common pool quotas to individual transferable quotas have on discards?

- What effect has the introduction of more-selective gears (SELTA) had on discard rates in the Kattegat?
- How has the closed area in the Kattegat affected discard rates?

The direction for future discard studies will largely depend on the future developments within EU policy. A discard ban is in the process of being introduced into EU fisheries. Therefore, the question is no longer whether a discard ban is the correct way of approaching the issue of discards, but how can such a management system be implemented correctly from its point of inception. The introduction of a discard ban under a CQM system opens up a range of new questions that need to be answered in relation to discards. For example:

- How will the additional removal of discards alter the functioning of the ecosystem?
- What is the suitable amount of monitoring required?
- How to incorporate stakeholders into the management and decision making process with regards to a discard ban and CQM?
- What are the implications of a CQM under an ITQ system? Will it reduce high-grading?
- How to monitor catches under a CQM system?
- Are EMSs the correct way to approach a CQM.
- What are the economic incentives to fishers under a discard ban?
- Will being able to document catches and discards under a CQM system with electronic monitoring offer fishers the possibility to obtain a higher price for their catch?
- How will previously discarded individuals be utilised? Are there sufficient capabilities within Denmark to be able to cope with the additional landings?
- Will markets need to be set up? Will the emergence of new markets result in new species being targeted and their stocks declining?

Finally, eliminating discards completely is an unrealistic challenge if trawling is to have a place within commercial fishing. The objective of minimising discards should be to assure the protection of all species, including rare, endangered and vulnerable species.

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Paper I

PLoS one

Fishery Discards: Factors Affecting Their Variability within a Demersal Trawl Fishery

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Abstract

Discards represent one of the most important issues within current commercial fishing. It occurs for a range of reasons and is influenced by an even more complex array of factors. We address this issue by examining the data collected within the Danish discard observer program and describe the factors that influence discarding within the Danish Kattegat demersal fleet over the period 1997 to 2008. Generalised additive models were used to assess how discards of the 3 main target species, Norway lobster, cod and plaice, and their subcomponents (under and over minimum landings size) are influenced by important factors and their potential relevance to management. Our results show that discards are influenced by a range of different factors that are different for each species and portion of discards. We argue that knowledge about the factors influential to discarding and their use in relation to potential mitigation measures are essential for future fisheries management strategies.

Introduction

Discards refers to the organisms of both commercial and non-commercial value that are caught during commercial fishing operations and returned to the sea, often dead or dying [1]. The practice of discarding occurs for a range of reasons, including, individuals caught are under the minimum landing size, species have a low or no market value, catch is damaged or is high-graded (i.e., lower valued individuals are discarded to maximize profits), or the species quota is reached. Three fundamental causes are responsible for the high level of discarding in European Union (EU) fisheries, namely the use of unselective fishing techniques, the failure to reduce fishing effort, and biological and environmental factors affecting the distribution of species [2]. A multitude of other factors also affect the practice of discarding, such as complex social [3], technical [4,5], economical [6–8], and legislative [9] reasons. In addition, the effect and relative importance of these factors will vary for different species, vessels, metiers (*fishing operations characterised by the same fishing gear and catch composition*) and fleets, and will fluctuate over time [10] and space [9]. As a further source of variation there is the individual choice by fishermen as to which part of the catch to retain and which to discard [3,10].

Furthermore, discarding has wider implications whereby ecosystem functioning and its biodiversity are negatively affected [11]. There are indications that discarding has altered the ecosystem functioning of some seabird communities [12,13] and has negative effects on charismatic and endangered species [14]. The European Commission (EC) considers discarding to be negative, both in terms of ecosystem functioning and economic

viability, and is committed to eradicating the problem [11]. The Common Fisheries Policy (CFP) reform, to be introduced in 2013, is set to eliminate the problem of discards through the introduction of a discard ban [15].

The most complex discard problems are found in mixed-species demersal trawl fisheries, and are responsible for most of the discards [2,3]. In the Kattegat, the demersal trawl fishery, the focus of this study, is the dominant gear type, accounting for approximately 80% of all fishing effort [16]. The fishery has been faced with regulatory measures for the recovery of the Kattegat cod (*Gadus morhua*), which has largely been unsuccessful so far [17]. The small mesh sizes currently and previously employed in the Kattegat are used to retain Norway lobster (*Nephrops norvegicus*) and sole (*Solea solea*). This may lead to high discard rates of juvenile round and flatfish species [17–19]. A similar occurrence has been observed in the North Sea beam trawl fishery for sole where high discarding of plaice (*Pleuronectes platessa*) occurs [20–22].

The present literature on discards has mainly been descriptive, with a focus on understanding discard rates of specific species [23], estimating the amount or proportion of total catch discarded from particular fisheries [9,24], species and length compositions of discards [6,25,26], as well as global discard estimates [14,27]. While these studies help provide a better insight into the discarding problem there is a lack of quantitative studies regarding discarding behaviours [4]. Therefore, further knowledge of the factors that influence discard rates is needed. Studies of such nature can help to gain an insight into the factors influencing the discarding process, and to predict future catches and discards [28]. The use of modelling approaches to discard data provides the

possibility to further disentangle the effects of different drivers [29] and offers important insights into the potential effectiveness of technological and area/time management measures for reducing fishery discards [30]. Modelling approaches can also prove useful in the process of systematically reducing by-catch in multispecies fisheries, complementing classic mesh retention experiments as aids to the development of strategies to reduce discard levels [30], and providing support for mesh size increases. Modelling discards is an area that has received little research effort, but requires research [4,29,31].

Past studies that have modelled the relationship between explanatory variables and discards have focused on discards as a whole [4,28,30,31] without the consideration that different portions are discarded for different reasons. For example, discarding of individuals under minimum landing size (<MLS) mainly occurs as a result of the MLS and juvenile abundance, while larger individuals, those above minimum landing size (≥MLS), are discarded for a variety of different reasons, including landing composition regulations, no quota, market forces, damaged, or the species has a low or no economic value. There are also many environmental [4] and vessel/gear specific parameters that have also been suggested to influence discard rates [17–19]. These will act differently on different portions of discards and may only influence one portion or species.

The high number of potentially influential factors stems from the complexity of the social, economic, management, and environmental forces acting on the system. If a theory of discards is to emerge, all potentially relevant factors should be considered [4]. This study elaborates on the existing knowledge by considering a larger number of factors that can potentially influence the variability of discards < and ≥MLS. We apply a Generalised Additive Model (GAM) using discard data from the Danish discard observer programme for the demersal trawl fishery in the Kattegat to identify the driving factors that influence discarding practices and those that could potentially be important for the development of management strategies.

Materials and Methods

Discard data

Since 1995 Denmark has collected data on catches and discards with the aim of sampling all demersal fisheries except the ones with very limited fishing effort and discard. In 2002 the EU identified the need to describe and quantify discards as part of the European Data Directive (1639/2001 and 199/2008). The data collected is stratified with regards to ICES area, quarter, and discard pattern of the relevant fisheries (e.g. fisheries with low discards are seldom sampled). Participation in the discard sampling programme is opportunistic, i.e. permission by the skipper is required, and as the observer has no relation to the control unit, the fishing practice is assumed to be unaffected by the observers presence. In order for the sampling programme to be representative of the fisheries in question, vessels of all sizes are sampled from all the main fishing harbours during the entire period of activity of a given fishery. Biological information (i.e., lengths, weights and otolith samples) are collected from the catch, together with vessel, gear, geographical position and environmental attributes (depth, bottom type).

For each observed haul, an estimate of the total catch weight is made by the fishermen and the observer in collaboration. The total catch is then sorted into the retained and discarded components by the commercial fishermen. The total weights of each individual species retained are recorded. If the abundance of a species is small, total numbers and lengths are recorded,

otherwise a subsample is taken, numbers and lengths recorded and raised accordingly. The total weight of the discarded portion is approximated, a subsample taken, and then sorted by the observer into species. Total weights and numbers of each discarded species in the subsample are determined and raised based on the total approximated discarded weight.

Between 1997 and 2008, 189 trips and 370 hauls were sampled within the Danish demersal bottom trawl fleet active within the Kattegat (Figure 1). The fishery was classified into mesh size categories (full mesh opening); 70–89 mm, 90–99 mm, and 100–120 mm. This was in order to understand the effect of mesh size while ensuring a reasonable number of observations within each size category. Initial analysis on the relative importance, in terms of landings (by weight), revealed that cod, plaice and Norway lobster were the 3 main commercial species targeted by the Danish demersal trawl fleet in the Kattegat (Figure 2). Sole, while caught in relatively low numbers, was the second most important species economically. Due to the large difference in landings from other species and the fact that Norway lobster is one reason why small mesh sizes are used, these 3 species are the main focus of this study. The Kattegat demersal trawl fishery is a mixed fishery and has the most comprehensive discard sampling in relation to other fisheries in the area, namely Danish seines and static gears (mainly gillnets).

Regulations

Regulations throughout the 12 year study period changed considerably. Mesh sizes increased and square mesh panels to improve selectivity became an option in the legislation. In 2005
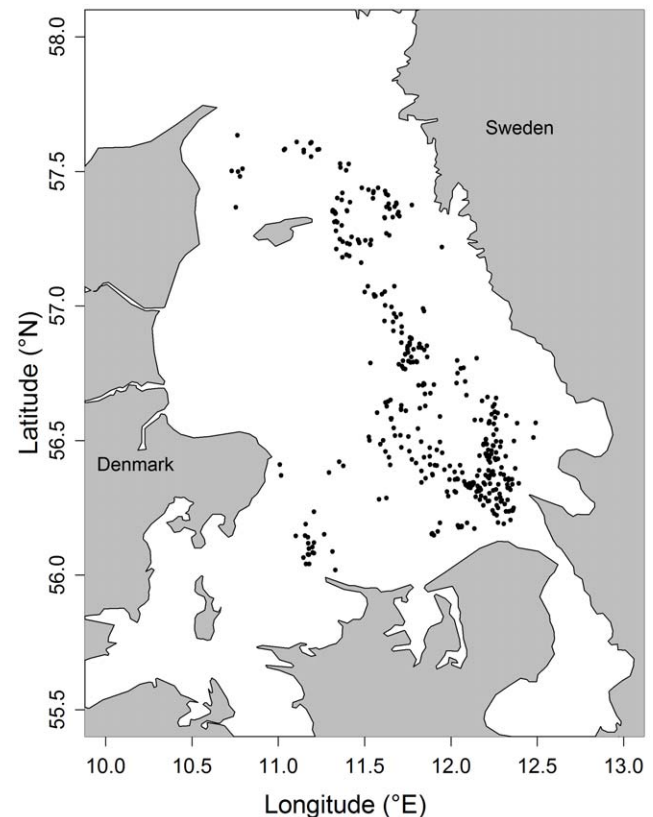

Figure 1. Map of the study area with the location of discards observer hauls used in the analysis. Locations represent the midway point of each haul.
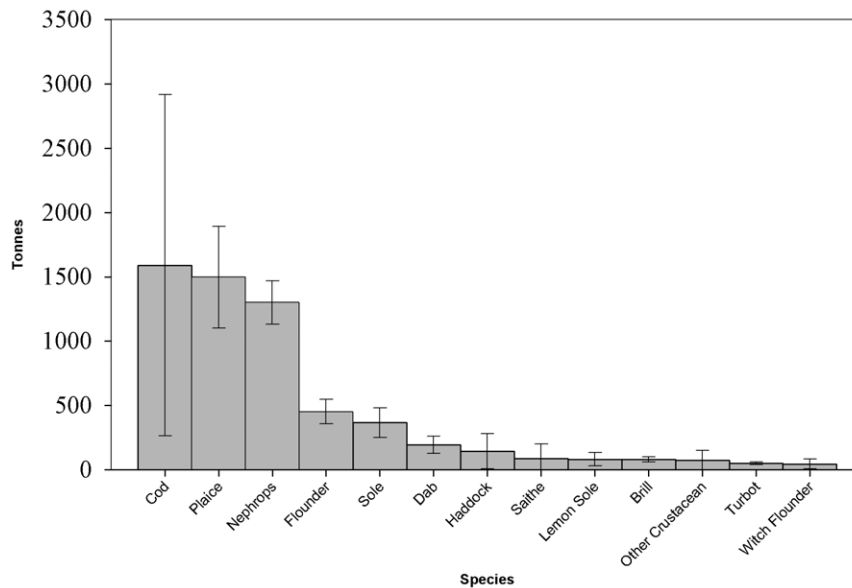doi:10.1371/journal.pone.0036409.g001

Figure 2. The official Danish landing statistics (www.fvm.dk) for the Kattegat: 1997 to 2008. Species are ranked according to their relative importance by weight. Average value in thousands with standard deviation for the 12 years investigated.
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the minimum diamond-mesh size in codends used in the Kattegat increased from 70 to 90 mm, unless a sorting grid is used in combination with a 70-mm square mesh codend (EC Council Reg. 27/2005). A 120-mm square mesh panel inserted in a 90-mm codend was introduced as a voluntary option in the legislation from 2005 [19]. From our knowledge, uptake of the window was very minimal by the industry in the Kattegat. A previous study found no significant improvement in selectivity for the three species investigated here [18]. Therefore, mesh sizes categories are grouped regardless of whether or not selectivity windows were present. Quotas were split into fortnightly rations which were continuously adjusted to the amount of quota left. In 2007 individual transferable vessel quotas were implemented in the Danish demersal fishery whereby a vessel is allocated an annual quota for each species. The cod quota decreased tenfold over the 12 year study period; from 5170 tonnes in 1997 to 465 tonnes in 2008. The quotas for plaice and Norway lobster both remained relatively stable during the same period; the plaice quota fluctuated around 2200 tonnes±500 tonnes; while the Norway lobster quota increased marginally, from around 3500 tonnes in 1997 to around 4000 tonnes in 2008. The MLS for the three species remained unchanged throughout the study period (i.e., cod 35 cm, plaice 27 cm, Norway lobster 13 cm (4 cm) total length/carapace.

Statistical analysis

To account for the unbalanced sampling design between explanatory variables, and describe the main spatial distribution changes over time, generalised additive models (i.e. GAMs, [32] were used. A quasi-Poisson distribution (log-link) was used because the data are counts without an upper limit, and overdispersed (i.e. variance exceeds the mean or contain a large number of zero observations). The quasi-likelihood approach assumes that the scale parameter Φ of the distribution is unknown, which makes it more suitable for over-dispersed data than the classical Poisson distribution [33]. The variance of a quasi-Poisson model is a linear function of the mean [34]. Rather than use density of discards (numbers per hour) as a response variable, we chose to model

numbers discarded per haul with the use of an offset variable (haul duration). The advantages of the offset approach compared to analysing densities are that the fitted values are always positive, the confidence intervals around the fitted values do not contain negative values, and we allow for heterogeneity within the context of a Poisson distribution [35]. Of further interest was what effect vessels had on discards, although we are not interested in knowing the exact nature of the vessel effect. Therefore we include vessel as a random effect. Here we assume that the variation around the intercept, for each vessel, is normally distributed with a certain variance.

A large number of potential variables were considered for each of the models and through exploratory analysis and a stepwise deduction using a priori knowledge a total of 11 variables were included in the analysis (Table 1). Some variables were only available or specific for a species or a subcomponent and therefore not included in all models, i.e. Juvenile abundance was only available for cod <MLS while quota utilisation was available for the cod and plaice models ≥MLS. To simplify the interpretation of the results, the maximum degrees of freedom (measured as number of knots k) allowed to the smoothing functions were limited for the variables total catch weight, juvenile abundance, vessel power and depth (k = 4). The full model was formulated as follows:

$$\text{Numbers discarded per haul} = \beta + s(\text{year}) + s(\text{long, lat}) +$$
$$s(\text{juvenile abundance}) + \text{quarter} + \text{mesh size category} +$$
$$s(\text{total catch weight}) + s(\text{vessel power}) + s(\text{depth}) +$$
$$s(\text{quota utilisation}) + \text{offset}(\text{haul duration}) +$$
$$\text{random effect}(\text{vessel}) + \varepsilon$$

where β is an overall intercept, s is an isotropic smoothing function (thin-plate regression spline), and ε is an error term.

Table 1. Summary of variables included in Generalised Additive Models of factors influencing discards.

Variable	Description	Comments
Year	Year haul was sampled	1997–2008
Quarter	Quarter haul was sampled	
Depth	Mean fishing depth of haul	In meters
Longitude, Latitude	Mean Longitude/Latitude of haul	In decimal degrees
Mesh size category	Codend mesh size of trawl	70–89 mm, 90–99 mm, 100–120 mm
Juvenile abundance	Abundance of age 1 and 2 individuals per quarter[1]	
Total catch weight	Total catch of all species	in kilograms
Quota utilisation	Amount of quota left[2]	
Vessel power	Engine size of vessel	Used as a proxy for the size of the trawl
Haul duration	Haul duration in hours	Used as an offset term
Vessel	Unique code for each vessel	Used as a random effect

Based on data collected as part of the discard sampling programme 1997–2008.
[1]Juvenile abundance was only available for cod.
[2]Quota utilisation was only assessed for cod and plaice. The quota for Norway lobster is not restrictive and therefore does not affect discards.
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The effect of juvenile abundance on discard rates was only tested for cod discards <MLS. No stock assessment is carried out for Norway lobster, and the plaice assessment is for Kattegat and Skagerrak combined, hence the variable year was used instead to account for the year effect. The variable "juvenile abundance" per quarter was calculated by applying a simple exponential decay function based on the relative number of individuals of age 1 and 2 caught during Baltic sea International Trawl Surveys (BITS) undertaken in the first quarter of each year. Natural mortality (M) and Fishing mortality (F) were taken from the official assessment (ICES, 2011) where we assume that M is constant during the year and F increases linearly during the year (this is to account for the growth, and subsequent increase in retention by the fishing gear, of an individual throughout the year). One year old cod are approximately 18 cm in length, which corresponds approximately to the L50 (length at which fifty percent of the fish are retained in the cod-end) of the smallest mesh size (70 mm) used [17]. It is also assumed that fish of age 2 in quarters 3 and 4 have length ≥MLS and are thus excluded from the index of juvenile abundance.

All potentially important covariates were included in the initial model where the least significant covariates were removed one at a time until all covariates were significant (P<0.05). The final models are then reduced versions of these full models. The analyses were performed using R software, a statistical environment for computation and graphics (http://www.r-project.org), and the R package 'mgcv' [36].

Results

A summary of the discard data is presented in Table 2. A total of 370 demersal hauls were analysed over the period 1997 to 2008 in the Kattegat. All models considered are presented in Table S1. The final models together with each covariates degrees of freedom, significance level and the deviance explained by the model are presented in Table 3. The final models explained between 49 and

Table 2. Summary of discard data collected onboard demersal trawls in the Kattegat for the three mesh size categories.

	Mesh size categories		
	70–89 mm	90–99 mm	100–120 mm
Years (no.)	8	9	11
Vessels (no.)	19	12	9
Horse power (kW)	331.8 (104.6)	367.1 (119.7)	388.0 (108.4)
Haul duration (hrs)	6.5 (1.8)	6.2 (1.7)	5.2 (1.6)
Hauls (no.)	168	132	70
Avg. catch weight (kg)	625.8 (393.2)	576.5 (307)	953.9 (713.2)
Avg. discard cod <MLS (no./hour)	31.2 (44.3)	14.9 (18.2)	13.8 (18.4)
Avg. discard cod ≥MLS (no./hour)	0.5 (1.2)	0.9 (3.4)	1.9 (3.6)
Avg. discard plaice <MLS (no./hour)	64.7 (104.1)	43.0 (56.4)	32.7 (54.7)
Avg. discard plaice ≥MLS (no./hour)	1.3 (4.1)	1.7 (6.1)	14.2 (25.7)
Avg. discard Norway lobster <MLS (no./hour)	450.5 (605.44)	273.6 (385.3)	6.6 (39.1)
Avg. discard Norway lobster ≥MLS (no./hour)	17.0 (75.3)	10.5 (20.2)	0.2 (1.2)

Standard deviations are in brackets.
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Table 3. Final models together with each covariates degrees of freedom, significance level and the deviance explained by the model.

model	α	βmesh	βquarter	s(Yr)	s(catch)	s(JuvAb)	s(lon,lat)	s(quota)	s(vessel kW)	s(depth)	s(Vessel)	DEV.EXPL(%)
cod												
<MLS	−2.97**	70–89 (1.85)**			2.96**	2.98**	22.08**					63.3
	90–99 (1.86)**											
≥MLS	−4.15**		Q2 (−0.30)	7.07**	2.82**		19.41**	2.80**				64.5
			Q3 (−2.20)**									
			Q4 (−1.49)*									
plaice												
<MLS (a)	−1.54**	70–89 (1.20)**		6.96**	2.42**		19.57**					61.8
	90–99 (1.03)**											
<MLS (b)	−1.21**		Q2 (0.59)**	5.97**	2.59**		19.59**					61.3
			Q3 (0.69)**									
			Q4 (0.97)**									
plaice ≥MLS	−2.30*	70–89 (−2.25)**	Q2 (−1.12)*	6.04**	1.60**		21.81**	2.77**		1.00*	18.46**	81.8
	90–99 (−1.62)**	Q3 (−2.32)**									
		Q4 (−1.54)									
Norway lobster												
<MLS	−2.54**	70–89 (1.38)*	Q2 (1.73)**	7.92**	2.63**		22.21**				20.32**	83
	90–99 (0.91)	Q3 (1.83)**									
		Q4 (1.75)**									
≥MLS	−6.07**	70–89 (2.37)*	Q2 (1.50)**	7.65**	2.05**		18.25**		1.32**	1.00*		49.2
	90–99 (1.78)	Q3 (1.91)**									
		Q4 (1.29)*									

Significance levels: 0.001 '**' 0.01 '*', α = intercept, mesh = mesh size category, quarter = quarter of the year, Yr = year, catch = Total catch weight, JuvAb = Juvenile abundance, lon = longitude, lat = latitude, quota = species quota utilisation, vessel kW = vessel power, depth = mean fishing depth, vessel = individual vessel id.
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83% of the deviance. Visual analysis of the model residuals revealed no violation from any of the model assumptions (i.e., normality and homogeneity of variance). The residuals were also inspected for spatial autocorrelation.

The GAMs showed that the relative importance of each variable was different for each species and portion of the discards, with a few similarities (Table 3). In all the final models total catch weight and the interaction between longitude and latitude had a significant effect on numbers discarded. A significant positive relationship between total catch weight and the amount of discards was observed for plaice discards < and ≥MLS and cod discards ≥MLS (Figures 3 & 4). A positive relationship was also observed for cod discards <MLS, however, only for total catch weights up to 1000 kg. For Norway lobster < and ≥MLS, discard numbers tend to decline after a certain point with increasing catch weight, although large uncertainty is associated to large catch estimates (Figure 5). The potential effect of few very large catches (i.e., >3000 kg) has been investigated by removing these observations and refitting the model. The effect of large catches was found to be irrelevant on the dome-shape response of discards rates to the total catch weight.

The effect of the spatial component was consistent for the two portions of the discards within each species, but marked differences were found among the species. Discards of cod were highest in the central eastern part of the Kattegat along the Swedish coastline and in the south western region close to Denmark. Plaice discards on the other hand exhibited a marked longitudinal gradient, with increasing discards westward. The lowest discards of Norway lobster occur in the south with a bimodal increase northward. However, an opposite local effect was observed for the two portions of the Norway lobster discards in the north-west tip of the study area. Here discards <MLS are at their highest while discards ≥MLS exhibit a decreasing trend in a westerly direction. Juvenile abundance had a significant positive effect on the discard rate of cod <MLS (Figure 4; Table 3). Norway lobster discards were highest during the third quarter (Table 3; Figure S1). Plaice discards were highest during the winter months; quarters 4 and 1 for subcomponents < and ≥MLS
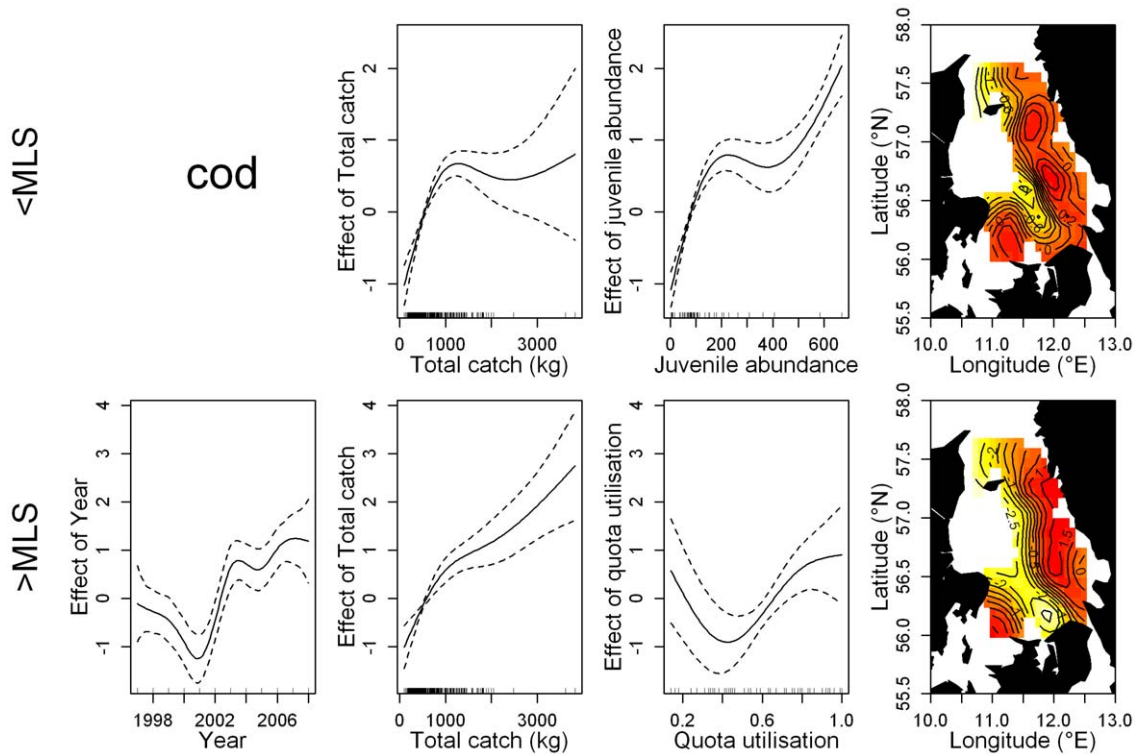
Figure 3. Effect of the significant smoothing functions (solid line) on the discard rate of cod in the Kattegat demersal trawl fishery. Cod <MLS (top row) and ≥MLS (bottom row). Dotted lines represent the 95% confidence limits. Vertical bars along the x-axis indicate observational values. The surface and contour lines describe the effect of 2-d smoothing function on the geographical coordinates.
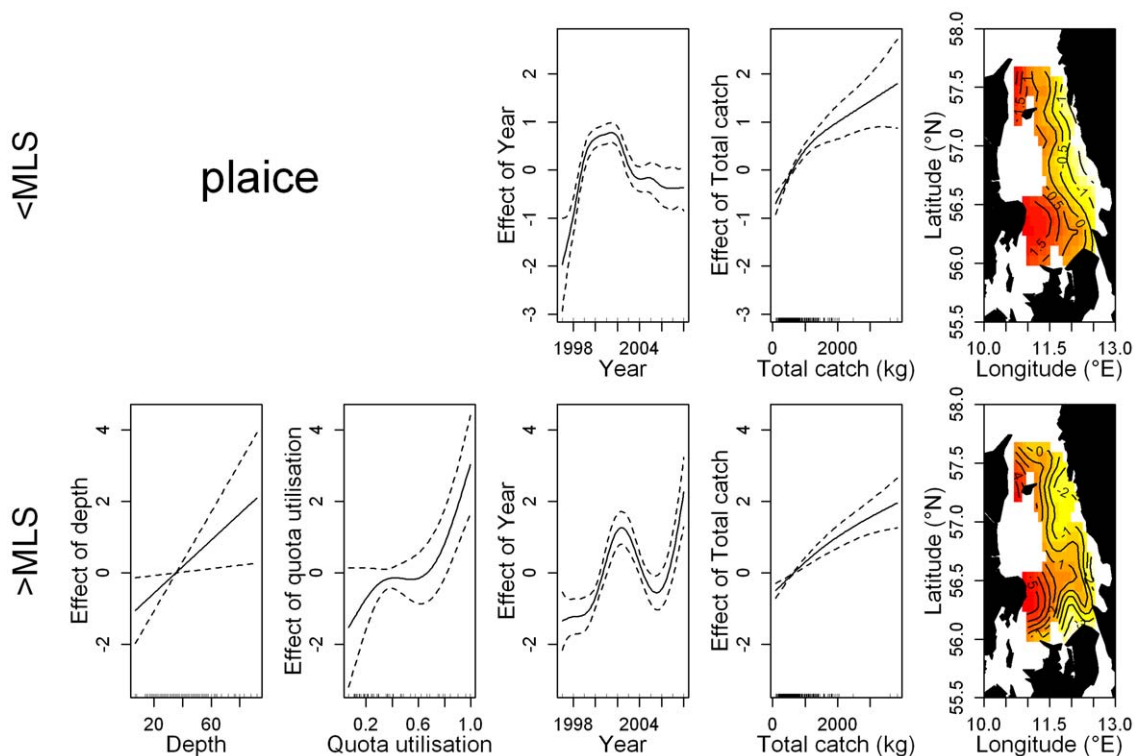doi:10.1371/journal.pone.0036409.g003

Figure 4. Effect of the significant smoothing functions (solid line) on the discard rate of plaice in the Kattegat demersal trawl fishery. Plaice <MLS (top row) and ≥MLS (bottom row). Dotted lines represent the 95% confidence limits. Vertical bars along the x-axis indicate observational values. The surface and contour lines describe the effect of 2-d smoothing function on the geographical coordinates.
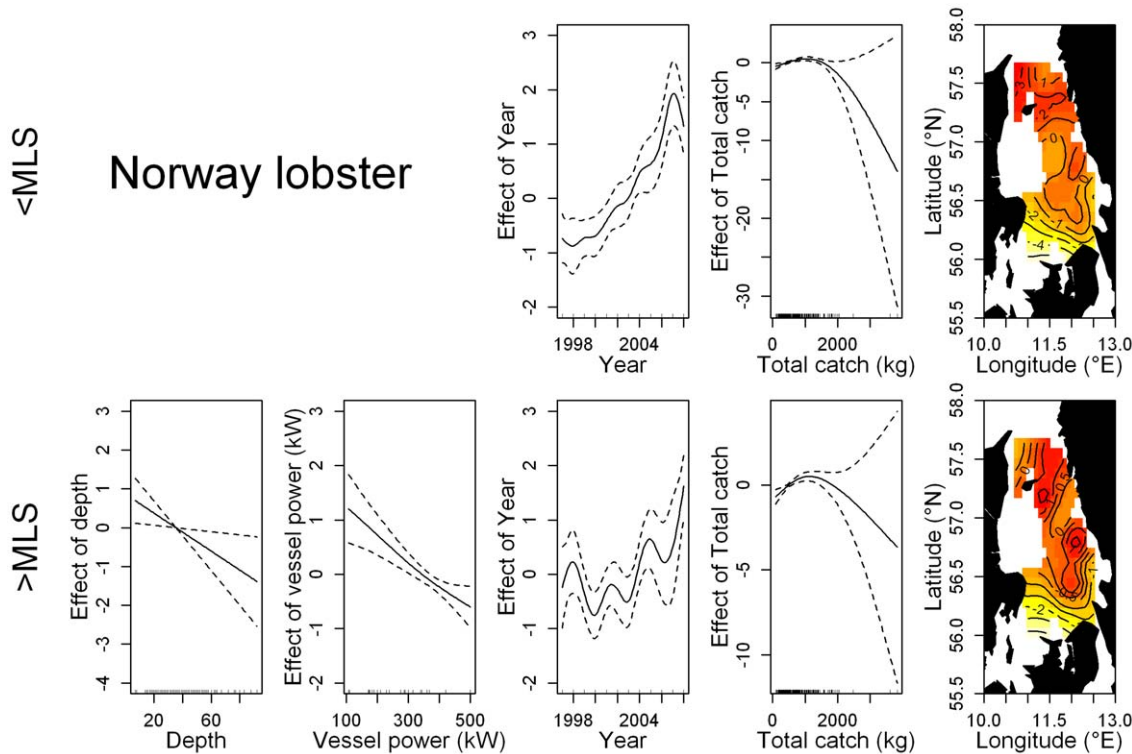doi:10.1371/journal.pone.0036409.g004

Figure 5. Effect of the significant smoothing functions (solid line) on the discard rate of Norway lobster in the Kattegat demersal trawl fishery. Norway lobster <MLS (top row) and ≥MLS (bottom row). Dotted lines represent the 95% confidence limits. Vertical bars along the x-axis indicate observational values. The surface and contour lines describe the effect of 2-d smoothing function on the geographical coordinates.
doi:10.1371/journal.pone.0036409.g005

respectively. Cod discards ≥MLS were highest in the first quarter of the year. Quota utilisation was assessed for cod and plaice discards ≥MLS and found to be highly significant. A positive linear relationship was observed for plaice, while cod discards first declined before increasing as the quota was fished up. Mesh size category was found to have a negative relationship with the amount of discards in all models except for cod discards ≥MLS (Table S1). Depth was non-significant in a majority of the models. For plaice ≥MLS, discarded numbers increase as depth increases while the opposite was observed for Norway lobster discards ≥MLS.

For plaice discards <MLS we end up with two competing final models. When mesh size category and quarter were both included in the model they compete with one another, resulting in non-significant parameters. While when either one was dropped from the model the other one becomes highly significant. Their contributing effects cannot be estimated simultaneously. This is most likely a result of the heterogeneity in the sampling across mesh sizes and quarters.

Discussion

Knowledge about the reasons why discarding occurs is considered a key element in the progress towards a theory of discarding [4]. We demonstrate that discard rates of the 3 main target species in the Danish Kattegat demersal trawl fishery are influenced by a multivariate and complex range of factors that differs for each species and their subcomponents. Previous studies that have investigated the factors influencing discards did not consider the size composition of the discarded catch, nor distinguish between the reasons that may drive discarding fish of

different sizes. Some factors may only be able to influence one subcomponent.

Dealing with discards ≥MLS is a much more problematic task as these are influenced by a range of factors that differ for vessels, fleets, seasons, area and species. Identifying the main influential factors of discards ≥MLS is also much harder. It is difficult to distinguish whether a vessel is discarding marketable fish due to market forces, low or no available quota or the individuals are damaged. Discards ≥MLS are often a result of market or regulatory constraints from the quotas and rations in place. Quotas in the Danish demersal fisheries for years 1997–2006 were split into fortnightly rations which were continuously adjusted to the amount of quota left. As the quotas were fished up the rations were reduced to try and sustain the quota throughout the year. This may explain why the numbers of cod discarded ≥MLS begin to increase after approximately 50% of the quota is fished up. A previous study found that over-quota discarding occurred towards the end of the year [37]. However, if other management regulations restricting the landing of a species are in force, such as a ration system, discarding of individuals ≥MLS may take place earlier in the year when these become restrictive. Regulatory discards <MLS are however controllable to a degree, based on factors such as mesh size, area fished and others influencing their magnitude [30].

In areas and/or periods when the abundance of individuals between minimum retention length and MLS is high, discards will subsequently be high. It would be beneficial to introduce management regulations to restrict the catching of a species until these individuals have reached a length ≥MLS. The obligation for vessels to move fishing ground, real time closures and areas closures are potential measures that could achieve this.

Total catch weight of cod and plaice was found to be positively correlated with discard numbers while a negative correlation was found for Norway lobster. For cod discards <MLS a positive trend was evident up to around 1000 kg where it appears that some sort of saturation point is reached. Similar trends have been found in selectivity trials where L50 increases as catch weight increases beyond a certain threshold [38]. This could be attributed to the meshes directly in front of the catch becoming stretched open, resulting in better selective properties for smaller individuals. This is where gear selectivity has the potential to improve, through the development of gears that provide more stable selectivity. A similar trend was not observed for plaice. It is possible that selectivity does not improve for plaice, and potentially other flatfish species, as catches increase, due to their morphology. Flatfish morphology likely fits better to a relatively closed diamond mesh. Size selectivity of Norway lobster is somewhat more difficult to achieve as it is largely dependent on the way the individuals come in contact with the meshes [39]. However, square meshes have been found to improve the selectivity of Norway lobster [40]. Codends with multiple escape areas having different mesh shapes seems to be a way to improve selectivity of both species [40].

The mismatch between gear selectivity and minimum landing size is a significant contributor to discards, especially for those <MLS [17,18]. In the Kattegat the minimum mesh size was increased from 70 mm to 90 mm in 2005. While this change was substantial, the 90 mm mesh size still results in high levels of discarding. Selectivity can also be affected by other factors than mesh size, for example twine thickness [17], which are not recorded in discard data. An additional increase in the mesh size would further reduce discards while also causing a loss of other commercially important species that are relatively small, namely sole and Norway lobster. This may suggest that benefits for both reducing discard rates and maintaining valuable catches could be derived from the use of efficient species selective devices rather than only mesh size regulations. Recent experiments conducted in the Kattegat suggest that escape windows can be made more efficient in releasing cod [41], and grids can, in general, be used to reject fish bycatch in directed Norway lobster fisheries [18,42].

Seasonal discarding was also found to be an influential factor and can be attributed to the targeting behaviour of the fishermen and the condition/behaviour of species during different seasons. For example, it is observed that plaice ≥MLS are discarded more during the first quarter of the year. This can be attributed to the physical condition of plaice throughout the year. In winter and early spring large plaice are of low condition and watery flesh, resulting in lower market value [37]. Therefore, low-value individuals caught at the beginning of the year will be discarded to save quota for higher valued individuals caught at the end of the year [37]. Avoiding the capture of plaice during the winter months when they are of poor physical condition could reduce the number of plaice ≥MLS being discarded. Norway lobster < and ≥MLS are discarded more during the summer when they are targeted the most, while cod in the Kattegat have traditionally been targeted during the first months of the year when higher densities occur due to spawning [43]. High discarding of cod ≥MLS is also observed when quota utilisation is low. This could be due to the targeting behaviour of the fishermen during the first quarter of the year and subsequently discarding more.

The spatial distribution of discards for the three species observed here were all different from one another. Therefore, when considering new management measures to reduce discards, the spatial distribution of discards, especially those

<MLS, also needs to be considered. Spatial management can provide a useful tool in protecting juvenile fish by reducing discard rates and can serve as a buffer against management errors and recruitment failure [3]. The most consistent benefit from spatial management, however, is that it provides the necessary economic incentive for fishermen to adopt selective fishing techniques that allow them conditional access to otherwise closed areas [3]. Our findings show that the spatial and temporal variability in the discard rates can potentially be exploited in a general strategy to reduce discards. A similar approach was proposed for the USA mixed species otter trawl fisheries of the Georges Bank-Southern New England region [30]. By limiting directed fishing to times and places where resources are segregated, the quantity of unintended catch could potentially be reduced [30].

A discard ban, which has been proposed for EU fisheries as a major change to the CFP, may result in spatio-temporal improvements to the exploitation of the stocks. The capture and subsequent retention of smaller individuals, as would likely be the case under a discard ban, has the potential to reduce economic revenue to the fishermen, depending on how the quotas are restructured. Therefore, under a discard ban, the issue of discarding becomes less of a concern and a set of new issues emerge, such as minimisation of the initial capture of juveniles that would rapidly fill fishing quotas, enforcement, and alterations in the ecosystem functioning, particularly on the sea bird [13] and benthic scavengers [44] that feed on discards at the surface and at the bottom respectively. If implemented correctly, a discard ban should create economic incentives for the industry to reduce the capture of smaller individuals through improvements in gear selectivity and the spatio-temporal distribution of the fishery. Moreover, it would also improve the reliability of scientific stock assessments by removing the current uncertainty associated with the estimation of discards. However, a discard ban also has the potential to encourage misreporting if not properly enforced. Discard bans have proven to be successful outside the EU (i.e., the demersal fishery in Norway, [45]), and its implementation within the EU fisheries will be dependent on understanding and compliance from the industry.

In our study of the Danish demersal trawl fishery it is evident that discards, and their subcomponents, are affected by a multitude of factors that differ depending on what species/subcomponent is being analysed. The same is valid for different fleets, gears, and areas. The factors that have been shown to influence the discard rates of cod, plaice and Norway lobster are highly species-specific and may not hold for other species. Therefore, extending this type of analysis to other discarded species is necessary to explain the overall discard behaviour in a fishery.

Supporting Information

Figure S1 Boxplots of the significant categorical variables of the generalised additive models.
(TIF)

Table S1 Summary of all models fitted together with their significant variables, GCV scores, and the deviance explained by the models. mesh = mesh size category, JuvAb = Juvenile abundance, catch = Total catch weight, lon = longitude, lat = latitude, quarter = quarter of the year, vessel kW = vessel power, depth = mean fishing depth, vessel = individual vessel id, haul dur = haul duration, yr = year, quota = species quota utilisation.
(XLS)

Acknowledgments

Author Contributions

Conceived and designed the experiments: JF VB NM TLC. Performed the experiments: JF VB NM TLC. Analyzed the data: JF VB NM TLC. Wrote the paper: JF VB NM TLC.

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Paper II

ARTICLE

The effect of regulation changes and influential factors on Atlantic cod discards in the Baltic Sea demersal trawl fishery

Jordan Feekings, Peter Lewy, and Niels Madsen

Abstract: The proportion of Atlantic cod (*Gadus morhua*) discarded in the Danish Baltic Sea cod trawl fishery has been as high as 40%. This, combined with a stock that has declined dramatically over the past 30 years, has led to numerous technical regulations being introduced to reduce the capture of juveniles and thus discards. One method that has been widely adopted in the Baltic Sea has been to improve gear selectivity, subsequently allowing young individuals to escape capture. To understand the effects that changes to gear selectivity and minimum landing size have had on discard rates, as well as the effects of a range of additional explanatory factors, generalized additive mixed models were used. Gear regulation changes enforced in the Danish demersal trawl fishery in the Baltic Sea and other factors, such as minimum landings size, juvenile abundance, catch mass, price, and their spatial and temporal distribution, were found to significantly affect discard rates. The newest and currently legislated gears were identified as having the lowest discard rates. The increase in minimum landing size from 35 to 38 cm has increased discard rates.

Résumé : Le pourcentage de morues franche (*Gadus morhua*) rejetées dans les pêches danoises au chalut dans la mer Baltique a déjà atteint les 40 %. Ce fait, combiné à une réduction dramatique du stock au cours des 30 dernières années, a donné lieu à l'introduction de nombreux règlements portant sur les techniques de pêche et visant à réduire la capture de juvéniles et ainsi, les rejets. Une approche largement adoptée dans la mer Baltique consiste à améliorer la sélectivité des engins pour permettre aux jeunes poissons d'éviter d'être pris. Des modèles mixtes additifs généralisés ont été utilisés pour comprendre les effets des modifications de la sélectivité des engins et d'une taille minimum des poissons débarqués sur les taux de rejet, ainsi que les effets de divers autres facteurs explicatifs. Il a été démontré que les changements à la réglementation relative aux engins de pêche mis en application dans les pêcheries au chalut de fond danoises dans la mer Baltique et d'autres facteurs, tels qu'une taille de débarquement minimum, l'abondance des juvéniles, et le poids, le prix et la répartition spatiale et temporelle des prises, influencent significativement les taux de rejet. Les engins prescrits les plus récents sont ceux auxquels sont associés les taux de rejet les plus faibles. L'augmentation de la taille minimum des poissons débarqués de 35 à 38 cm s'est traduite par une augmentation des taux de rejet. [Traduit par la Rédaction]

Introduction

Discarding is a major problem in commercial fisheries around the world (Kelleher 2005). It refers to the practice of catching organisms of both commercial and noncommercial value only to return them to the sea, often dead or dying. It occurs for a variety of reasons and is influenced by an even wider range of factors (Catchpole et al. 2005; Rochet and Trenkel 2005; Feekings et al. 2012). The fundamental causes responsible for the high level of discarding are the use of unselective fishing techniques, the failure to reduce fishing effort, and biological and environmental factors affecting the distribution of species (Johnsen and Eliasen 2011).

In the Baltic Sea (ICES subdivisions 22–32), Atlantic cod (*Gadus morhua*) is economically the most important species, whereby the majority are caught by the trawl fishery (ICES 2012). Annual catches of cod have decreased substantially over the past 30 years, from a peak of around 400 000 tonnes (t) in 1984 to less than 65 000 t in 2008 (ICES 2012). However, in recent years the stocks have shown signs of improvement. The decline was caused by a combination of poor reproductive success and high fishing effort (Bagge et al. 1994; Köster et al. 2005; Heikinheimo 2008). Consequently, large efforts have been spent trying to improve the state of the Baltic Sea cod stocks.

A main management measure to reduce the capture of juvenile cod, and subsequently discards, has been through improvements in trawl selectivity. Over the past two decades various changes have been introduced to increase gear selectivity in the Baltic Sea cod trawl fishery (Madsen 2007; Suuronen and Sardà 2007; Wienbeck et al. 2011). As a further step in reducing discards, the European Commission is planning on progressively eliminating discards (e.g., through a discard ban, catch quotas, fully documented fisheries (e.g., closed-circuit television (CCTV) cameras)) in all European Union (EU) fisheries as part of the Common Fisheries Policy (CFP) reform to be introduced in 2014 (COM 2012). The EU contributes considerable funding to projects focusing on gear selectivity and discard reduction (Suuronen and Sardà 2007). Most trawl selectivity experiments are conducted with newly constructed fishing gears. The materials used in fishing gears are known to change over time, and the selective devices might be rigged and fished in other ways when used by commercial fishermen than during the original scientific experiments (Tschernij and Holst 1999; Madsen 2007; Suuronen et al. 2007). Consequently, when introduced into a fishery, the gear may not perform as expected. Therefore, assessing their effectiveness under commercial settings is necessary. By analysing the discard data, the nature of the commercial setting can be accounted for and the

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Fig. 1. Spatial distribution of sampling locations (eastern (Δ) and western (+) Baltic Sea) for the Danish discard sampling program for Danish demersal trawlers 1997–2010.
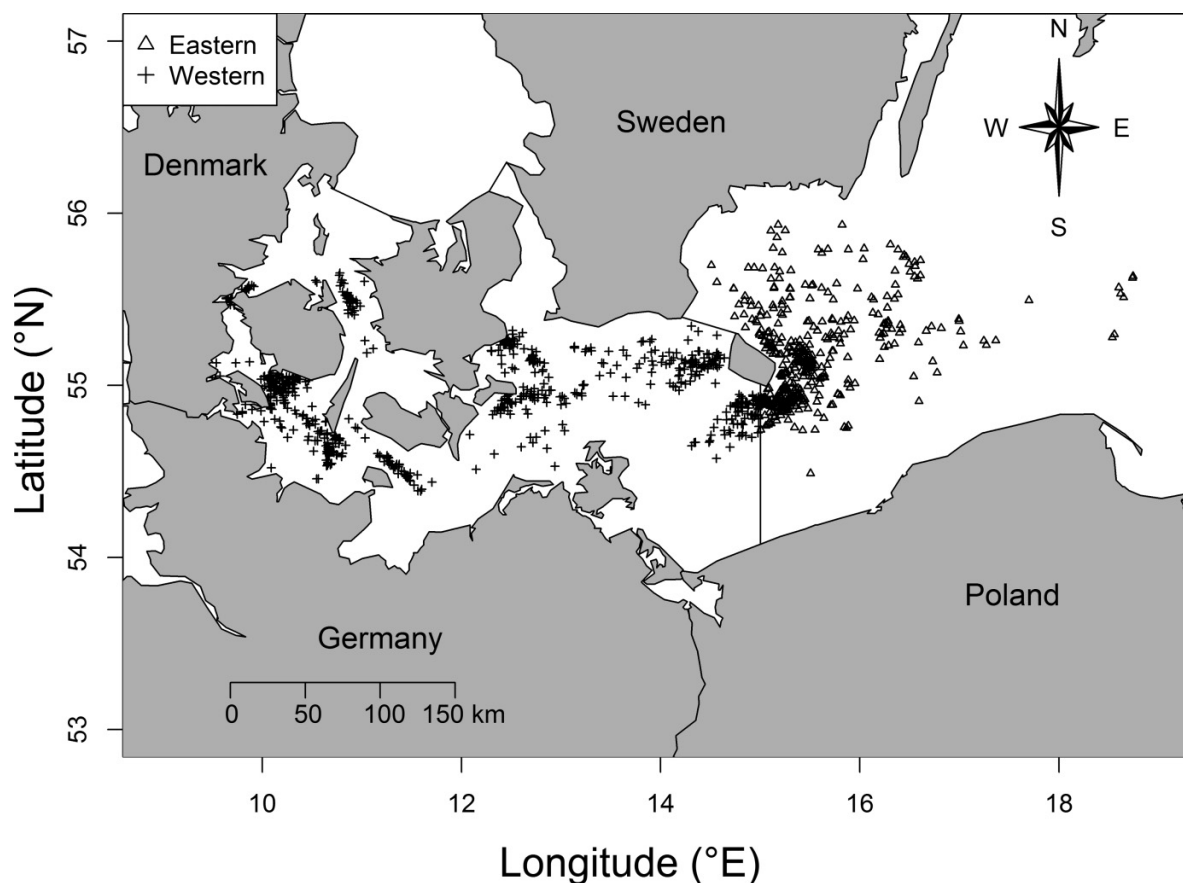
changes in discard rates due to improvements in selectivity can be assessed.

This study aims to understand (*i*) the effects that changes to gear selectivity and minimum landing size (MLS) have had on discard rates and (*ii*) account for a wide range of factors that can influence discard rates and may blur a potential effect of improved selectivity. Here we focus on the Danish Baltic Sea demersal trawl fishery (herein referred to as the demersal trawl fishery) during the period 1997–2010. Finally, we discuss how retrospective analyses of gear selectivity improvements together with factors influential to discards are important for fisheries managers in relation to discard mitigation measures.

Methods and materials

Baltic Sea cod and fishery

Cod in the Baltic Sea consist of a western and eastern stock (Fig. 1; Bagge et al. 1994). Stock assessments are carried out separately on the two stocks, and management regulations are also different in terms of seasonal spawning closures and total allowable catches. The demersal trawl fishery is considered a single species fishery; however, a reasonable bycatch of other species, such as European flounder (*Platichthys flesus*), European plaice (*Pleuronectes platessa*), and dab (*Limanda limanda*), are taken in the western Baltic. Therefore, the stocks have been analysed separately.

Since 1995, gear regulation changes relevant to the Danish fleet have resulted in a substantial increase in selectivity of cod (Fig. 2). The estimated effect from the changes in selectivity is a continuous increase in L50 (i.e., the length of an individual that has a 50% probability of being retained after entering the codend), giving a total increase of about 15 cm for the period investigated (Fig. 2). Until 1995, selectivity was regulated solely by increasing the mesh

size (Madsen 2007). In 1995, the first selective devices were introduced into the fishery (Madsen 2007). Known as the Danish window, the codend consisted of a 105 mm square-mesh window and was an alternative to a standard 120 mm diamond-mesh codend. An improved version of the Danish window, referred to as the New Danish window, was introduced in 1999. In the beginning of 2002, the Bacoma codend with a 120 mm square-mesh window was implemented in the demersal trawl fishery as an alternative to a standard 130 mm diamond-mesh codend (Madsen 2007; Suuronen et al. 2007). The increase in L50 (~10 cm; Fig. 2) was too high and caused large short-term economic losses (Tschernij et al. 2004). Subsequently, the Bacoma 120 mm codend was either replaced with the less selective alternative (standard 130 mm codend) or illegally manipulated to decrease its selectivity (Suuronen et al. 2007). To prompt fishermen to use selective codends, the MLS of cod was increased from 35 to 38 cm in 2003 (Fig. 2; Suuronen et al. 2007). Following the MLS increase, the standard 130 mm codend was prohibited and the mesh size of the Bacoma window was decreased to 110 mm. This mesh size better matched the new 38 cm MLS (Valentinsson and Tschernij 2003). As an alternative to the Bacoma 110 mm codend, the T90 110 mm codend (meshes turned 90 degrees) was introduced in 2006 (Suuronen et al. 2007). However, very few fishermen within the Danish fleet used it, and subsequently no hauls have been recorded using this gear in the discard data. In 2010, the minimum mesh size of the Bacoma window and T90 codends were increased to 120 mm (referred to herein as New Bacoma 120 mm and T90 120 mm codends, respectively). The length of the Bacoma window was also extended. This was to prevent selectivity from decreasing at high catch rates (ICES 2009; Madsen et al. 2010). Furthermore, regula-

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Fig. 2. Changes in codend selectivity (L50 and selection range, SR) for the regulated towed gears and changes in minimum landings size (MLS) throughout the study period 1997–2010. L50 indicates the length of an individual that has a 50% probability of being retained after entering the codend. An asterisk (*) indicates minimum mesh opening of 120 mm from 1 January 2010 in subdivisions 22–24 and from 1 March in subdivisions 25–32. The selectivity parameters for the T90 120 mm codend are taken from Wienbeck et al. (2011). All other selectivity values are taken from Madsen (2007). The selectivity estimates used for the Bacoma 120 mm and the New Bacoma 120 mm are unchanged, since the only difference is the window length, which is not expected to make any difference in relation to the used selectivity estimates obtained with relatively low catch rates.
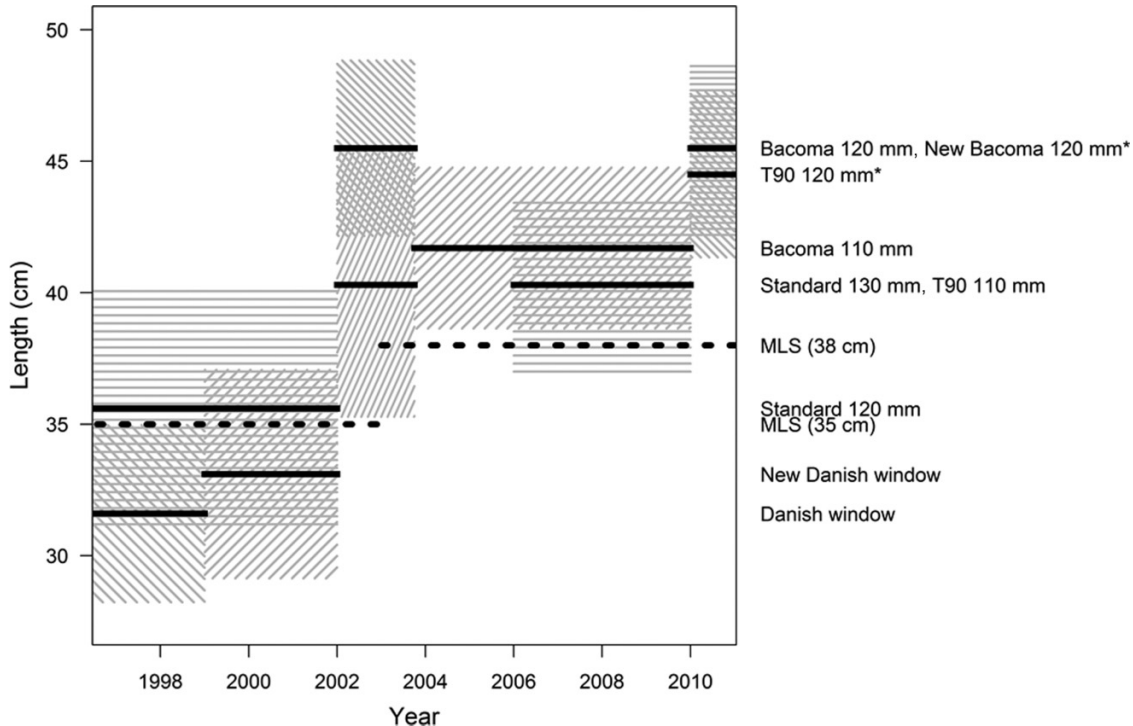
tions also concern the type of netting and its thickness (EU Regulation No. 2187/2005).

Discards can also be influenced by landings limitations. The Danish quota for cod more than halved over the study period, from approximately 50 000 t in 1997 to roughly 18 000 t in 2009. The quotas for cod in the eastern and western Baltic were restrictive only in 2004. Despite the quotas not being restrictive, landings are also regulated by fortnightly rations, which may have been restrictive on smaller time scales. Unfortunately this information is poorly documented. Subsequently, quota was not included as a factor in the analysis. Furthermore, discards of cod larger than the MLS only accounted for approximately 5% of the total discards (Table 1).

Discards

Denmark has been collecting information on discards as part of an at-sea scientific observer program since the mid-1990s, with the aim of sampling all species from all demersal fisheries except the ones with minimal fishing effort and discards (Feekings et al. 2012). Information on the sampling strategy and data collection methods has previously been described by Feekings et al. (2012). In total, 1121 discard observer hauls took place in the demersal trawl fishery over the period 1997–2010 (435 in the eastern Baltic (ICES areas 25–32) and 686 in the western Baltic (ICES areas 22–24); Fig. 1).

Owing to the complexity of the system, the practice of discarding is influenced by a large range of factors. The high number of influential variables stems from the interconnectedness of the social, economic, management, and environmental factors acting on the system (Feekings et al. 2012). Therefore, analysing the effects of changes in technical measures without accounting for such factors would neglect to account for the variability in dis-

cards that occurs as a result of this complexity. The forces acting on discards are also length-dependent, whereby discards under MLS are affected by a suite of factors that are different from those that are influential to discards over MLS. Consequently, we analyse discards under and over MLS separately. We identified a number of important explanatory variables that were considered in the analysis and their potential effects are described (Table 2).

To describe the temporal trends in cod discards, the proportion of discards to total catch (discards/(landings + discards)) in numbers per year (referred to herein as discard ratio), together with their standard deviations, were plotted for years 1997–2010. To illuminate the effect of an increase in MLS on discards, the expected discard ratios (if the MLS had remained at 35 cm) were plotted over time, as well as being accounted for as a factor in the model. The expected discard ratios were calculated by including those individuals between 35 and 37 cm that were discarded as part of the landings, therefore assuming that all individuals in this size range would not have been discarded had the MLS remained at 35 cm.

Statistical analysis

Generalized additive mixed models (GAMMs) were used to describe the relationship between the total numbers discarded per haul under and over MLS for the eastern and western Baltic cod stocks and a range of explanatory variables and to account for the unbalanced sampling design between explanatory variables. The numbers discarded per haul (herein referred to as discard rate) were assumed to follow a negative binomial distribution. Modelling the response variable as an integer rather than a density is important because it provides the possibility of including zero observations in the analysis and because these observations make up an important part of the total observations. The identity link was applied to describe the relationship between the mean value

Table 1. Summary of discard data collected on-board demersal trawls in the Baltic Sea for the period 1997–2010.

	S120	DW	NDW	S130	B120	B110	New B120	T90 120
No. of trips	13	80	100	31	85	302	25	30
No. of hauls	21	145	175	56	141	495	45	43
No. of vessels	6	30	36	20	42	63	15	10
Haul duration (h)	4.2 (1.1)	5.3 (3.0)	4.7 (2.1)	3.4 (1.5)	4.7 (2.0)	4.7 (2.3)	4.3 (1.9)	4.3 (1.2)
Discard length (cm)	30.0 (3.2)	29.4 (4.6)	29.8 (4.3)	31.1 (4.6)	30.9 (4.2)	33.9 (3.9)	33.0 (3.7)	33.2 (4.1)
Catch length (cm)	40.1 (11.0)	40.6 (10.0)	39.9 (7.5)	36.9 (9.5)	40.9 (8.5)	42.9 (8.0)	43.1 (7.6)	43.7 (9.7)
Discard proportion < MLS (%)	97.8	95.3	96.1	97.3	97.2	90.2	96.5	90.5
Discard ratio (numbers)	0.31 (0.33)	0.34 (0.32)	0.27 (0.29)	0.50 (0.27)	0.27 (0.26)	0.16 (0.15)	0.16 (0.17)	0.19 (0.19)

Note: Standard deviations are in parentheses. S120 = standard 120 mm diamond-mesh; DW = Danish window; NDW = New Danish window; S130 = standard 130 mm diamond–mesh; B120 = Bacoma 120 mm; B110 = Bacoma 110 mm; New B120 = New Bacoma 120 mm; T90 120 = T90 120 mm.

Table 2. Model components and their functions in describing discard rates.

Model components	Description
Codend	Effect of codend and its associated selectivity parameters on discards. The more selective the gear, the lower the discards.
Quarter	Capture the seasonal effect of discarding that occurs because of species' environmental preferences (e.g. bottom type, salinity).
MLS	Minimum landing size and its association with the selectivity of the gears used is a main determiner of discards. Increasing the MLS results in a larger number of individuals caught being under the MLS and subsequently discarded.
s(juvenile abundance)	Incorporates year- and quarter-specific differences in discards (i.e. capturing annual and seasonal variation in recruitment).
s(longitude, latitude)	Capture the spatial distribution of discarding that occurs because of species' environmental preferences (e.g. bottom type, salinity).
s(log catch mass)	The larger the catch, the larger the discards.
s(log haul duration)	The longer a haul duration, the larger the catch and discards.
s(vessel power)	Large vessels have the potential to tow larger nets, subsequently catching, and discarding, more fish.
s(price)	The mean weekly market price of cod in the size range directly above MLS (size class 5) for the Baltic Sea was used as an indicator of high-grading. This is the size class that obtains the lowest market price, and hence, the first to be high-graded. A lower market price has the potential to increase high-grading. The data were retrieved from the sales slips register.
Random(vessel)	Accounting for vessel differences in discard rates (e.g., the differences that occur as a result of the skipper effect, vessel-specific sorting behaviours, and vessel type).

of the dependent variable and 11 covariates. The following GAMM model was used:

$$E(N) = \alpha + \beta(\text{codend}) + \gamma(\text{quarter}) + \delta(\text{MLS}) + s(\text{juvenile abundance}) + s(\text{vessel power}) + s(\text{longitude, latitude})$$
$$+ s[\log (\text{catch mass})] + s[\log (\text{haul duration})] + s(\text{price}) + \text{random effect(vessel)}$$

where N denotes the number of cod discarded per haul, codend represents the eight gears described (Table 2), quarter is for the four quarters of a year, MLS is a factor equal to 0 for years 1997–2002 and 1 for years 2003–2010 for which MLS was, respectively, 35 and 38 cm, juvenile abundance is the number of age 1- and 2-year-old individuals, vessel power is the size of the vessel (kW), longitude and latitude show the location of the observations, catch mass the log-transformed mass of the catch (kg), haul duration is the log-transformed length of each observation (hours), price is the mean weekly market price of cod in the size range directly above MLS, vessel is the individual vessel effect, and s is an isotropic smoothing function, uniform in all orientations, used to define smooth terms (thin-plate regression spline; Wood 2003, 2006).

MLS and juvenile abundance should influence only discards less than MLS, while price should affect only discards greater than MLS. Therefore, these variables were not included in all models. The variable juvenile abundance was calculated per quarter by applying a simple exponential decay function based on the relative number of individuals of age 1 and 2 caught during Baltic Sea International Trawl Surveys undertaken in the first quarter of each year (Feekings et al. 2012). Natural mortality (M) and fishing mortality (F) were taken from the original assessment (ICES 2012),

where we assume that M is constant during the year and F increases linearly during the year (this is to account for the growth, and subsequent increase in retention by the fishing gear, of an individual throughout the year).

The discarding behaviour of individual vessels may differ because of differences among vessel type, skipper effect, or vessel-specific sorting behaviours (Hilborn and Ledbetter 1985; Tschernij and Holst 1999; Poos and Rijnsdorp 2007). To account for such differences, vessel was used as a random effect variable. Here we assume that the variation around a vessel's mean is normally distributed around zero with an unknown variance (to be estimated). The analyses were performed using the R (R Development Core Team 2011) package "mgcv" (Wood 2011).

All covariates were included in the initial model. The least significant covariates (the one with the largest P value) were removed one at a time until all covariates were significant ($P < 0.05$; Zuur et al. 2009). Categorical terms (codend and quarter) that were nonsignificant were combined and the models refitted. Combined categorical terms are referred to as newcodend and halfyear in the model. The final models are then reduced versions of these full models.

The final model fit was tested using the randomized negative binomial test developed by Kristensen et al. (2006). The test was

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Fig. 3. Discard ratio, in numbers, of cod per year, showing mean (black line) and standard deviation (shaded area). The lower line represents the discard ratio assuming the MLS had remained at 35 cm for years 2003–2010.
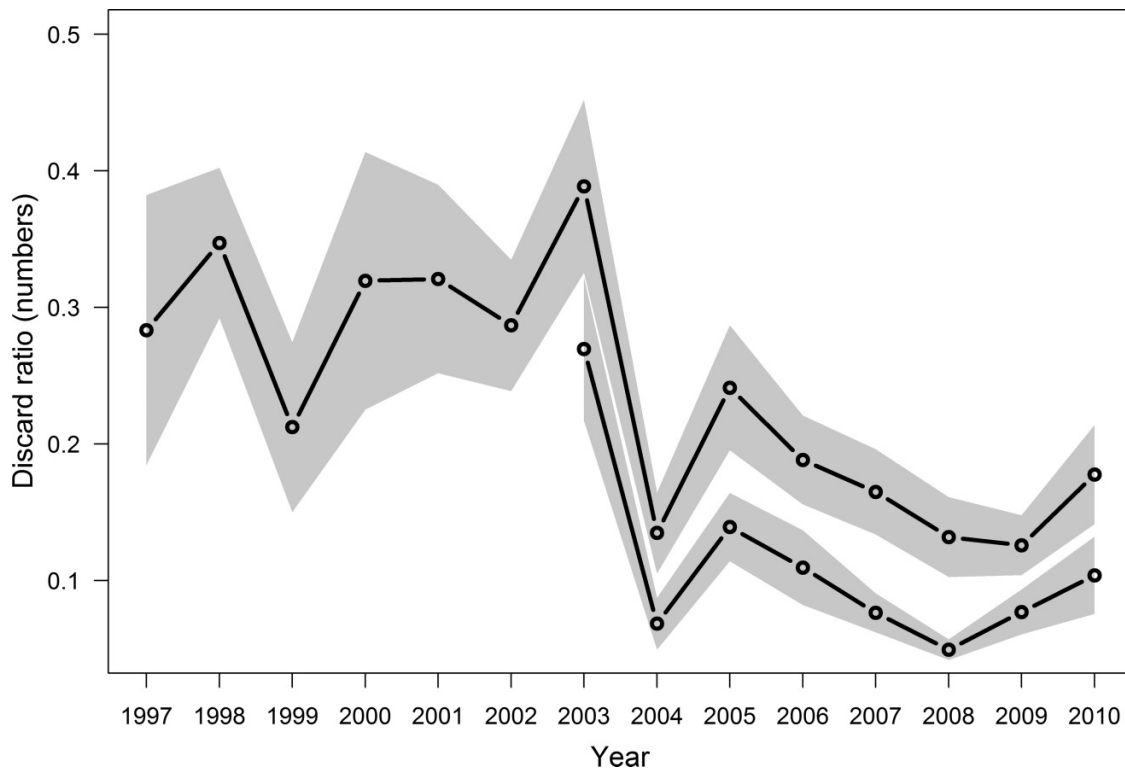
modified to account for distributions with different mean values with the same size parameter. The test generates the variable U, which should follow a uniform distribution if the model assumption holds. Instead of considering the variable U, we transform U by the quantile function of the normal distribution, $qnorm(U) = Z$. If the model assumption holds, Z is normally distributed. Hence, the model validation was checked by testing if Z is normally distributed, which was performed graphically by a QQ plot and quantitatively by the Kolmogorov–Smirnov test.

Results

The overall discard ratio of cod was 23.5% (±0.007 SE), but the ratio varied substantially by gear and subsequently year (Fig. 3; Table 1). Approximately 95% of cod discarded were under MLS. The newest gears (Bacoma 110 mm, New Bacoma 120 mm, and T90 120 mm) had the lowest discard ratios as well as the highest mean discards and catch lengths. The discard ratio of cod in the demersal trawl fishery fluctuated throughout the study period (1997–2010) while exhibiting a general overall decline (Fig. 3). Also, the expected discard ratio, if the MLS had remained at 35 cm, was substantially lower than the observed discard ratio (Fig. 3).

Effect of technical measures on discards

Since the eastern and western Baltic were analysed separately, as were the portions of discards under and over MLS, we end up with four final models (Table 3; Table S1[1]). For both the eastern and western cod stocks, the newest and currently legislated gears (New Bacoma 120 mm and T90 120 mm codends) had the lowest discard rates of individuals under MLS (Table 3). The highest discard rates of cod less than MLS for the eastern stock were observed for the initial Bacoma 120 mm and the standard 130 mm diamond-mesh codends. For the western stock, the standard 130 mm

diamond-mesh codend again had the highest discard rate of cod less than MLS. The discard rate of the first Bacoma codend (Bacoma 120 mm) was significantly higher than the Bacoma 110 mm and the New Bacoma 120 mm codends. Cod discards less than MLS in the eastern Baltic Sea were also found to be significantly affected by the increase in MLS such that the discard rate, all things (factors) being equal, was less before the MLS was increased from 35 to 38 cm in 2003. No significant effect of the increase in MLS was detected for the western stock. No significant differences in the discard rates of cod greater than MLS in the eastern Baltic were observed for the different codends, except for the standard 130 mm codend, which had a significantly lower discard rate than all other codends (Table 3). In the western Baltic, the discard rates of cod greater than MLS were significantly higher for the older codends in relation to the Bacoma and T90 codends.

Additional factors influential to discards

The seasonal effect on discards of cod was significant in three of the final models (Table 3). Discards of cod over and under MLS in the eastern Baltic were significantly higher in quarter 2 than all other quarters. In the western Baltic, no significant difference among quarters was found for discards under MLS, while discards of cod over MLS were significantly lower in quarter 4 than all other quarters. In both regions, the interaction between longitude and latitude was significant for discards of cod less than MLS (Table 3; Fig. 4). In the eastern Baltic, discards under MLS were lowest in the Bornholm Basin and increased in a northeasterly direction. The spatial distribution of discards under MLS in the western Baltic was rather uniform throughout. The only tendency observed was a negative trend between the Danish islands of Zealand and Funen. The effect of juvenile abundance was significant only for the western Baltic where a positive relation-

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Table 3. Final model results.

Eastern stock					Western stock				
<MLS									
Categorical term	**Estimate**	**SE**	**z value**	**P value**	**Categorical term**	**Estimate**	**SE**	**z value**	**P value**
Intercept	5.02	0.15	33.43	<0.001	Intercept	3.64	0.11	33.80	<0.001
Bacoma 120 mm	0.80	0.14	5.86	<0.001	Bacoma 120 mm	0.36	0.16	2.24	0.025
New Bacoma 120 mm	−0.49	0.21	−2.31	0.02	Danish window	0.92	0.19	4.98	<0.001
Standard 130 mm	0.72	0.31	2.32	0.02	New Danish window	0.68	0.17	3.90	<0.001
T90 120 mm	−0.46	0.16	−2.88	0.0039	Standard 120 mm	0.73	0.37	1.99	0.047
Quarter 2	0.11	0.11	3.03	0.0024	Standard 130 mm	1.70	0.20	8.34	<0.001
MLS 35 cm	−0.68	0.14	−4.87	<0.001					
Smooth term	**edf**		χ^2	**P value**	**Smooth term**	**edf**		χ^2	**P value**
s(long., lat.)	20.25		59.64	<0.001	s(long., lat.)	21.34		166.72	<0.001
s(log catch mass)	2.85		280.25	<0.001	s(log catch mass)	2.86		246.98	<0.001
s(log haul duration)	5.60		51.73	<0.001	s(log juv abund)	1.00		12.33	<0.001
Random term	**SD**		χ^2	**P value**	**Random term**	**SD**		χ^2	**P value**
s(vessel)	2.52		131.31	<0.001	s(vessel)	1.42		101.47	0.00212
Eastern stock					Western stock				
≥MLS									
Categorical term	**Estimate**	**SE**	**z value**	**P value**	**Categorical term**	**Estimate**	**SE**	**z value**	**P value**
Intercept	1.50	0.22	6.87	<0.001	Intercept	−0.095	0.19	−0.50	0.62
Standard 130 mm	−1.48	0.67	−2.20	0.028	Danish window	1.29	0.30	4.25	<0.001
Quarter 2	0.45	0.21	2.17	0.03	New Danish window	1.21	0.30	3.96	<0.001
					Standard 120 mm	1.49	0.59	2.52	0.01
					Standard 130 mm	2.24	0.32	6.98	<0.001
					Quarter 4	−0.76	0.22	−3.45	<0.001
Smooth term	**edf**		χ^2	**P value**	**Smooth term**	**edf**		χ^2	**P value**
s(log catch mass)	7.12		91.26	<0.001	s(price)	2.99		11.46	0.0095
					s(log catch mass)	1.00		126.13	<0.001
					s(log haul duration)	1.00		8.84	0.003
Random term	**SD**		χ^2	**P value**	**Random term**	**SD**		χ^2	**P value**
s(vessel)	3.71		167.16	<0.001	s(vessel)	3.46		230.94	<0.001

Note: "edf" are estimated degrees of freedom.

ship was detected. The mean weekly market price of size sorting class 5 had a weak but significant effect on discard rates of cod greater than MLS in the western Baltic (Table 3; Fig. 5). No significant effect of price was detected in the eastern Baltic. Log catch mass was found to be positively related to the discard rates of cod in all four final models (i.e., the larger the catch the higher the level of discarding).

Finally, the model validations of the four final models are demonstrated in the QQ plots in Fig. S1[1] and indicate good model fits. Further, the Kolmogorov–Smirnov tests of the four models resulted in respective P values of 0.38, 0.14, 0.56, and 0.36, indicating that the final models were not rejected.

Discussion

Increasing codend selectivity has been a major management measure in the Danish demersal cod trawl fishery. It was one of the first regions within the European Community where selective devices were adopted into legislation (Madsen 2007). Subsequently, an evaluation of the potential effects is highly important for future management strategies, also in relation to other fishing areas. Particularly because earlier findings suggest that there was no marked effect (ICES 2005; Suuronen and Sardà 2007; Suuronen et al. 2007). Our results confirm those of previous findings. No significant reduction in discard rates during the first part of the period investigated was observed. This could be caused by several factors: (*i*) noncompliance because of economic losses, as reported by Suuronen et al. (2007); (*ii*) the improvements were too small to be detected by the models with the variability in the available data; (*iii*) the increase in selectivity expected from scientific experiments is not present under commercial settings because the gears are rigged and fished differently. For the latest gears intro-

duced in the period investigated (Bacoma 110 mm codend in the western Baltic, the New Bacoma 120 mm, and T90 120 mm), a significant effect on discard rates was observed, suggesting a significant overall reduction in discard rates for this period, although several other factors have been influential.

The increase in MLS resulted in a significant increase in the rate of cod under MLS being discarded in the eastern Baltic, while no significant effect was observed in the western Baltic. Both influential and noninfluential effects of changes to MLS have been reported for a range of species (Stratoudakis et al. 1998; Borges et al. 2006). After the increase in MLS, approximately 40% of cod caught were discarded. Similar discard ratios were found by Suuronen et al. (2007) for the Swedish fleet. One of the main objectives of increasing the MLS was to encourage fishermen to use more selective gears (Suuronen et al. 2007). Instead, increasing the MLS resulted in an increase in discards. The expected discard ratios if the MLS had remained at 35 cm were substantially lower than the observed discard ratios. Therefore, lowering or even removing the MLS is warranted if discards are sought to be instantaneously reduced. This would result in the increased retention of smaller individuals on-board. From an ecological perspective, this is of little concern as a large majority of the individuals are already dead (Thurow and Bohl 1976; Evans et al. 1994). Their initial capture is of most concern. Hence, further increasing selectivity may be a better alternative. Despite discard ratios of cod in the Baltic Sea being lower than the neighbouring regions during recent years (e.g., Kattegat (30%–50% in mass; ICES 2011) and the North Sea (50%–75% in numbers; ICES 2011)), individuals are still caught at a relatively young age. Therefore, improving selectivity further will give great long-term benefits (Madsen 2007). However, this
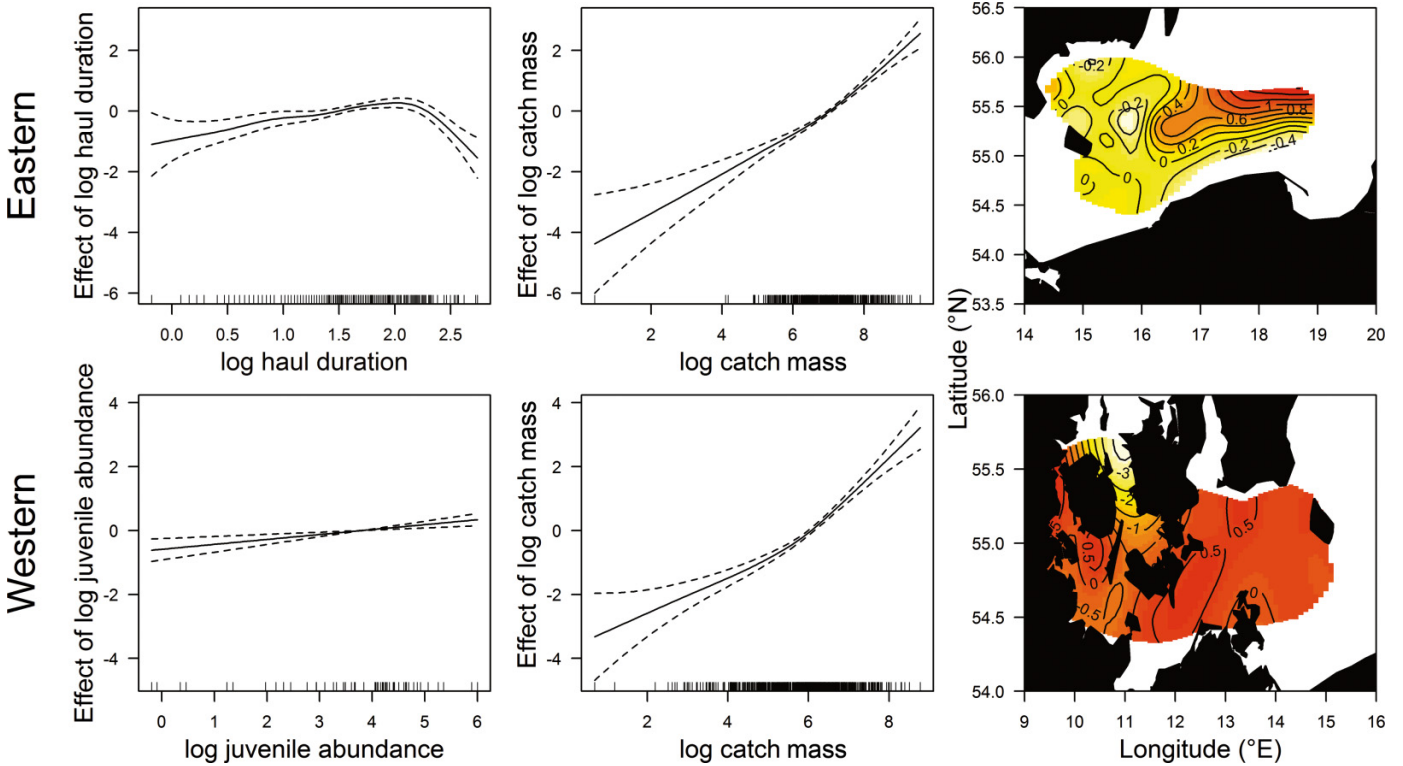
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Fig. 4. Model-predicted effects of significant smoothing functions (solid lines) on the discard rates of cod greater than MLS in the eastern (top row) and western (bottom row) Baltic Sea 1997–2010. Dotted lines represent the 95% confidence limits. Vertical bars along the *x* axis indicate observational values. The surface and contour lines describe the effect of the two-dimensional smoothing function on the geographical coordinates. Red area indicates highest discard rate. Only areas covered by observations are indicated.

Fig. 5. Model-predicted effects of significant smoothing functions (solid lines) on the discard rates of cod less than MLS in the eastern (top row) and western (bottom row) Baltic Sea 1997–2010. Dotted lines represent the 95% confidence limits. Vertical bars along the *x* axis indicate observational values.
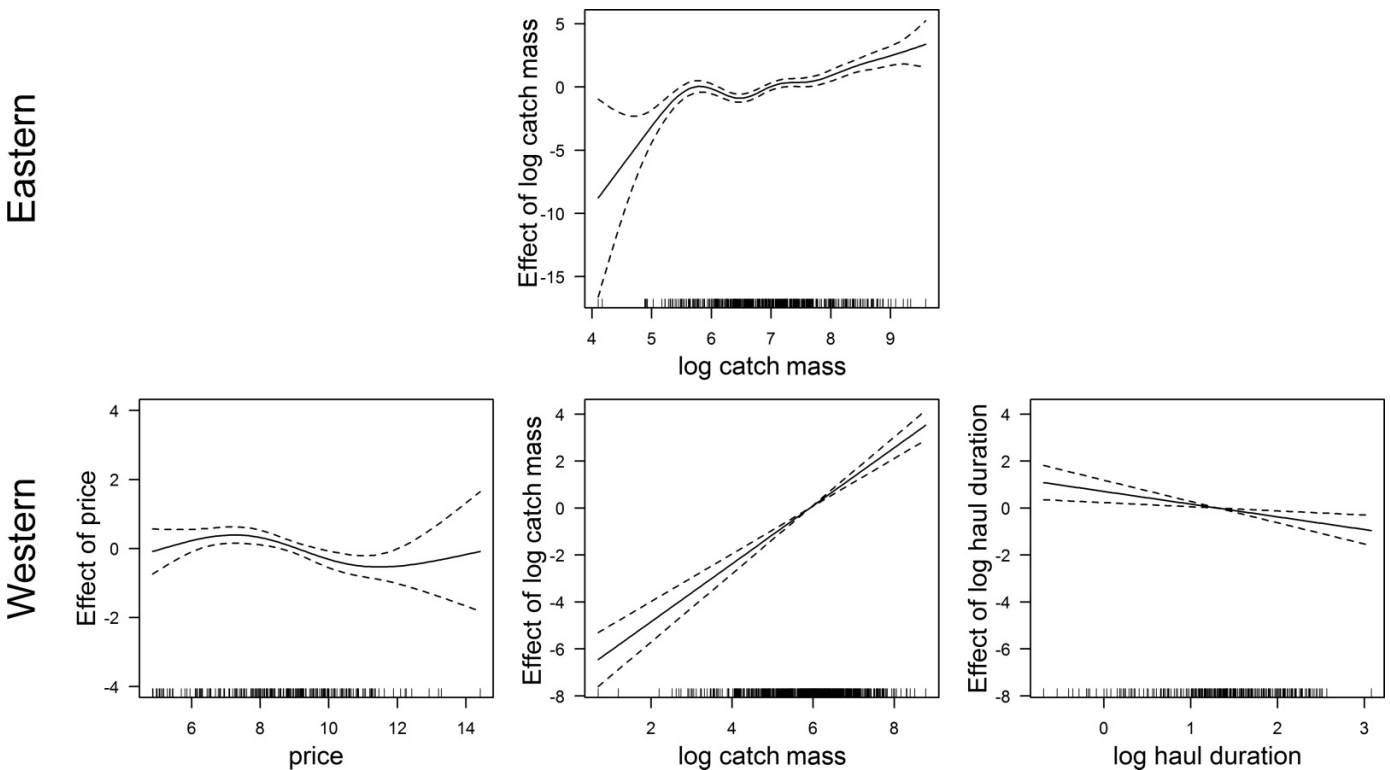
would more than likely result in short-term economic losses,which has been observed in the past (Suuronen et al. 2007).

Significant spatial and temporal variations in discard rates were observed in both the eastern and western Baltic. Spatial and seasonal variations in discards have also been observed in other areas for a variety of commercial species (Rochet and Trenkel 2005; Viana et al. 2011; Feekings et al. 2012). The variation across seasons has been attributed to the targeting behaviour of the fishermen and the condition or behaviour of species during different seasons (Viana et al. 2011; Feekings et al. 2012). Incorporating the spatio-temporal distribution of Baltic cod into management offers the possibility of reducing the quantity of unintended catch and therefore the quantity discarded. It also provides the possibility to introduce economic incentives for fishermen to adopt selective fishing techniques that allow them conditional access to otherwise closed areas (Catchpole et al. 2005).

Here we also aimed to relate the interannual variability in discards of cod larger than MLS to market price. However, the price estimates used in the models had no market effect (except for a slight significant negative correlation in the western Baltic). Previous studies that have reported on the influences of market price on discards found contrasting effects for a range of species (Rochet and Trenkel 2005). Furthermore, Graham et al. (2007) found the effect of market price on discards to be species-dependent. The lack of an effect that was observed may be a result of the low number of discards over MLS or because the price estimates used as inputs to the GAMMs were weekly averages and therefore did not account for price variations across ports. Price differences across ports may result in a vessel travelling to another port to obtain a higher price rather than discard part of the catch over MLS. Knowledge about a vessel's port of origin and port of landing could help to better understand the effect of price on over MLS discarding. Discards greater than MLS may also result from limitations of capacity, processing, or quota (Cook 2003). For mixed fisheries where species quotas are restrictive, quotas can instil a considerable amount of discarding, especially when fishermen are unable to avoid catching fish for which they have no quota while targeting other species for which quota still exists. Hence, these factors, when available, should be considered to provide a more thorough picture of greater than MLS discarding.

Here we have demonstrated that there are many factors that contribute to discards and the effectiveness of gear-based technical measures. If catch losses are great, fishermen will attempt to improve their economic situation. Therefore, the successful implementation of gear improvements, as indicated by this study, depends on overcoming the short-term economic losses associated with their use (Cook 2003) and the fishermen's acceptance (Tschernij et al. 2004; Suuronen et al. 2007; Madsen et al. 2010). When short-term catch losses are too large to be absorbed, gears may be manipulated and rules circumvented such that potential long-term gains may never materialize (Suuronen and Sardà 2007; Suuronen et al. 2007). Nielsen and Mathiesen (2003) found that economic incentives were the driving force behind noncompliance within Danish fisheries. Thus, to provide more effective implementation of management strategies, these should be implemented progressively, when the stock is showing signs of recovery, and incorporate industry and fishermen in the decision-making process (Madsen 2007; Suuronen et al. 2007). Management strategies should also create incentives for fishermen to alter their behaviour, so that the entire sector can benefit both environmentally and economically from the use of fishing methods that reduce discards (Suuronen and Sardà 2007).

Mortality of cod after escape through trawl codends is low (Suuronen et al. 1996, 2005); however, mortality of cod discarded in trawl fisheries is high (Thurow and Bohl 1976; Evans et al. 1994). This results in a direct loss to the fishery and to the recruitment of the stock (Diamond and Beukers-Stewart 2011). Discards also contribute to total fishing mortality. Despite this, discards are seldom

counted against species quotas, which in the EU are landings quotas and not catch quotas. With the reform of the CFP, a catch quota management system has been proposed together with a discard ban. Such a management system, if implemented correctly, has the potential to create economic incentives for the industry to reduce the capture of smaller individuals through improvements in gear selectivity and the spatio-temporal distribution of the fishery (Feekings et al. 2012). It would also provide the possibility for industry-driven selectivity improvements to not catch individuals less than the MLS that would subsequently count against their quota. With the state of the eastern Baltic cod stock being the strongest in 20 years (Eero et al. 2012), such a management system appears to be a step in the right direction. Such changes to management would be relatively simple for the Danish Baltic Sea demersal trawl fishery because of its almost single species nature and the fact that discards are on the decline. However, such changes to more complex fisheries, such as the mixed demersal trawl fishery in the North Sea, would prove more problematic because of the diversity of species, sizes, and morphological characteristics that are caught within this fishery.

The low sampling levels that are often observed in at-sea scientific observer programs together with the high variability in discard rates that occurs can lead to uncertainties when identifying the driving forces behind discards. However, with the plan to introduce a catch quota management system and full documentation of catches (e.g., CCTV cameras) as part of the CFP reform, the possibility to analyse total catches at a fleet or area level may become possible.

Overall, our results show that when gear regulations are implemented correctly, they are an effective management measure. Their effectiveness is influenced by a diverse range of factors that if unaccounted for may distort a potential effect of improved or hampered selectivity. Therefore, for successful implementation, the effects of these additional influential factors should be accounted for.

Acknowledgements

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Supplementary material

Table S1. Model selection results. Full model selection is show for Eastern Baltic discards less than MLS. Initial and final models only are shown for Western Baltic discards less than MLS, Eastern Baltic discards greater than MLS, and Western Baltic discards greater than MLS.

| Model	Predictors	AIC	DEV.EXPL(%)
Eastern stock < MLS			
Initial	$\gamma \approx \alpha$+ s(log catch weight)+s(longitude, latitude)+ s(log haul duration) + MLS + β codend + s(log Juvenile Abundance) + β quarter + s(log vessel power) + random(vessel)	5127.3	65.0
Final	$\gamma \approx \alpha$+ s(log catch weight)+s(longitude, latitude)+ s(log haul duration) + MLS + β new codend + β halfyear + random(vessel)	5127.3	63.3
Western stock < MLS			
Initial	$\gamma \approx \alpha$+ s(log catch weight)+s(longitude, latitude) + s(log Juvenile Abundance) + β codend + MLS + β quarter + s(log haul duration) + s(log vessel power) + random(vessel)	7106.5	55.2
Final	$\gamma \approx \alpha$+ s(log catch weight)+s(longitude, latitude) + s(log Juvenile Abundance) + β new codend + random(vessel)	7104.4	53.9
Eastern stock ≥ MLS			
Initial	$\gamma \approx \alpha$+ s(log catch weight)+ β codend + β quarter + s(price) + s(log vessel power) + s(longitude, latitude) + s(log haul duration) + random(vessel)	2384.5	64.9
Final	$\gamma \approx \alpha$+ s(log catch weight)+ β new codend + β half year + random(vessel)	2385.2	58.7
Western stock ≥ MLS			
Initial	$\gamma \approx \alpha$+ s(price) + s(log catch weight) + s(log haul duration) + β codend + β quarter + s(longitude, latitude) + s(log vessel power) + random(vessel)	2510.4	60.9
Final	$\gamma \approx \alpha$+ s(price) + s(log catch weight) + s(log haul duration) + β new codend + β half year + random(vessel)	2511.2	59.0
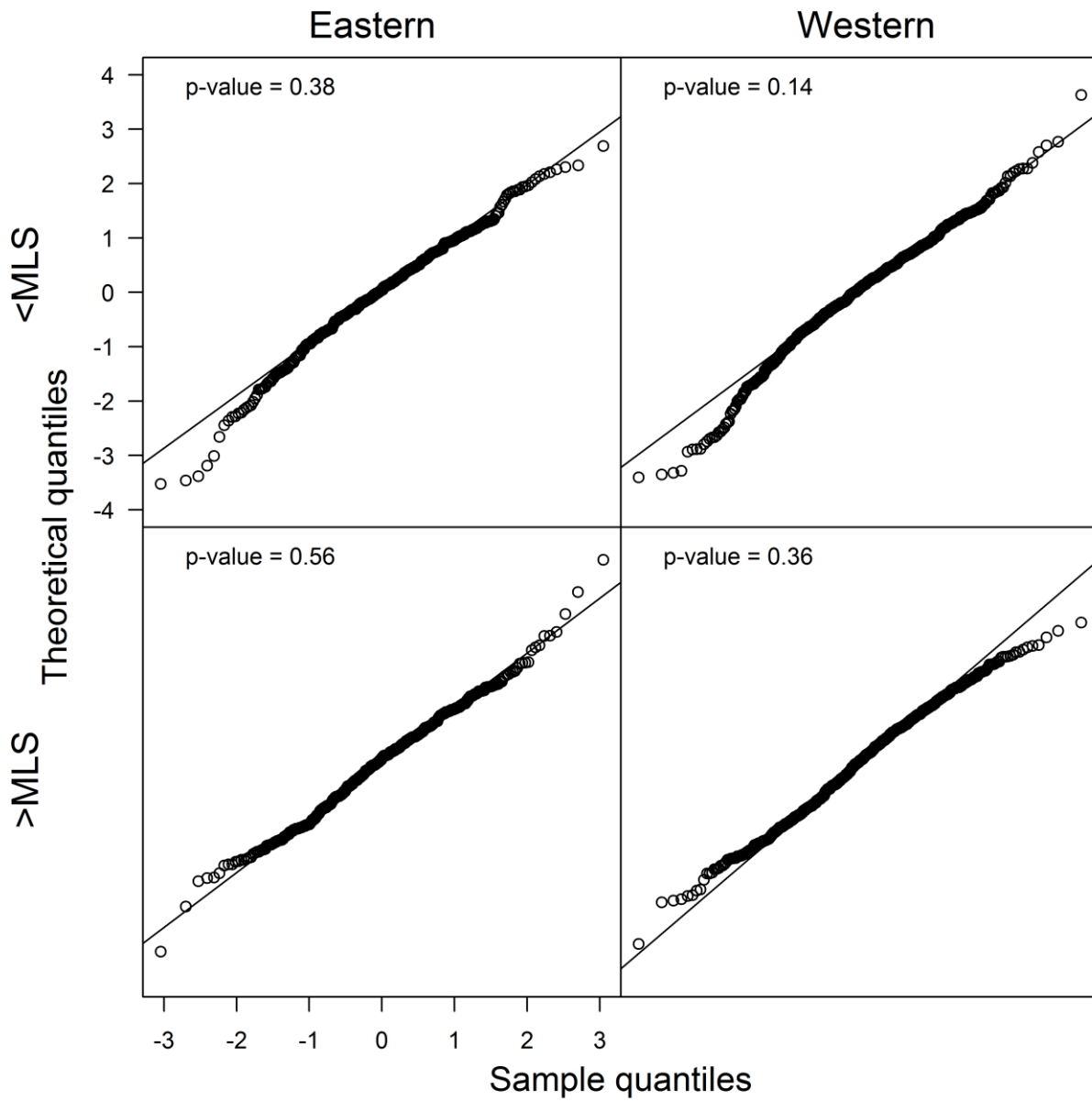
Figure S1. Model validation qq-plots. *P*-values represent results from the Kolmogorov-Smirnov tests.

Paper III

Spatio-temporal variability of North Sea cod discards

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Summary paragraph

During commercial fishing operations parts of the catch are returned to the sea either dead or dying as a response by fishers to either market forces or regulatory restrictions (Kelleher 2005). This practice, referred to as discarding, is currently one of the most important issues within commercial fishing (Hall et al. 2000; Hall and Mainprize 2005), and has prompted a commitment from the European Council to end discarding through a phased ban (Anon 2007). The spatio-temporal heterogeneity in species distributions is often reflected in the highly variable patterns of discarding in space and time (Liggins and Kennelly 1996; Machias et al. 2001; Murawski 1996; Stobutzki et al. 2001; Catchpole et al. 2005). Knowledge on the spatio-temporal pattern of discards is imperative to researchers and regulators (Dunn et al. 2011) but is often lacking (Viana et al. 2011). Here we show discard rates of small and large cod (*Gadus morhua* L.) in the North Sea to be highly variable in their spatial and temporal (interannual and seasonal) distributions. Our analyses revealed that depth, time, location, gear type and mesh size, as well as individual vessel characteristics, to be correlated with discard rates of cod. We anticipate our results will be a starting point for the consideration of the spatio-temporal distribution of discards into fisheries management. We discuss how such information can be used to improve future fishing activities and their subsequent catch compositions under a discard ban.

Introduction, Results and Discussion

Many of the world's commercial fisheries are characterised by overfishing, illegal landings, and high discarding. While many of these fisheries are beginning to rebuild (Worm et al. 2009) high discarding remains an issue (Myers et al. 1997; Davies et al. 2009), increasingly so with the move to reduce the environmental impact of fishing on the ecosystem (Pikitch et al. 2004; Kempf 2010). Discards are a waste of resources and incur increased fishery costs (Pascoe 1997). Also, the practice of discarding threatens endangered species, damages habitats, impacts the food web and affects ecosystem function and biodiversity (Alverson et al. 1994; Votier et al. 2004; Anon 2007; Votier et al. 2010; Zhou 2008). Consequently, fisheries discards have received considerable attention from policy makers, NGO's and more recently the media (www.fishfight.net) as a result of the European Commission (EC) revising its approach to manage the problem (Anon 2007).

During 1992-2001, the average yearly level of discards in the world's marine fisheries was estimated to be 7.3 million tonnes (Kelleher 2005). Throughout this period, the North Sea (ICES subarea IV) accounted for the highest level of discarding in the world, some 13% (909,109 t) of global discards (Kelleher 2005). Of this total, 60–70% of discarded resources are roundfish and flatfish species arising mostly from the European roundfish, flatfish and Norway lobster (*Nephrops norvegicus* L.) demersal trawl fisheries (Catchpole et al. 2005).

The cod stock in the North Sea has been below sustainable levels since the late 1960s and great concern has been expressed about the decline in cod biomass and recruitment (Cook et al. 1997; Beaugrand et al. 2003). Cod discards relative to catch were the highest on record in 2008, accounting for approximately 75 % in numbers (50 % in weight) (ICES 2012). However, the most recent advice has shown signs of improvement. While still above the historical average, discards are now approximately 55% in numbers (25% in weight) (ICES 2012). The declining cod stock within the North Sea has resulted in management actions such as changes in technical measures (for example, mesh size increases), fishing effort restrictions, closed areas and the establishment of a cod recovery programme; all of which have attempted to reduce the amount discarded. However, these measures have not been supported by detailed knowledge of the spatio-temporal distribution of discards. To elucidate the discarding patterns of cod in the North Sea we employ generalized additive mixed models (GAMMs) using discard data from 11370 fishing events collected throughout the period 2003 – 2010. Data were collected across seven European Union (EU) Member States as part of the EU Data Collection Framework (DCF). Furthermore, we aim to account for the variability in discards that occurs as a result of depth (Moranta et al. 2000; Sánchez et al. 2004; Feekings et al. 2012), gear and its associated mesh size (Murawski 1996; Enever 2009), and vessel specific characteristics (Machias et al., 2004; Rochet and Trenkel, 2005; Feekings et al. 2012). Discards

are a consequence of different drivers - in the case of cod which has an associated minimum landing size (MLS) and quota, fish are either too small to be legally landed or they are discarded as a response to quota restrictions. Analyses revealed highly significant spatio-temporal heterogeneity among small (<MLS) and large (≥MLS) cod throughout the North Sea on both interannual and season time scales (Table 1; $P < 0.001$). The spatial patterns indicate discarding of small cod to be highest along the north-eastern coast of the United Kingdom (UK) and throughout the north-eastern and north-central North Sea (Fig. 1). The region along the north-eastern coast of the UK coincides with the spatial distribution of the Norway lobster fishery that is part of the trawl and seine 70-99 mm fishery (Supplementary Fig. 1) and is a consequence of the relatively small mesh sizes required to catch Norway lobster (Frandsen et al. 2011).The spatial distribution of small cod discards throughout the north-eastern and north-central North Sea corresponds with the roundfish fishery, the main fishery for cod. These areas of high discard rates for small cod have previously been defined as suitable habitats for age-1 and age-2 cod (Blanchard et al. 2005). The spatial distribution of large cod discards also coincides with the main roundfish fishery and are highest throughout the north-eastern North Sea, however, large cod discards extend further north than discards of small cod (Fig. 2).

Table 1. The final model results of fitting Generalized Additive Mixed Models to the numbers of cod discarded in the North Sea less than and greater than the minimum landing size.

Small cod (<MLS)					Large cod (≥MLS)				
Categorical terms	Estimate	SE	*Z*-value	*P*-value	Categorical terms	Estimate	SE	*Z*-value	*P*-value
Intercept	4.20	0.32	-13.08	< 0.001	Intercept	-7.62	0.47	-16.23	< 0.001
Beam Trawl 80 – 119 mm	-0.04	0.19	-0.22	0.824	Beam Trawl 80 – 119 mm	1.12	0.24	4.69	< 0.001
Trawl & Seine ≥100 mm	-0.38	0.28	-1.33	0.18	Trawl & Seine ≥100 mm	1.96	0.35	5.61	< 0.001
Trawl & Seine 70 – 99 mm	-0.07	0.29	-0.23	0.82	Trawl & Seine 70 – 99 mm	0.85	0.35	2.41	0.02
GOV Trawl	1.53	1.21	1.26	0.21					
Smooth terms	edf	Ref.df	Chi.sq	*P*-value	Smooth terms	edf	Ref.df	Chi.sq	*P*-value
te(longitude, latitude)	19.43	20.74	507.54	< 0.001	te(longitude, latitude)	23.85	23.99	321.91	< 0.001
te(log depth)	2.92	2.99	69.16	< 0.001	te(log depth)	2.93	3.00	34.57	< 0.001
te(month)	1.98	2.00	102.40	< 0.001	te(month)	1.56	1.90	6.59	0.03
te(year)	3.97	4.00	198.61	< 0.001	te(year)	3.86	3.99	171.36	< 0.001
Random term	Std.Dev		Chi.sq	*P*-value	Random term	Std.Dev		Chi.sq	*P*-value
s(vessel)	29.92		15528.53	< 0.001	s(vessel)	46.19		11092.34	< 0.001
Deviance explained	40.1%				Deviance explained	58.7%			
Theta	0.23				Theta	0.25			

Footnote: Z-values are the parameter estimates divided by their estimated standard deviations. The deviance estimated in the models, analogous to the residual sum of squares, is a measure of the fit of the model and can be considered a pseudo R-squared (Swartzmann et al. 1992).
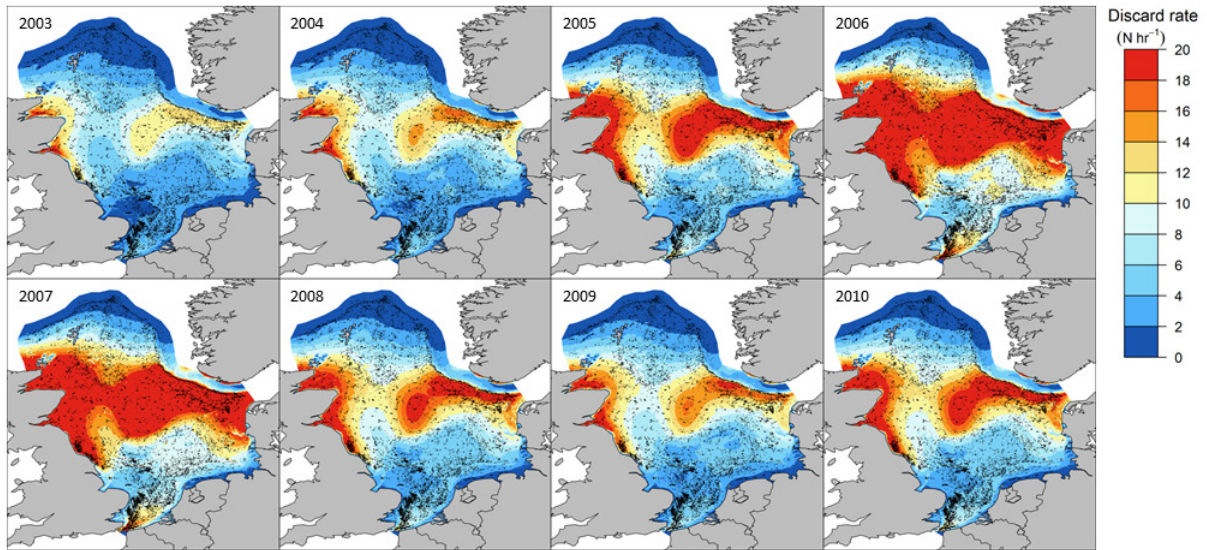
Fig. 1. Model predicted densities of small cod discards in North Sea 2003 – 2010.
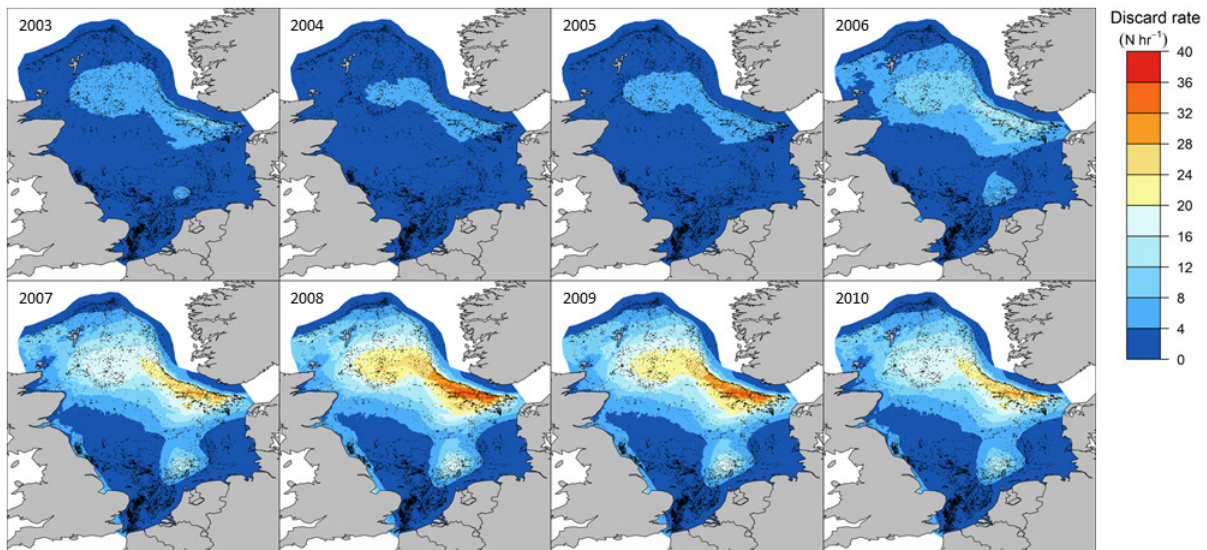

Fig. 2. Model predicted densities of large cod discards in North Sea 2003 – 2010.

The interannual variability for both small and large cod discards was highly significant ($P < 0.001$). Discard rates of small cod fluctuated throughout the study period, with the highest levels of discarding occurring in 2006 and 2007 (Fig. 1 & Fig. 3) coinciding with a year of relatively high recruitment of new young cod entering the population in 2006 (ICES 2012). The rate of large cod being discarded increased between 2005 and 2007 (Fig. 2 & Fig. 3). The main problem is that once a vessels quota is met, fishers

are legally obliged to discard large fish; this is exacerbated when fishers are unable to avoid catching fish for which they have no quota whilst targeting other species and the geographic distribution of the quota is not the same as that of the fish. The increase in large cod discards can therefore be the result of highly restrictive cod quotas (Horwood et al. 2006) and between in 2005 and 2007 the total allowable catch for North Sea cod decreased by 25% (ICES 2012). This discarding of large cod accounts for approximately 50 % of discards within some fisheries (Supplementary Table 1).
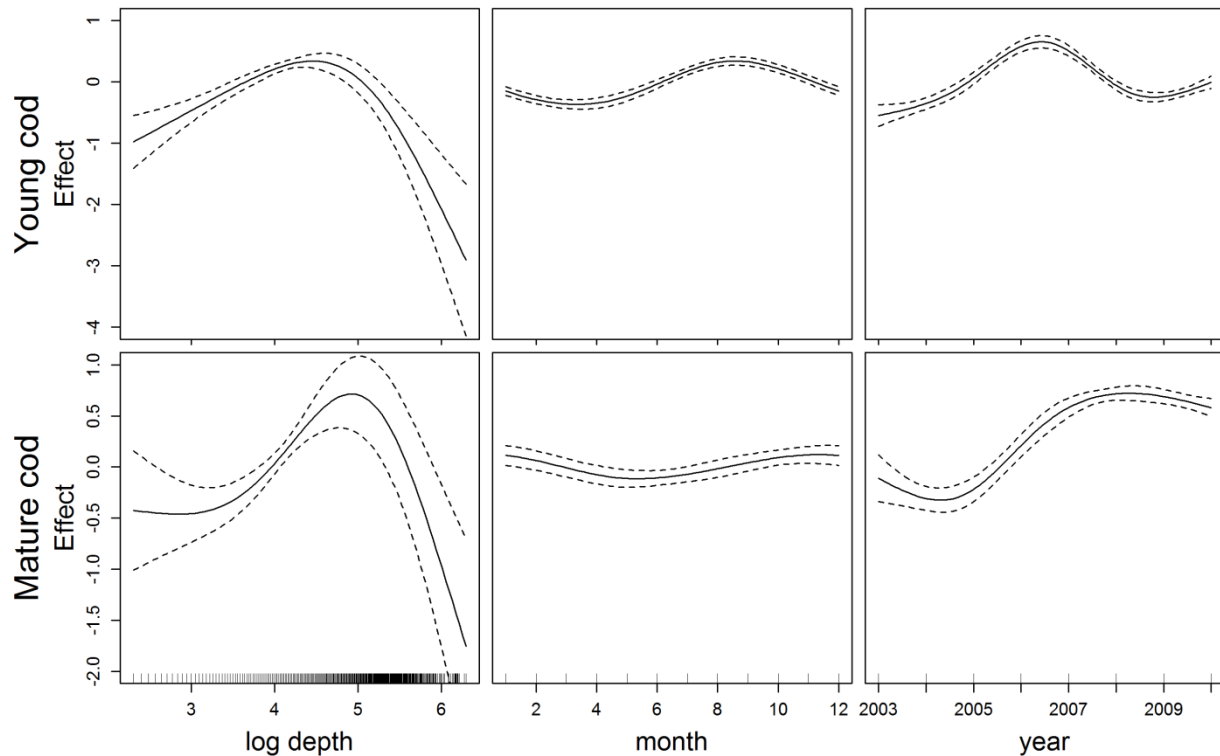

Fig. 3. Model predicted effects of significant smoothing functions (solid lines) on the discard rates of small (top row) and large cod (bottom row) in the North Sea, 2003 – 2010. Dotted lines represent the 95% confidence limits. Vertical bars along the x-axis indicate observational values.

The seasonal patterns in cod discard rates were significant for both small and large cod ($P <$ 0.001 & $P < 0.03$ respectively; Table 1), indicating discarding of small cod to be highest in late summer and early autumn while discarding of large cod to be highest in winter (Fig 3; Supplementary Fig. 2 & 3). Discarding of small cod in the late summer and early autumn reflects the species biology, fisher's behaviour, or both (Viana et al. 2011). Higher discarding of large cod in the beginning of the year can be attributed to the targeting behaviour of the fishery and fishers attempting to sustain the quota throughout

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the year, while the increase in large cod discards towards the end of the year occurs due to quota restrictions (Feekings et al. 2012).

Depth and vessel specific characteristics are also influential on discard rates of both small and large cod, while gear type was only significant for discards of large cod (Table 1). The significance of depth is due to the depth structure of the North Sea, with the shallow areas along the east coast of the UK displaying high rates of small cod discards. Discards of large cod occur further north in deeper waters.

Here we have described the spatio-temporal distribution of small and large cod discards throughout the entire North Sea and conclude that clear seasonal and interannual changes have taken place. Knowledge of the spatio-temporal distribution of discards provides valuable information for management. The mortality of cod imposed by discarding could be reduced by defining areas where the use of more selective fishing methods are mandatory and ensuring that vessels catching cod have sufficient quota to land it. Despite regulations aimed at reducing cod discards, little improvement over the period 2003 – 2010 is evident.

Discards are still a substantial source of mortality to the stock, a stress on the ecosystem, a waste of a protein source, and an impediment to stock assessments. A ban on discards will likely face economic, regulatory, and political hurdles. Under a discard ban several issues emerge including: i) How to minimise the capture of juveniles and large cod for which there is no quota under a discard ban; ii) How to ensure discarding does not take place. The success of a discard ban will depend critically on complementary management measures addressing these issues.

Co-management from fishers and other stakeholders in the science and decision-making process is essential (Beddington et al. 2007; Hall and Mainprize 2005; Gutiérrez et al. 2011; Kraak et al. 2012). Collaboration between managers and stakeholders provides the opportunity to implement spatio-temporal management solutions in real-time. Voluntary real-time closures have been successfully used to limit catches of juvenile cod in Norway (Graham et al. 2007), and the EU has recently implemented a similar scheme in the North Sea and Skagerrak, with the aim of protecting juvenile gadoids (Holmes et al. 2011). In fact, the present study would be useful to define spatio-temporal closures that would not be real-time (thus avoiding a costly and complicated monitoring) by showing where and when discard rates are high; For example, a summer closure in the 55-58°N band would avoid a space and time where discard rates of small cod are high. Whether this would avoid catching many small cod still depends on whether effort is high in this area during summer. Spatio-temporal fishery closures can also provide an incentive by providing better fishing opportunities to fishers who are using more selective fishing methods (Hall and Mainprize 2005; Catchpole et al. 2005; Dunn et al. 2011). Moreover, the proposed European discard ban includes a change, whereby all catches, not just landings, will be deducted from a quota, and fishing

operations will stop once the catch quotas are met. To maximize their revenue, fishers would avoid catching immature fish, which yield no return, and avoid an early end to their fishing season by adopting more selective fishing practices. The outputs from this study will also assist in identifying how different fisheries would be affected by this change.

The successful implementation of the proposed discard ban will require striking a delicate balance between the incentives to discard and incentives to retain unmarketable catches (Gezelius 2008). Incentives to retain currently unwanted catches can generate incentives to pursue such catch intentionally. Removing incentives to pursue such catch can create incentives to discard and misreport.

Our study is, to our knowledge, the first comprehensive synthesis of spatial and temporal discard data collected by EU Member States across a wide geographical area. Demonstrating detailed spatial temporal patterns for the first time, provides opportunities to further develop spatial management and facilitating the move to discard free fisheries as part of the proposed reforms of the CFP and illustrating what the impacts will be on European fishing fleets.

Methods Summary

Discard data

Discard data used in this study were collected in accordance with the European Data Collection Regulation and the Data Collection Framework (DCREC no. 1639/2001 and DCF EC no. 199/2008). All European Community Member States, since 2002, are obliged to collect information on landings and discards. The DCR and DCF do not outline the methodology for the data collection; consequently differences can exist in each member state's data collection programme. This analysis therefore focuses on the numbers discarded as these were collected uniformly by all member states. Discards less than and greater than MLS (small and large cod respectively) were analysed separately. The fisheries were classified based on mesh size and gear in accordance to STECF classifications (STECF 2011).

Survey data

Data from the biannual international bottom trawl survey (IBTS) are used to complement the discard data. The IBTS survey is conducted annually in January/February and in August/September, by a number of research vessels simultaneously, spanning almost the entire North Sea (Heessen and Daan 1996). All participating vessels use a standardized fishing gear. All fish are counted, except for large samples where a subsample is taken. Length-frequency distributions of catches by species in the survey are estimated for each single haul based on the counts.

Statistical analysis

Generalised additive mixed models (GAMMs, Hastie and Tibshirani 1990; Zuur 2009) were used to describe the main spatial distribution changes over time while accounting for the unbalanced sampling design between explanatory variables. Because our data are counts, numbers discarded per haul, without an upper limit and overdispersed, a negative binomial distribution (log-link) was used. To account for the variability in haul duration and individual vessels, haul duration was modelled as an offset variable and vessel as a random variable. A backward stepwise elimination (Zuur et al. 2009) was used where all covariates were included in the initial model and the least significant covariates were removed one at a time until all covariates were significant ($P < 0.05$). The analyses were performed using the R (R 2011) package 'mgcv' (Wood 2011).

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Author Contributions

GA, JJP, NM and JF designed the research with input from EvH and TC. EvH, TC, MJR, AP, JU, SV contributed data. The statistical analysis/model was formulated in collaboration between GA, JJP and JF. JF wrote the text with inputs/comments from all co-authors.

Supporting Online Material

Methods

Discard data

In accordance with the European Data Collection Regulation and the Data Collection Framework (DCR EC no. 1639/2001 and DCF EC no. 199/2008), all European Community Member States, since 2002, are obliged to collect information on landings and discards. The DCR and DCF do not outline the methodology for data collection. Consequently, differences can exist in each Member State's national data collection program. This results in issues when merging different member state's discard data. Therefore, this analysis focuses only on the numbers discarded and not the whole catch. The analysis in this study is based on data from 2003-2010 and comprises 11370 hauls in 1189 trips sampled by Member States within the North Sea demersal fisheries (Figure 1). The fisheries were classified based on mesh size and gear in accordance to Scientific, Technical and Economic Committee for Fisheries (STECF) classifications (STECF 2011). Selection of vessels and acceptance of fishers having an observer on-board differs across Member States and fleets. This results in quasi-random sampling, where not all vessels have an equal probability of being selected. As a result, some fisheries may be sampled more than others simply because of acceptance from fishers. Although collection methods differ across Member states, data sampling is stratified with regards to quarter, and fishing gear. In order for the sampling programme to be representative of the fisheries in question, vessels of all size are sampled from the main fishing harbours during the entire period of activity of a given fishery.

Survey data

Data from the biannual international bottom trawl survey (IBTS) for years 2003-2010 are used to complement the discards data. The IBTS survey is conducted in January/February and in August/September, by a number of research vessels simultaneously. The survey spans almost the entire North Sea (Heessen and Daan 1996). All participating vessels use a standardized fishing gear. The mesh size of the net is 40 mm, and is thus smaller than the gear used in most commercial fisheries. All fish are counted, except for large samples where a subsample is taken. Length-frequency distributions of catches by species in the survey are estimated for each single haul based on the counts. All individuals in the size range 0- 35 cm were included in the dataset. The survey is added as a separate "fishery" category to distinguish it from the commercial discards estimates. Differences in catchability between the survey and the commercial observations are absorbed in the "fishery" covariate. Adding the survey data in this manner gives independent and uniformly estimated spatial and temporal distribution of cod, while allowing for differences in catchability.

Management measures

Throughout the 8 year study period the EU minimum landing size (MLS) was 35 cm. However, Denmark and Belgium imposed a national MLS on their fleets. The MLS for the Danish fleets in 2003-2008 was 40 cm. The national MLS implemented for the Belgium fleets was 40 cm from the beginning of 2004 until the 30th of June 2008 and then increased to 50 cm on the 1st of July 2008.

Numerous management measures have been introduced to reduce the fishing mortality on North Sea cod. North Sea cod are managed under joint agreements and management plans between the EU Council of Ministers and Norway. Since 2000 the EU has set highly restrictive cod TACs. For 2003, a cod TAC was agreed that was consistent with a 65% reduction in fishing mortality. This was underpinned by effort restrictions in the EU zone of the North Sea (Horwood et al., 2006). The Recovery Plan was finalised in 2004 for cod stocks in the North Sea, Kattegat, west of Scotland, and the Irish Sea and revised in 2008 [European Union. 2008. Council regulation (EC) No 1342/2008 of 18 December 2008 establishing a long-term plan for cod stocks and the fisheries exploiting those stocks and repealing Regulation (EC) No 423/2004. Official Journal of the European Union, L348: 20-33]. For North Sea cod, it aimed to ensure sustainable exploitation of the cod stocks on the basis of Maximum sustainable yield through achieving fishing mortality rates of 0.4 year^{-1} or below [Article 5(2) and Article 8(4)] (Kraak et al. 2012).

Statistical analysis

Generalised additive mixed models (GAMMs, Hastie and Tibshirani, 1990; Zuur 2009) were used to describe the main spatial distribution changes over time while accounting for the unbalanced sampling design between explanatory variables. The application of additive models to fisheries data has been described in great detail by Swartzman et al. 1992. As our data are overdispersed counts (i.e. variance exceeds the mean or contain a large number of zero observations) a negative binomial distribution (log-link) was used. Since discard sampling programmes differ across countries, a simple means of combining data was required. The easiest and most effective way of achieving this was to analyse discard numbers per haul. Because the duration of the hauls is dependent on the decision of the skipper, no pre-described sampling design can be ascribed. Subsequently, the haul durations differ per observation. To account for the variability in haul durations, an offset term was used. The advantages of the offset approach compared to analysing densities are that the fitted values are always positive, the confidence intervals around the fitted values do not contain negative values, and we allow for heterogeneity within the context of a negative binomial distribution (Zuur et al., 2009). A backward stepwise elimination (Zuur et al. 2009)

was used where all covariates were included in the initial model and the least significant covariates were removed one at a time until all covariates were significant ($P < 0.05$). The final models are then reduced versions of the full models. The analyses were performed using R software, a statistical environment for computation and graphics (http://www.r-project.org), and the R package 'mgcv' (Woods, 2011).

To simplify the interpretation of the results, the maximum degrees of freedom (measured as number of knots k) allowed to the smoothing functions were limited for the variables month and log depth ($k = 4$). The full model was formulated as follows:

$$\eta(\mu_\iota) = \beta_0 + \beta_1(Fishery_i) + f_1(Longitude_\iota, Latitude_\iota) + i_1(Month_i) + f_2(Year_i)$$
$$+ f_3(LogDepth_i) + j_i(LogTime_i) + k_i(Vessel_i) + \varepsilon_\iota$$

where η is the log link function of the expected mean number discarded μ_i, β_0 is the model intercept, β_1 is a categorical term defining the different fishing gears, f_i are smooth terms fitted with thin plate regression splines, i_l is a cyclic cubic regression spline, j_i is the model offset, k_i is a random variable, and ε_i an error term.

## Supplementary Tables & Figures

Supplementary Table 1. Averages and proportions for different discard categories per fishery. Standard deviations are in brackets.

|	Trawl & Seine ≥ 100 mm	Trawl & Seine 70-99 mm	Beam Trawl ≥ 120 mm	Beam Trawl 80-119 mm	GOV Trawl
No. of trips	439	193	9	367	-
No. of hauls	4014	1608	373	5722	4315
No. of vessels	153	97	5	128	11
Haul duration (hrs.)	4.7 (1.6)	4.1 (1.6)	2.4 (0.2)	2.5 (0.8)	0.5 (0.03)
Depth (m)	126.5 (74.9)	63.3 (43.5)	40.7 (11.1)	38.0 (14.6)	74.3 (40.9)
Avg. Disc <MLS (no. /hr.)	4.0 (25.1)	2.4 (6.9)	19.9 (35.7)	4.2 (14.4)	6.5 (30.0)
Avg. Disc ≥MLS (no. /hr.)	2.3 (8.6)	2.3 (10.8)	5.3 (10.5)	1.4 (9.6)	-
Avg. discard length (cm)	34.3 (10.3)	34.5 (10.3)	34.6 (9.9)	34.3 (10.5)	18.6 (11.2)
Discard proportion < MLS (%)	52.2	54.9	79.4	74.9	-

Supplementary Fig. 1. Spatial distribution of sampling locations for the 7 national discard sampling programmes (left) and IBTS trawl survey (right) 2003 - 2010.
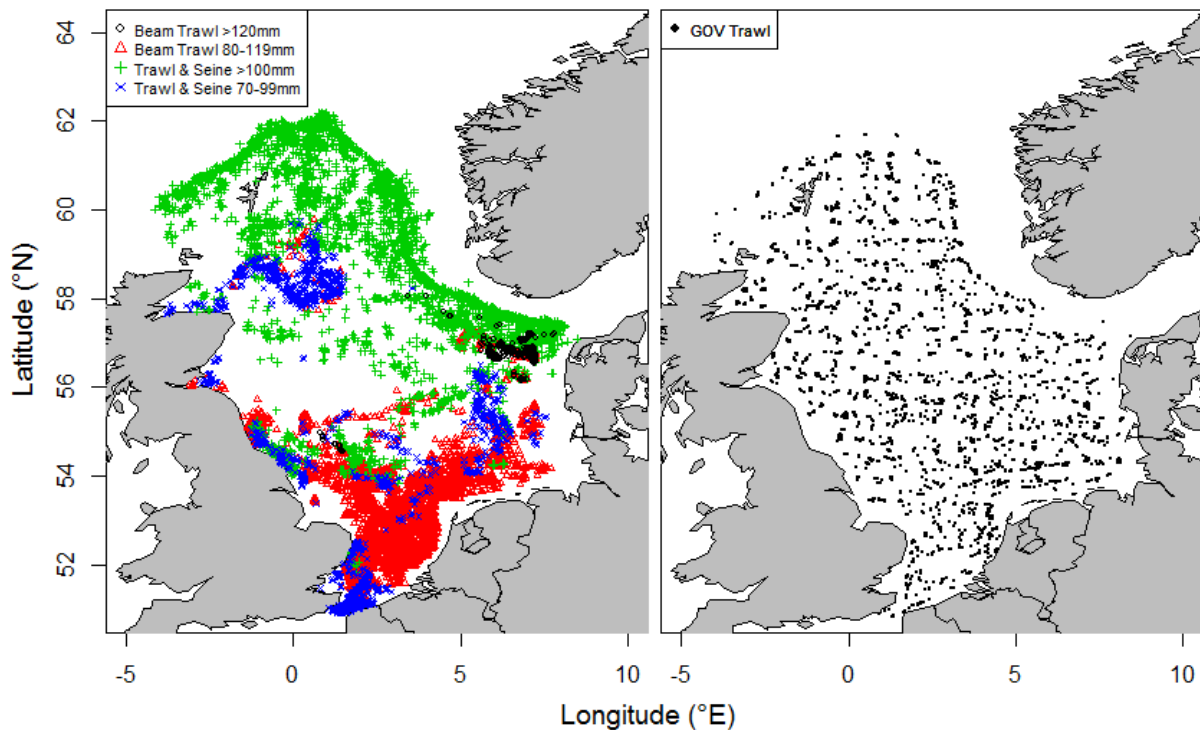
Supplementary Fig. 2. Model predicted monthly densities of young cod discards in North Sea.
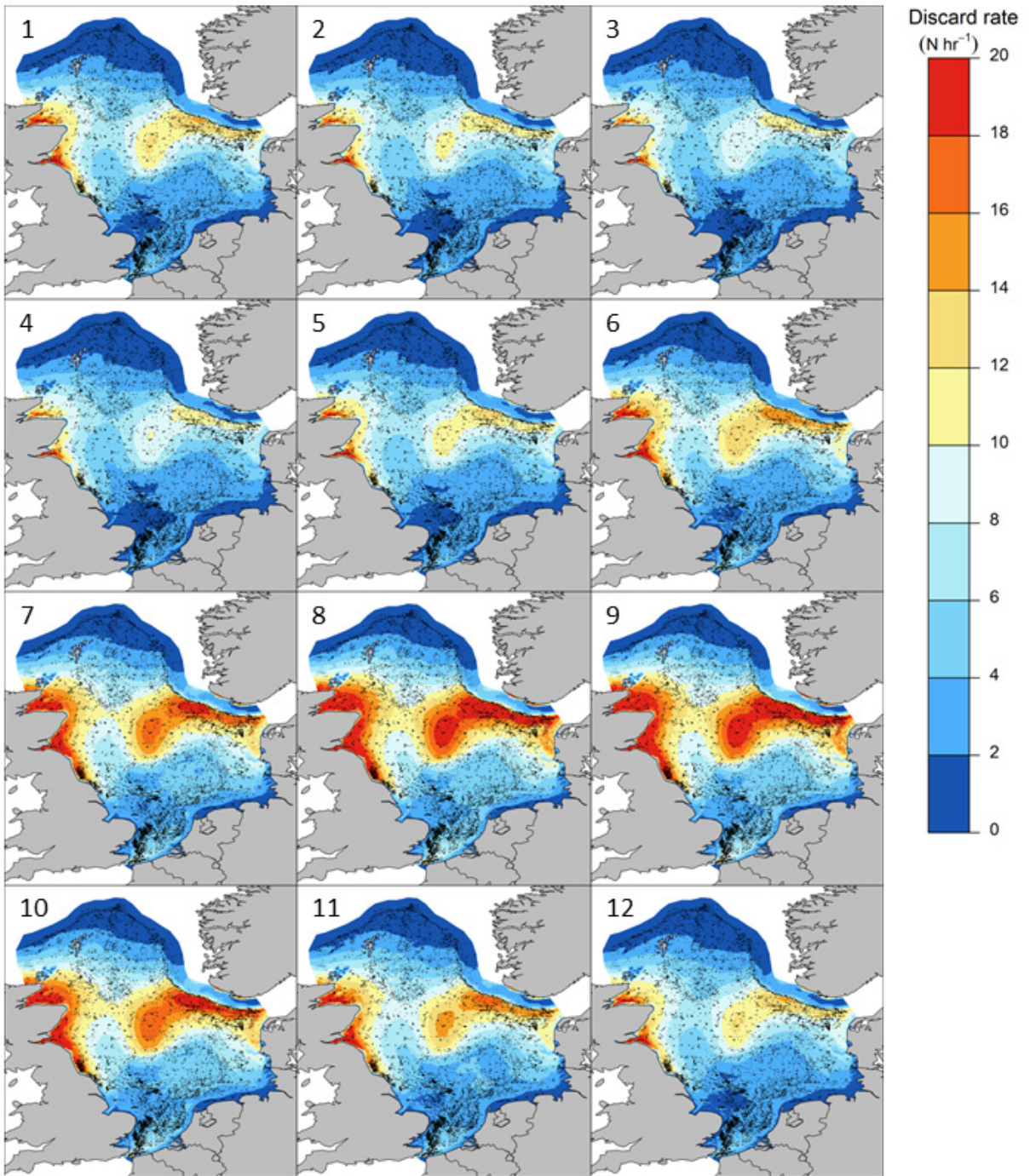
Supplementary Fig. 3. Model predicted monthly densities of mature cod discards in North Sea.
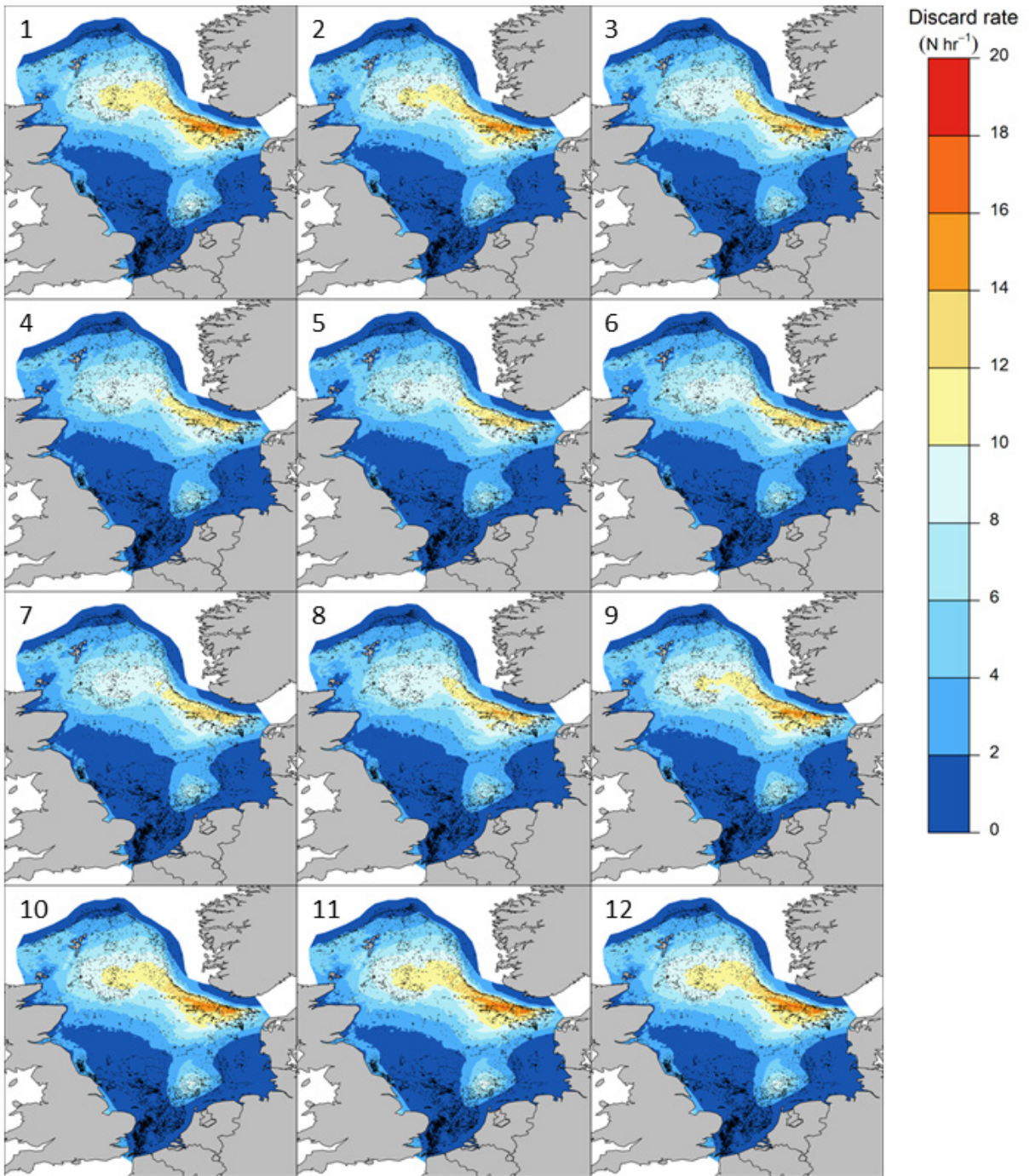
Paper IV

Discarding of plaice (*Pleuronectes platessa*) in the Danish North Sea trawl fishery

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ABSTRACT

Plaice (*Pleuronectes platessa*) plays an important role in the North Sea benthic ecosystem and is by weight the most important commercial flatfish species in the North Sea demersal fishery. There is a high discarding of plaice in the active demersal fisheries in the North Sea. The change in fisheries management towards a more ecosystem based approach, together with a greater focus on sustainability, has caused a severe need for action. Subsequently, the European Commission is preparing regulations to reduce or even ban discards. The trawl fisheries are commercially the most important Danish fishery targeting plaice. Here we analyse discard data collected onboard Danish vessels in the period from 1998 to 2008. We describe the general patterns in these data by dividing them into three mesh size categories: 80–99 mm, 100–119 mm and ≥120 mm to reflect implemented technical measures of relevance. We analyse the landed and discarded portions in these mesh size categories and link the discarding to the minimum landing size. We employed a GAM model to assess how discarding of plaice below the minimum landing size is connected to relevant factors that could be of relevance from a management perspective. We identified a statistical significant effect of mesh size category and area. We discuss the results in relation to potential mitigation measures to be implemented in future fisheries management strategies.

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1. Introduction

Plaice plays an important role in the North Sea benthic ecosystem, being one of the most abundant flatfish species and one of the most important species for the fishery (Daan et al., 1990; ICES, 2008; Sparholt, 1990). Nevertheless, the plaice fishery in the North Sea is characterized by a high discard rate, approximately 50% by weight of the catches (ICES, 2011). High mortality of discarded plaice (50–100%) is indicated from both beam trawls (Kaiser and Spencer, 1995; Van Beek et al., 1990) and trawls (Evans et al., 1994; Millner et al., 1993). Discard survival will likely depend on several factors like the fishing gear, fish and fishing conditions and an additional mortality caused by sea bird predation (Evans et al., 1994; Garthe and Hüppop, 1994; Hudson and Furness, 1988; Votier et al., 2004). Consequently, measures to reduce the amount of plaice discarded in the North Sea fisheries would greatly benefit the stock (ICES, 2011) and reduce the anthropogenic impact on the marine ecosystem. The reduction of discards is also a main issue in the 2012 revision of the European Common Fisheries Policy and a key aspect in voluntary certification of fisheries (Marine Stewardship Council, www.msc.org).

Several technical measures have been applied in the North Sea to reduce the fishing mortality on juvenile plaice. Of relevance for active demersal gears are mesh size regulations, a partially closed area (the plaice box) and a minimum landing size. The closed area (plaice box) is placed along the continental coast where vessels lager than 300 HP have not been allowed to fish since 1995 in an attempt to reduce discards of juvenile plaice (Pastoors et al., 2000; Van Keeken et al., 2007) that are concentrated in this area (Pastoors et al., 2000; Rijnsdorp and Pastoors, 1995; Van Keeken et al., 2007). There has not been proven any clear effect on the stock (ICES, 2011; Pastoors et al., 2000) but it is likely that this will increase survival of juvenile plaice. There are several mesh size regulations in force, and today the use of meshes 80–99 mm is only allowed in the southern North Sea (South of 55°N or 56°N east of 5°E) whereas the minimum mesh size in the North is 100 mm (ICES, 2011). Recent estimates on the selectivity of plaice in trawls have been made (Frandsen et al., 2009; Frandsen et al., 2010; Frandsen et al., 2011) making it possible to assess the selectivity in relation to discard mitigation measures.

Discarding of plaice is recognized as a major management problem and several aspects of plaice discarding in the North Sea have been assessed (Aarts and Poos, 2009; Berghahn and Purps, 1998; Depestele et al., 2011; Dickey-Collas et al., 2007; Evans et al., 1994; Poos et al., 2010; Van Beek et al., 1990). There is, nevertheless, a lack of publications that analyse and describe the general discard pattern of plaice in the North Sea trawl fishery with focus on potential mitigation measures. A particular reason is that discard sampling programmes are often expensive and require a large number of man hours, while providing data which are spare in relation to the total effort in a given fishery (Aarts and Poos, 2009; Dickey-Collas et al., 2007). However, discard

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analyses are of importance for inclusion in stock assessment and the associated management advice (Dickey-Collas et al., 2007).

Several technical measures have been implemented over the years, however, the level of discarding remains high (ICES, 2011). Therefore, evaluating factors that can be used by managers to reduce discard amounts and rates is of importance. Discard reduction is also an important facet of the long term management plan implemented in 2008, aiming at having the stock within safe biological limits (ICES, 2010). Trawls are the most important fishing gear targeting plaice in the Danish North Sea fishery. Discard data from the Danish fishery have been collected since the nineties. This provides the possibility to assess discarding for a relatively large number of hauls. Here we analyse discard data collected from the Danish demersal trawl fishery in the North Sea in the period from 1998 to 2008. The main aim is to analyse the discard data with particular focus on factors that could be important for management strategies in the future.

2. Methodology

2.1. Discard sampling

Danish discard data was originally collected under a national programme and later (2002) in accordance with the European Data Directive (1639/2001). Data sampling is stratified with regard to: ICES sub-division, quarter, and defined by mesh size categories. Sampling is carried out on board commercial vessels voluntarily participating in the discard sampling programme. The observer has no relation to the control units, whereby it is assumed that the fishing practice is unaffected by his presence. The vessels and trips are chosen to be representative of all important fishing harbours, the entire period, all vessel sizes and all durations of trips in the given fishery. The criteria for hauls included in this analysis are that they are conducted in the North Sea with demersal trawls in the period from 1998 to 2008 with a mesh size of 80 mm or larger. The trawls are fished in single or multiple rigs often from a single vessel, but in a few cases pair trawling is recorded (trawls towed by two vessels). These fisheries are generally defined by being mixed fisheries targeting species for human consumption. The fishery is complex, having fluctuating catch compositions and targeting a wide range of species.

2.2. Relevant factors

Many social (Catchpole et al., 2005; Rochet and Trenkel, 2005), technical (Rochet and Trenkel, 2005; Stratoudakis et al., 2001) as well as environmental (Catchpole et al., 2005; Rochet and Trenkel, 2005) factors can be considered as having a potential effect on discards. However, this analysis is focussed on variables that are relevant for management of the fishery because they are directly controllable. Factors considered include geographical area, season (quarter) and mesh size category (defined by mesh size intervals). The discard rate could be largely positively correlated with juvenile abundance. Although this is not controllable by managers it has the potential to distort the effect of other influential factors such as mesh size and was therefore also considered. We considered the size spectrum from the minimum retention length up to the MLS by using recruitment data for age classes one and two from the official assessment (ICES, 2010). We assessed juvenile abundance by year and quarter, assuming mortality (natural and fishing mortality) to be constant throughout the year. Year classes one and two were assessed separately and also their effects combined.

We defined haul location as the midpoint in the tows (straight line between start and end). To increase the number of observations for each time period we used quarter (start 1st January) rather than month. Since mesh size will influence the selectivity in the gears, it is considered to be a main factor affecting the discarding of plaice (ICES, 2011; Van Keeken et al., 2007). We divided mesh size into

three main categories: 80–99 mm; 100–119 mm and $\geq$120 mm (largest observed size is 127 mm), reflecting the regulations. These include minimum mesh sizes of 80 mm in the southern North Sea, 100 mm in the northern North Sea and 120 mm for the whitefish fishery in the northern North Sea. This division also ensures a reasonable number of observations per mesh size category. The conventional mesh size is noted by the discard observers but is not necessarily measured. Since mesh size regulations have changed over the years, some of the discard observations are not in line with current legislation, having smaller mesh sizes than allowed in the area today. The mesh sizes are reported to be somewhat larger (around 5 mm) than the minimum allowed. This is to avoid potential conflict with the legislation, since the size of the mesh can decrease over time. It is rarely that the conventional mesh size is below the minimum allowed. The use of selective devices is not well documented in the discard data. However, square mesh panels have been implemented in legislation in some fisheries during the study period. These panels are inserted with the objective of improving the selectivity of gadoids, particularly cod, and do not fit well to the morphology of plaice. Subsequently, they are not expected to influence the selectivity of plaice (Frandsen et al., 2010; Madsen et al., 2006) since the minimum allowed mesh sizes in the square mesh panels are not substantially higher than that used in the rest of the codend. A minimum landing size (MLS) of 27 cm is effective for the whole investigated period.

Since a mismatch between mesh size and MLS is likely to be influential on discard rates we used recently published data on plaice selectivity in trawls (Frandsen et al., 2009, 2010, 2011) to assess this relationship further. We estimated mean values from 4 experiments assuming the selection factor (L50 (50% retention length)/mesh size) and also the selection ratio (selection range (75% retention length − 25% retention length)/L50) to be constant. This is because the selection range can increase with L50 and hence mesh size (Madsen, 2007). The average selection factor was estimated to 2.15 (range 2.04–2.28) and the average selection ratio to 0.146 (range 0.108–0.182).

2.3. Comparisons of mean values

Mean values for each mesh size category were estimated for discards and landings (no./hour), discard proportions (no.) and lengths (cm) of discards and landings. To conduct a more detailed analysis of discarding mean values in relation to MLS the total proportion below MLS, two length intervals just below MLS (23–24 cm and 25–26 cm) and a length interval just above MLS (27–28 cm) were estimated. All mean values were compared pair-wise by a two-sample t-test. In principle, this requires that the mean values approximately follow normal distributions. The observations are not normally distributed particularly because most cases include zero discard observations. However, the positive observations exceed the zeros in most cases. It was examined by bootstrap if the mean values approximately follow a normal distribution. This was done for each set of observations as follows: 1) draw a random sample of the observations with replacement; 2) calculate the mean; repeat steps 1 and 2 10,000 times; draw a histogram and a qq plot of the 10,000 simulated means. The plots indicate that the normal approximations seem reasonable. The applicability of the t-test is further justified by Sullivan and D'Agostino (1992), even in cases with up to 50% zero observations.

2.4. Modelling discarded numbers under MLS

To describe the main reasons for discarding that are relevant to management we modelled discarded numbers under MLS per haul as the response variable. However, as haul durations differ per observation we may simply measure a large number of discards because the haul duration was long. To account for this we used log haul duration as an offset term, whereby the fitted values are always positive,

the confidence intervals around the fitted values do not contain negative values, and we allow for heterogeneity (Zuur et al., 2009). To account for the unbalanced sampling design between explanatory variables, and describe the main spatial distribution, generalized additive models (i.e. GAMs, Hastie and Tibshirani, 1990) were used with the assumption of an underlying binomial distribution (log link). A negative binomial distribution was chosen a priori seeing as the data are counts without an upper limit, and overdispersed (i.e. variance exceeds the mean or contain a large number of zero observations). The full model was formulated as follows:

Numbers discarded under MLS per haul
= mesh size category + s(longitude, latitude) + quarter
+ s(juvenile abundance) + offset(log haul duration) + ε

where s is an isotropic smoothing function (thin-plate regression spline), and ε is an error term.

Covariates included in the initial model were removed one at a time until all covariates were significant ($P<0.05$). To simplify the interpretation of the interaction between longitude and latitude, the maximum degrees of freedom (measured as number of knots k) allowed to the smoothing function was limited to k = 20. To check for violation of independence within the spatial term variograms of the residuals were used (Zuur et al., 2009). No spatial correlation was present in any of the models fitted. The analysis was performed using R software, a statistical environment for computation and graphics (http://www.r-project.org).

3. Results

General information on the discard sampling data is provided in Table 1. Fewest years are covered by the 80–99 mm mesh size category where there are data from 2001 to 2005, whereas the 100–119 mm mesh size category has data from 1998 to 2002, 2004 and 2007. Most years are covered by the ≥120 mm mesh size category which covers the years 1998, 2002–2008. The mean horsepower increases with the mesh size (Table 1). The mean haul duration is highest for the 80–99 mm mesh size category (Table 1). The highest numbers of hauls were recorded for the ≥120 mm category. The relative proportion of hauls having discard observations decreases with increasing mesh size. There is about the same amount of hauls with landings and discards in the two smallest mesh size categories but a higher number of landings than discards in largest mesh size category. Landings have not been length measured in all hauls (only total landings weight recorded in 15 hauls) for the 100–119 mm mesh size category (Table 1).

The geographical distribution of the observations is indicated in Fig. 1. There is some difference in the spatial distributions of the mesh size categories. The 80–99 mm mesh size category was located mainly in a narrow area in the central North Sea, while the ≥120 mm mesh size category was more widely distributed with more observations in

the northern parts of the North Sea. Means for different discard and landing categories, together with their 95% confidence intervals and significance levels from the comparisons of the three mesh size categories (t-test) are presented in Table 2. The mean discard rate, in numbers per hour, in the ≥120 mm mesh size category was approximately a factor of 10 and a factor of 30 lower than in the 100–119 mm and 80–99 mm mesh size categories respectively. Differences are statistically different between all mesh size categories. No statistically significant differences were observed in mean landings (numbers) per hour for the three mesh size categories. Mean discard proportions are statistically significantly different for all categories. The mean length of discarded plaice increases significantly with increasing mesh size category. The main part of the discarding is below MLS for all categories, but lowest in the ≥120 mm category, having 69% of the discards below MLS. In total, more than half of the discards are in the size range 23–26 cm, with the main part being in the 25–26 cm interval just below MLS. The discarded proportion in the 27–28 cm interval, just above MLS, is lowest in the 80–99 mm and 100–119 mm mesh size categories while highest in the ≥120 mm mesh size category.

Selection curves indicating the selectivity of 80, 100 and 120 mm mesh sizes are shown in Fig. 2 together with the MLS. The retention of plaice at MLS (27 cm) for the three mesh sizes are 100%, 98% and 67% respectively, indicating a mismatch also for the 120 mm mesh size. The 10% retention lengths are 15, 16 and 22 cm respectively.

The correlation between juvenile abundance and discards under MLS was found to be negative in the GAM model. Inspection indicated that this was probably caused by the structure of the data. A low number of observations were recorded in years with high juvenile abundance. These also corresponded to the largest mesh size category. Having no significant positive correlation, juvenile abundance was excluded from the analysis. Quarter was found to be non-significant and removed from the final model. Significant variables in the final model are presented in Table 3. The discard rates in the ≥120 mm and 100–119 mm mesh size categories were significantly lower than the 80–99 mm mesh size category. However, the difference between the 100–119 mm and ≥120 mm mesh size categories was non-significant. The interaction between longitude and latitude on discard rates was also found to be highly significant (Table 3, Fig. 3). Discarding is highest in the area closest to the plaice box in the south east and decreases with increasing distance. The proportion of null deviance explained by the final model was 85.3%.

4. Discussion

This is the first detailed study of the general discarding pattern of plaice in the North Sea trawl fishery (but see Poos et al., 2013–this volume). There are some clear patterns of importance for the future management strategy. The discard data analysed here are not collected by random sampling since it is stratified and representative under certain criteria. The data used covers a very limited amount of the relevant fishery (<1%) and there is not a complete overlap in the data of the three mesh size categories concerning time and area. Some of this is accounted for in the modelling by including some important variables, but there might be other influential variables, for example, ecological factors (Pastoors et al., 2000), that are not fully accounted for in this study. Although there was no effect of increasing discard rates with high year class strengths of juvenile plaice more detailed recruitment data and discard data in time and space are needed to assess this in further detail. The indication of a clear area effect is in agreement with the expectation of a high number of younger plaice closer to the nursery grounds (Rijnsdorp and Pastoors, 1995; Van Keeken et al., 2007) and the plaice box. This also suggests that discard mitigation measures for plaice are more important the closer the fishery is conducted to the plaice box.

The present analysis suggests that mesh size is highly influential on the amounts and rates of plaice discarded. The discarded proportion of

Table 1
General information for the three mesh size categories per haul. Standard deviations are indicated in brackets.

	80–99 mm	100–119 mm	≥120 mm
Years (no.)	5	7	8
Vessels (no.)	9	13	33
Mean engine power (hp)	533 (290)	639 (219)	777 (195)
Mean haul duration (hrs)	6.5 (1.3)	5.9 (1.4)	6.3 (1.4)
Trips (no.)	10	22	71
Hauls total (no.)	57	139	423
Hauls without plaice (no.)	1	63	230
Hauls with discard (no.)	53	63	129
Hauls with plaice landings (no.)	56	64[a]	187

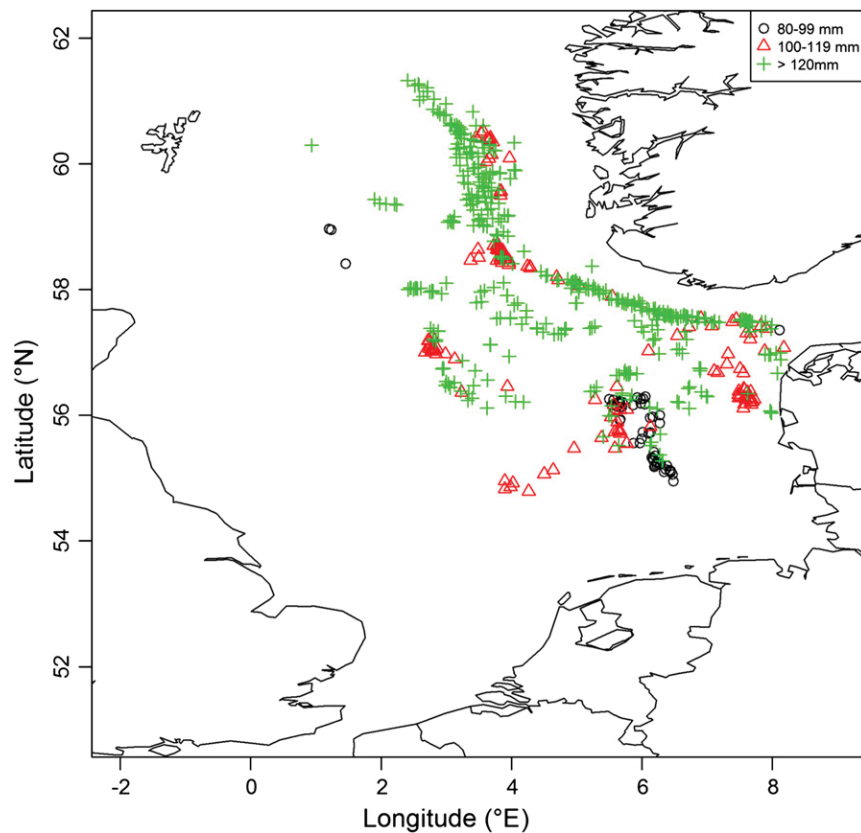[a] No length measurements in 15 hauls.

Fig. 1. Geographical indication of place for individual hauls (midpoint of tow).

the catch for the 80–99 mm mesh size category is in the same order of magnitude as observed in the management of plaice today (ICES, 2011) which is mainly caused by the 80 mm beam trawl fishery in the southern North Sea. Van Beek et al. (1981) estimated a selection factor of 2.1 in beam trawls, which is close to the selection factor we used (2.15), suggesting that the selectivity in beam trawls and otterboard trawls are comparable. More recent selectivity estimates from beam trawls would, however, be valuable.

The demonstrated mismatch between MLS and gear selectivity describes the discards even when using 120 mm mesh size. A MLS that corresponds to 25% retention has been mentioned as a management objective (Reeves et al., 1992). This would equal a mesh size of 135 mm (using selection estimates presented in the Results section) but still causes discarding, suggesting that it is also relevant to consider the MLS. The present analysis suggests that lowering the MLS by 2 cm would reduce about half of the discards in the ≥ 120 mm mesh

size category, whereas 4 cm could potentially reduce discards by 54–86% for the three mesh size categories. An increase in the selectivity clearly reduces the discard level. However, efficient species selective devices are needed to retain commercial species like sole and Norway lobster that are relatively smaller than plaice. Recent experiments suggest that some devices are efficient in releasing plaice in Norway lobster directed fisheries (Madsen et al., 2010) and pulse trawls might be able to increase catchability and selectivity of sole (ICES, 2011). The dynamics and complexity of the mixed fishery (Andersen et al., 2010; Rijnsdorp et al., 2007) where plaice is mainly targeted, as in the Danish fishery, makes the potential effect of an increase in selectivity more unpredictable in economic terms. However, this effect may be less consequential seeing as the fishery is dependent on many species. The length at first maturity is 20–24 cm in males and 30–35 cm in females (Van Keeken et al., 2007). Therefore, changing the fishing strategy by lowering the MLS could increase

Table 2
Means for different discards and landings categories per haul. Confidence limits (95%) are indicated in brackets. Asterisks in the first, second and third columns represent the significance levels from the two-sample t-tests for the comparisons between 80–99 mm and 100–119 mm, 100–119 mm and ≥ 120 mm, and 80–99 mm and ≥ 120 mm mesh size category respectively.

	80–99 mm	100–119 mm	≥ 120 mm
Discard (no./hour)	146.0 (102.3–189.7)***	47.2 (23.8–70.6)***	5.4 (3.5–7.3)***
Landed (no./hour)	70.9 (56.7–85.1)[ns]	66.1 (33.7–98.5)[ns]	68.2 (49.1–87.3)[ns]
Discard proportion (no.)	0.546 (0.481–0.611)***	0.177 (0.119–0.235)***	0.069 (0.051–0.087)***
Discard length (cm)	23.3 (22.3–24.3)**	25.0 (24.8–25.2)***	26.0 (25.7–26.3)***
Landed length (cm)	31.5 (30.8–32.2)[ns]	32.8 (30.2–35.5)[ns]	33.4 (32.9–33.9)***
Discard<MLS	0.960 (0.938–0.982)**	0.892 (0.848–0.936)***	0.689 (0.629–0.749)***
Discard proportion 23–24 cm	0.260 (0.215–0.305)[ns]	0.270 (0.224–0.316)***	0.147 (0.109–0.185)***
Discard proportion 25–26 cm	0.279 (0.216–0.342)***	0.588 (0.540–0.636)**	0.486 (0.432–0.540)***
Discard proportion 27–28 cm	0.029 (0.017–0.041)***	0.083 (0.057–0.109)***	0.212 (0.165–0.259)***

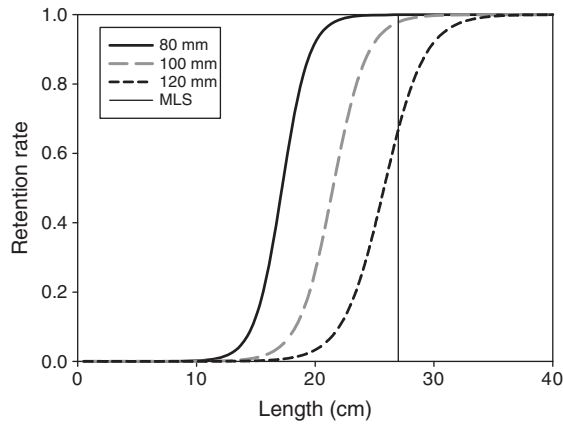ns: non significant (P ≥ 0.05).
**P < 0.01.
***P < 0.001.

Fig. 2. Selection curves for three relevant mesh size categories. The MLS is indicated by the vertical line.

Table 3
Final model results estimated by the GAM model. SE indicates standard error.

Parameter	Estimate	SE	Z-value	P-value
Categorical terms				
Mesh size category 80–99 mm	−3.25	1.11	−2.95	0.003
Mesh size category 100–119 mm	−3.75	1.08	−3.46	<0.001
Mesh size category ≥120 mm	−3.82	1.07	−3.57	<0.001

Parameter	Estimate	SE	Chi squared	P-value
Smooth terms				
s (longitude, latitude)	18.5	18.8	716.3	<0.001

Number data points: 619.
Dispersion parameter: 0.48.
AIC: 3012.

fishing mortality in areas with more small plaice and could reduce the spawning stock, particularly for females.

The majority of the North Sea Danish trawlers in the fishery for human consumption species fish with meshes ≥120 mm. While this fishery has a relatively low discard rate of plaice, a considerable portion is comprised of individuals over MLS. This is most likely a result of catch restrictions causing additional discards. Also, the higher proportion of plaice discarded just above MLS could potentially be a symptom of imprecise sorting (by eye) that could be solved. The plaice landings in the ≥120 mm mesh size category suggest that it is possible to substantially reduce discarding without reducing landings at the same time. With very few additional means it would be possible to eliminate the discarding of plaice in the ≥120 mm fishery in accordance with the direction of future fisheries policy. Abandoning a MLS and avoiding fishing without having sufficient possibility to land plaice (e.g. quota) seems to be a realistic option, especially if the goal is to eliminate discards completely. In this respect, there are considerable prospects in using new technology like electronic monitoring (Kindt-Larsen et al., 2011), that will make it possible to change from landings to total catch quotas and hence support new strategies for fisheries managers and give fishers the incentive to avoid discarding part of their catch.

From an isolated fisheries management point of view, a reduction in high discard rates would be expected to benefit the stock considerably. A more holistic approach should take into consideration the general ecosystem effects (Botsford et al., 1997; Greenstreet and Rogers, 2006; Jennings, 2005). Since discards are an additional food resource for opportunistic benthic scavengers (Groenewold and Fonds, 2000) and sea birds (Evans et al., 1994; Garthe and Hüppop, 1994; Hudson and Furness, 1988; Votier et al., 2004), their removal from the system may disrupt the dynamics in other parts of the marine ecosystem.
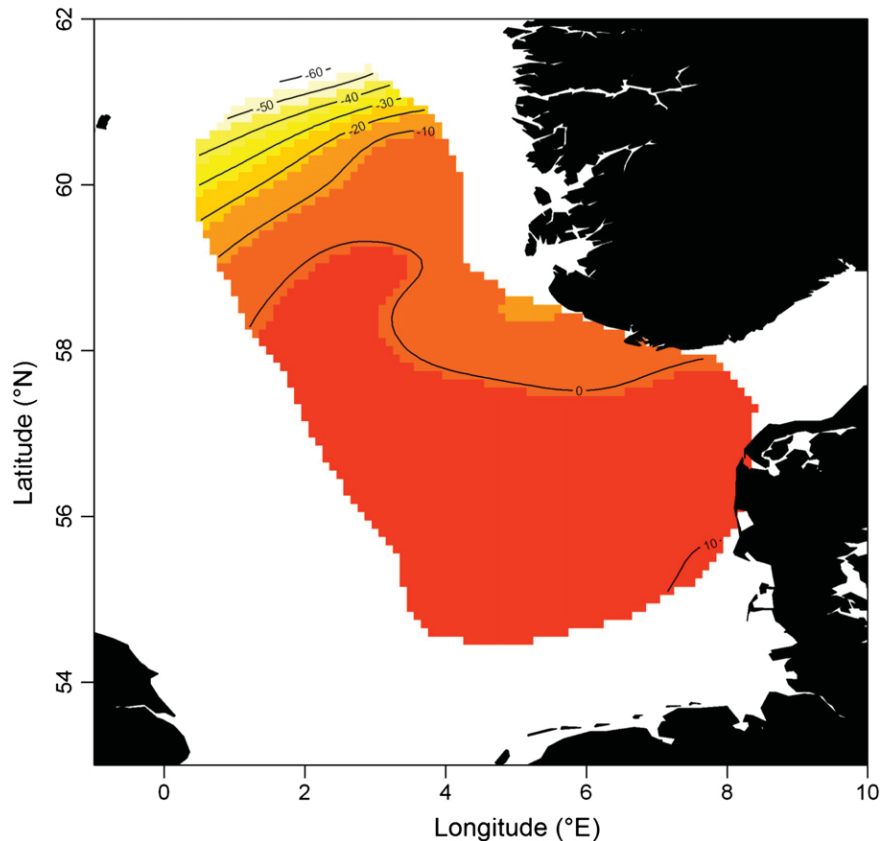

Fig. 3. Geographical indication of the relative numbers discarded under MLS per haul predicted by the GAM model. The surface and contour lines describe the effect of the two dimensional smoothing function on the geographical coordinates. Red area indicates highest discard rate. Only areas covered by observations are indicated.

Acknowledgement

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