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DiscardLess

Strategies for the gradual elimination of discards in European fisheries

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Deliverable 1.1

Review on the current effect of discarding on fish stocks and marine ecosystems; on data, knowledge and models of discarding; and identification of knowledge gaps in all case studies

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Summary

Discards are defined as the part of the catch that is brought on deck only to be returned to the sea.

This Deliverable is the result of an extensive litterature review on the state of knowledge regarding the biological, ecological and technical aspects of the discard issue. Additionally, factsheets have been collated for the various DiscardLess case studies regarding the more specific knowledge available for the various regions.

The main findings are summarized as follows:

Discarding occurs for a number of reasons including low or no commercial values, undersized fish, or the absence of quota. Discards are driven by a variety of factors (sociological, technical, legislative, environmental and biological) which form a complex network of often interwoven causes and effects, and are highly species, fisheries, and areas specific. However, the root cause is the lack of selectivity of fishing gears or operations, notably in trawl fisheries.

Methods for reducing discards are often fishery, fleet, or area specific. They can be grouped into two categories:

- Improvements to gear selectivity. For trawls, most of the selection happens in the codend, and modifying mesh size and/or shape is the simplest and most commonly used measure to improve size selection. Additional techniques aiming at sorting the catch in the codend according to morphology include selective grids and square mesh panels (or escape windows). Alterations of other parts of the trawls are also explored, as are the use of deterrent such as physical, acoustic, and electronic modifications. In the case of multi-species fisheries, the range of minimum landing sizes (implemented to protect smaller immature individuals) cannot be all harmonised with the selectivity of fishing gears. Short-term economic losses due to reduced marketable catch are identified as the main reason discouraging the uptake of more selective gears.

- Temporal and spatial closures. The main objectives are to avoid areas of high juvenile abundance or high by-catch species abundance. While effective, spatial closures can have side effects (e.g. increase of the fishing effort in other places) and low acceptance (do not affect all fishers equally depending on their home ports, preferred fishing grounds or countries). Real Time Closures, i.e. triggered by information gained in real time aboard fishing vessels, provide more flexibility than permanent or seasonal closures, and account for the variability in the timing and location of large bycatch of juveniles. It requires a strong collaboration between managers and the industry, and efficient dissemination of information to fishers. In general, the use of closures is becoming less prescriptive and more incentive based.

Discard bans have been implemented in several countries (Norway, Iceland, New Zealand and Canada). In Europe, the obligation to land catches of regulated species is the cornerstone of the 2013 Reform of the Common Fisheries Policy, to be gradually implemented between 2015 and 2019.

The retention of discards include additional costs for the industry, related to sorting, storage and landing of unwanted fish, therefore a discard ban should theoretically encourage fishers to develop fishing techniques and strategies to become more selective. Implemented together with a strong monitoring program, it also ensures that more accurate catches are recorded, and subsequently allows more accurate total allowable catches being set. If currently discard size classes are counted against the available species quota, a decrease in overall fishing pressure may also result from such a ban. However, a discard ban will only be effective with extensive monitoring, which may not be economically viable. It might also develop, if not carefully set up, a new market for discards and therefore establish incentives for their capture. Finally, although selective fishing benefits the fisheries (through reduced sorting time, simpler catch handling and processing , and more space on-board for higher valued commercial species), and reduces the overall target species fishing mortality (through reduced catch of the juvenile stages), 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. It is argued that selective fishing alters the existing community structure, spectrum of biodiversity, and species and size diversity. The question of the desirability and the feasibility of unselective fishing (i.e. 'balanced harvesting') is still a highly debated case.

Mortality of discarded fish can represent a significant portion of total fishing mortality. Discarded fish can die from damages inflicted during the fishing process (gear, handling on deck…), from predation by seabirds, or midwater/bottom dwelling scavengers, or from the impossibility to return to a suitable habitat. The survival rate of discarded fish varies depending on the species and the fishing technique and is generally unmeasured. It subsequently represents a large source of uncertainty in estimates of fishing mortality worldwide. The effect of discarding on fish population logically depends on the survival rate of discarded individual. Solutions to mitigating discards mortality have been explored, however, the escape of unwanted organisms before hauling should be promoted during fishing, as the mortality of escapees are considerably lower than of discards for a majority of species.

There are indications that discarding has altered the ecosystem functioning of some seabirds communities and has negative effects on charismatic and endangered species (such as sea turtles and marine mammals). A reduction in discards may lead to a food shortage for seabirds as well as some scavenger species and possible shifts in species composition. The effect of this shortage depends on the ability of the seabirds and scavengers to compensate by switching to other food sources. This may limit the direct effects on these species, but may also cause unpredictable cascading effects on other species through increased predation and/or competition. Recent and current research focus on the identification of the main scavenging species feeding on discards, and estimate consumption rates using traps or video surveys.

In EU, the collection of discards data has been framed in the Data Collection Framework and national sampling program have evolved accordingly. Data are shared and used through the STECF and the ICES. In spite of the resources allocated by the EU and Members states, less than 1% of the fishing trips are sampled because of the high cost of sampling programs. Therefore the discard estimates used in the assessment and management advice for European stocks are considered the best available knowledge, but they remain highly uncertain.

In the ICES areas, the integration of discards in stock assessment has improved, and in 2013, 26 stocks included discards in their assessment. ICES has generalised the basis of advice as being catch advice, instead of the previously used landing advice, and advice sheets include a mention on discard estimates either quantitative or qualitative.

In the context of the DiscardLess project, ecosystem models allow to study the interactions between the effect of the removal of the flow of discards to the ecosystem, as well as the effect of new fishing strategies on fish stocks. Five types of models will be used in the project: Osmose, Ecopath w Ecosim, Atlantis, StrathE2E and ISIS-Fish, distributed over 7 Case studies. They differ in their assumption and settings as well as in their representation of the discarding process and the fate of discards.

Contents

1 Introduction

Discarding is a global problem. Throughout this review, 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 performed by the UN FAO have been as high as 27 million tonnes annually (Alverson et al., 1994). The most recent global 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). An update of these worldwide estimates is foreseen for 2016. 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 (otter trawl and beam trawl) fisheries (Catchpole et al., 2005a). The most recent figures from the North Sea revealed that approximately 40 % of the catch (in weight) from demersal fisheries was discarded and consisted mainly of plaice and dab (Quirijns and Pastoors, 2014). Consequently, the main focus throughout the review is on discards within demersal trawl fisheries, however, those fisheries which are less problematic are also discussed.

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; Bicknel et al., 2013) 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 committed to eradicating the problem (EC, 2007). Indeed, the obligation to land catches of regulated species is the cornerstone of the 2013 Reform of the Common Fisheries Policy (EU, 2013), to be gradually implemented between 2015 and 2019.

High discard rates have been reported in many trawl fisheries worldwide, with European fisheries being no exception. For example, the proportion of the catch discarded within Danish demersal trawl fisheries has historically 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; cf also DiscardLess Deliverable D2.1). In order to be able to propose solutions, the driving factors need to be identified. The effect and relative importance of these factors will vary for different species, vessels, and fleets, and will fluctuate over time and space. Consequently, it is important to define such driving factors for each individual fishery and area.

The most common means of reducing 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).

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.

In the present review (sections 2 and 3), 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.

The following sections focus on the effects of discarding, both economical and ecological (section 4), and how discards data are used in the evaluation and management of fish stocks (section 5).

Finally, the ecosystem models used in the project are described with an emphasis on how they model both the discarding process and the fate of discards in the ecosystem (section 6).

Section 8 contains the factsheets prepared by all Case Studies in the DiscardLess project. They describe the current situation in each of them, and thus define the state prior to the implementation of the Landing Obligation.

2 Current discarding practices

2.1 Problematic fisheries.

Discarding occurs in nearly almost all fisheries worldwide, however, there are a few which are responsible for the bulk of the discards. These are mainly mixed demersal trawl fisheries. The main reason behind the high discards in these fisheries is due to their mixed nature, where many different species are caught and sometime targeted simultaneously. Optimising the size and species selectivity of a trawl to match the often very different morphological and behavioural characteristics of the target, and for that matter non-target, species is a very challenging task which has been studied for almost a century.

2.2 Causes of discarding

Knowledge about the various factors influential to discarding is essential when designing management strategies to maximise landings and minimise discards (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).

2.2.1 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 (e.g. mesh twine thickness, number of meshes, size and placement of selective devices) 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).

2.2.1.1 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 (Madsen and Valentinsson, 2010; Feekings et al., 2012). On the other hand, selective improvements in the Baltic Sea demersal cod trawl fishery were needed to reduce catches of undersized cod (Feekings et al., 2013;Figure 1).

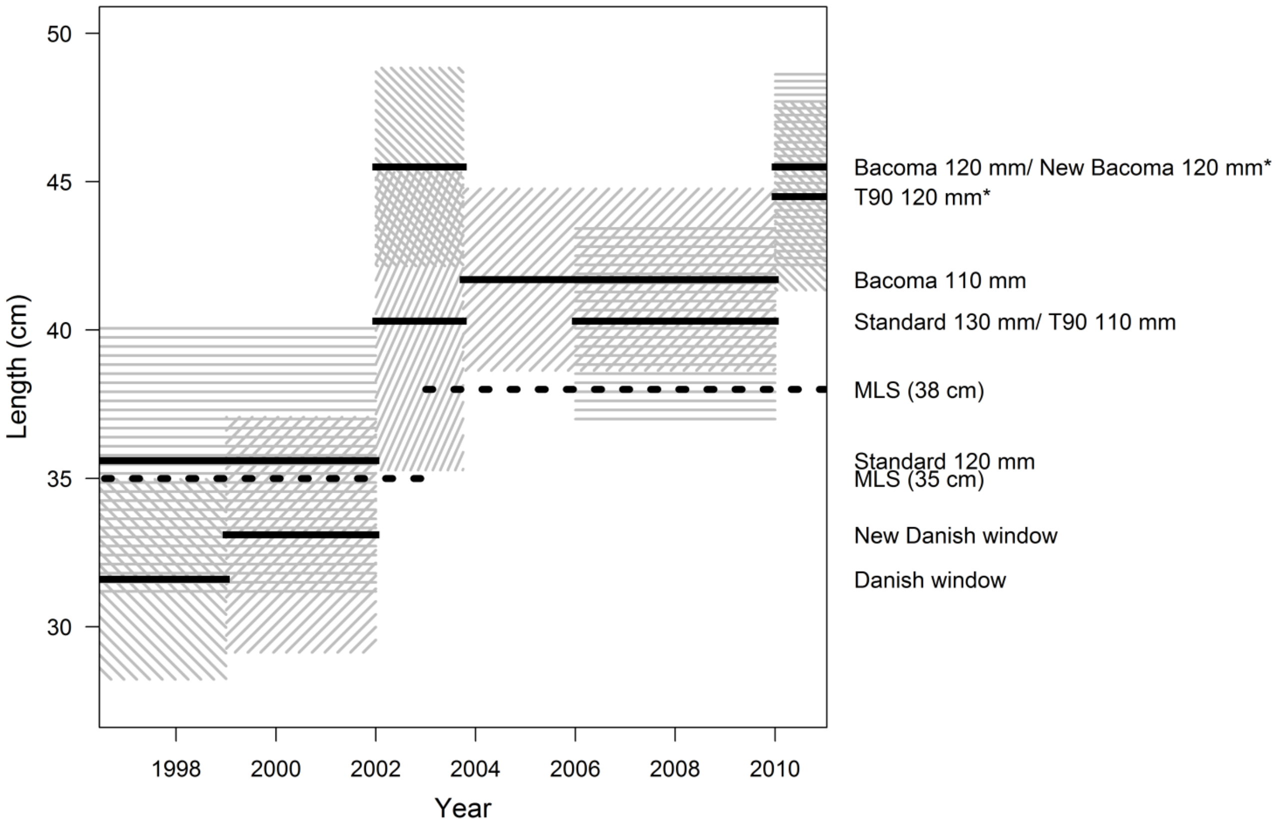

*Figure 1: Changes in codend selectivity (L50 and selection range, SR) for the regulated towed gears and changes in minimum landings size (MLS). *minimum mesh opening of 120 mm from 1 January 2010 in subdivisions 22–24 and from 1 March in subdivisions 25–32. The selectivity parameters for the T90 120 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 (Feekings et al., 2013).*

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.

2.2.1.2 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; Feekings et al., 2012). 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).

2.2.1.3 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 (Murawski, 1996; Machias et al., 2001; Feekings et al., 2012). Conversely, Rochet and Trenkel (2005) found discarding increased nonlinearly and may be due to fish being damaged by long hauls or clogged nets preventing escapement. 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).

2.2.1.4 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 (e.g. longer trips result in more discards) on discards of the most perishable species (cuckoo ray, hake and red gurnard), while a negative effect for megrim and *Nephrops*.

2.2.1.5 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 a stern trawler has a lower size selectivity when compared to a side trawler due to the hauling procedure, e.g. how slack the net becomes during hauling (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. 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). The vessel effect was found to have a significant effect on discards in a majority of cases; Feekings et al., 2012; Feekings et al., 2013). Despite this, Borges et al. (2001) found the vessel characteristics not to be influential.

2.2.2 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).

2.2.2.1 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 (Reeves, 1990; Weber, 1995; Murawski, 1996; Rochet et al., 2002; Rochet and Trenkel, 2005; Borges et al., 2006; Feekings et al., 2012). 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), and may thus not be accurate (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). 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.

2.2.2.2 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 (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; Feekings et al., 2012).

2.2.2.3 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 (Figure 2; 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; Feekings 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). Feekings et al. (2012) 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).

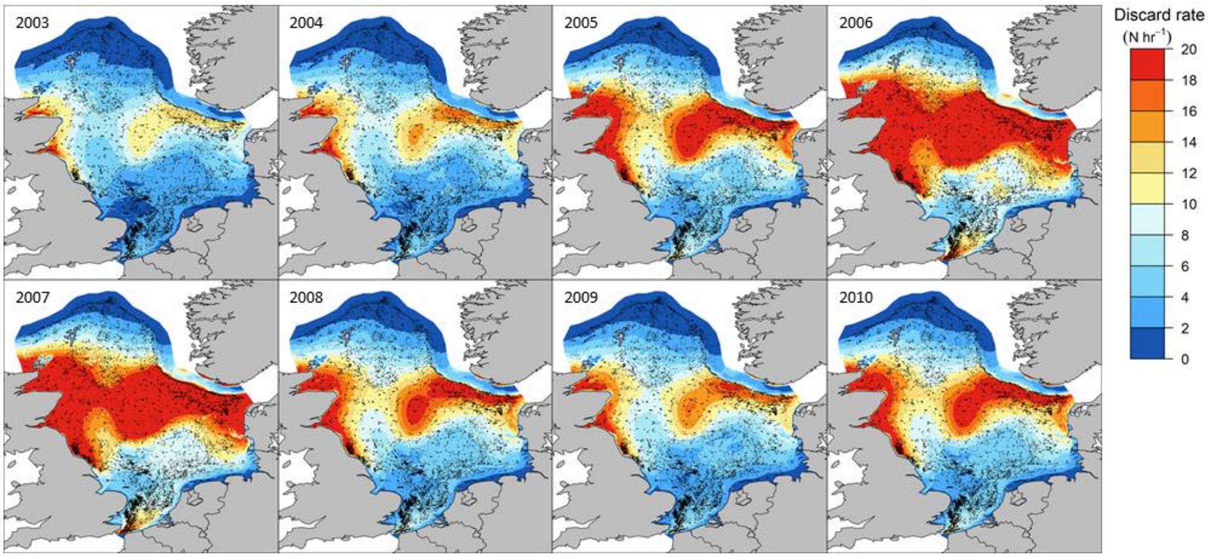

Figure 2: Spatiotemporal distribution of young cod discards throughout the North Sea 2003 – 2010. Predicted from Generalised Additive Mixed Model (Feekings, 2012).

2.2.3 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).

2.2.3.1 TAC/ quotas

Within the European Union, Total Allowable Catches (TAC) are defined for most commercial species. 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 has long been the case for North Sea cod (Ulrich et al., 2011), although the situation is improving now (ICES, 2015). 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).

2.2.3.2 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 (Figure 3). 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 (Feekings et al., 2013). 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 (Graham et al., 2007; Frandsen et al., 2009; Madsen and Valentinsson, 2010; Feekings et al., 2012). 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).

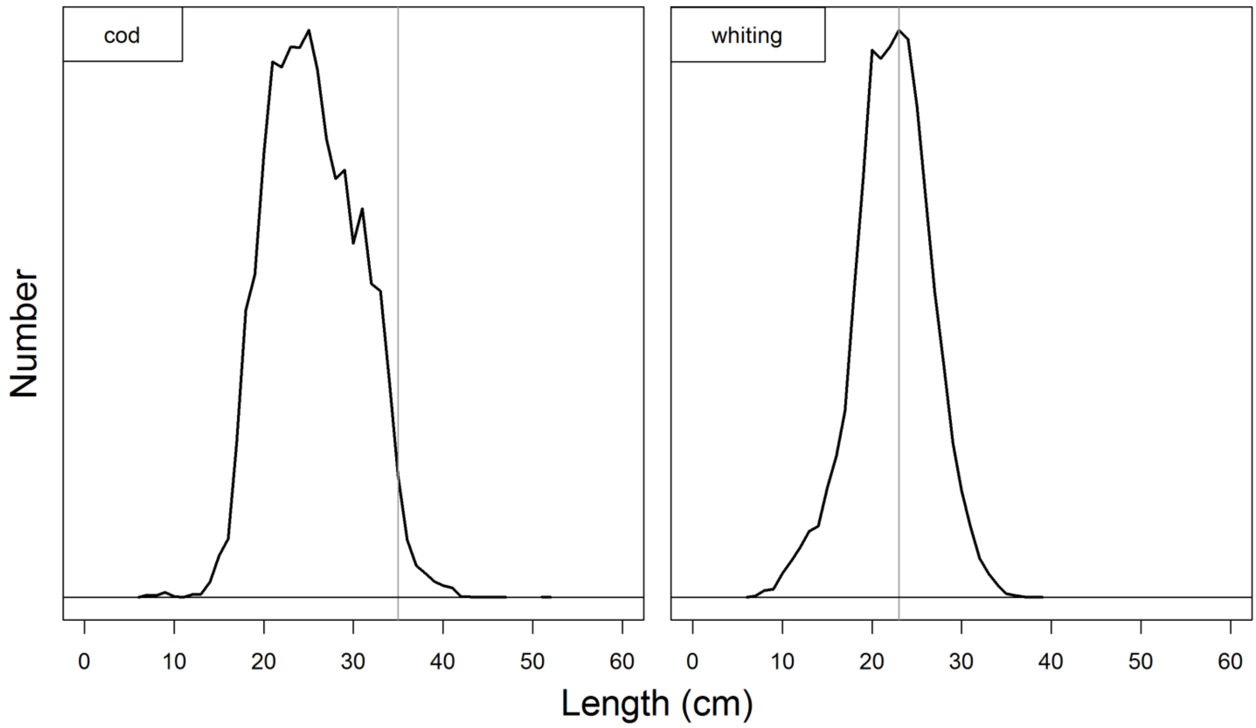

Figure 3: An example of the length compositions of the discarded portion of the catch for commercial (cod) and non-commercial (whiting) species in the Kattegat. Vertical grey lines represent the MLS for cod (35 cm) and whiting (23 cm) (Feekings, 2012).

2.2.3.3 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.

2.2.4 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 (see also DiscardLess Deliverables D2.1 and D2.2) ;

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 (Figure 4) 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 (Graham et al., 2007; Feekings et al., 2013).

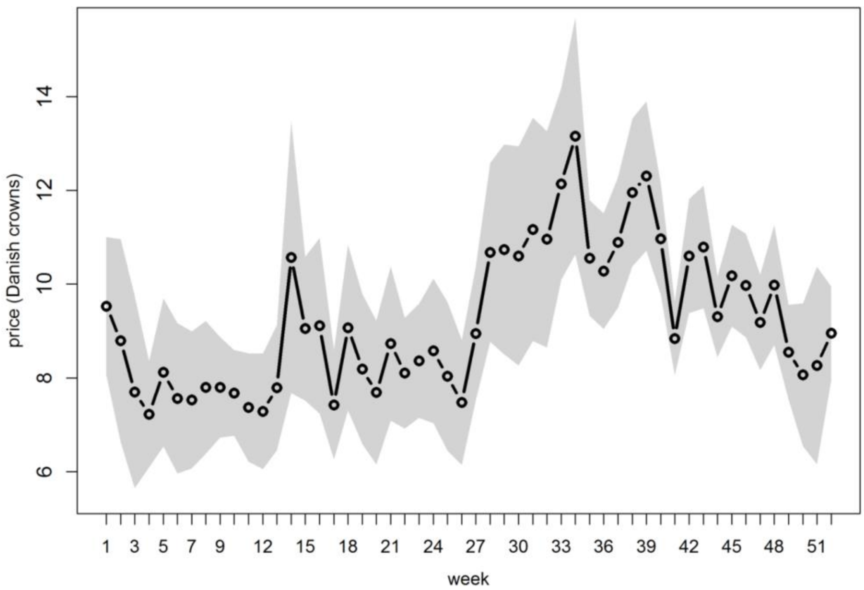

Figure 4: Variations in market price for cod in the Baltic Sea throughout the year. Grey band represents the variation (standard deviation) in price across ports (Feekings, 2012).

The economic analysis of discarding is further detailed in DiscardLess deliverable D2.1

2.2.5 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 (Feekings et al., 2013). 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).

2.2.6 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).

3 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 (see review and SWOT analysis in Sigurðardóttir et al., 2015). 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 either discussed or implemented within EU to develop discard free fisheries.

3.1 Discard rates

Most methods for reducing discards focus on discard rates and/or ratios. Here we defined discard rates as either the number or weight per unit effort (DPUE) while discard ratios as the proportion (percentage) of the total catch that is discarded. 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 (Feekings et al., 2013).

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. 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 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 their emergence with EU fisheries is increasing (Little et al., 2014).

3.1.1 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). Questions are though raised on the ecological impact of selective fishing in the long-term (see chapter on unselective fishing below). 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 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 (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 (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 applications 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, 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 often preferred in fisheries where the objective is to reduce cod bycatch while maintaining bycatches of commercially important flatfish species (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 (Madsen, 2007; Feekings et al., 2013) 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 (Wardle, 1993). For example, Krag et al. (2010) designed a selective trawl based on the behavioral differences between haddock and cod as they enter a trawl, i.e., cod stay close to the seabed whereas haddock rise above it.

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). Recent efforts have been made which attempt to catalogue the various modifications to trawls which can improve species and size selection (Seafish, 2015; Frandsen et al., 2015).

3.1.2 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 (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; Beare et al., 2013). Consequently, the use of closures is becoming less prescriptive and more incentive based (Graham et al., 2007, Kraak et al, 2015). 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.

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.

3.1.3 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, Little et al., 2014). 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 realtime 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, Little et al., 2014).

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).

3.1.4 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 (Kelleher, 2005; Feekings et al., 2013) 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.

3.2 Total discards

3.2.1 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.

3.3 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 fisheries and the complimentary management measures which can help achieve a sounder utilisation of fish resources.

3.3.1 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; see also DiscardLess deliverable D5.1). The European commission is following suit and has agreed on introducing a landings obligation, entailing that no fish of a regulated stock, be it above or below the MLS, can be thrown overboard. It is important to highlight that the landings obligation 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 should theoretically 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, if implemented together with a strong monitoring program, 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 (cf DiscardLess WP6). 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.

3.3.2 Catch Quota Management

The primary objectives of a Catch Quota Management (CQM) system 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, Ulrich et al., 2015). Under a CQM system all fish are counted against quota, regardless of size and marketability. 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).

3.3.3 Electronic monitoring

The success of a CQM system 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; Needle et al., 2014; Ulrich et al., 2015). 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, Ulrich et al., 2015). 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, Needle et al., 2014). 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, Needle et al., 2014). 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.

3.3.4 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; O'Neill et al., 2014). 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.

3.3.5 Stakeholder involvement

Incorporating stakeholders 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). Fishers are the most qualified people to develop and improve discard mitigation techniques (Gilman et al., 2006) and the earlier they 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. Their 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).

3.3.6 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). However, the question of the desirability and the feasibility of unselective fishing ('balance harvesting') is still a highly debated case (Burgess et al., 2015).

4 Effects 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).

4.1 Survival 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; STECF, 2013). 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, STECF, 2013) 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 before hauling should be promoted during fishing (Broadhurst et al., 2006). This is because the mortalities of discards are considerably greater than of 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 suite 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 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).

4.2 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.

These aspects are treated in more details in DiscardLess deliverables D2.1 and D2.2

4.3 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).

With the reform of the Common Fisheries Policy and the introduction of a Landing Obligation there is concern that the reduction in discards may lead to a food shortage for some scavenger species and possible shift in species compositions. The effect of a reduction in food for seabirds might be expected to lead to decreased populations of the species most dependent on discards such as large generalist seabird species (Bicknell et al., 2013). Furthermore, a reduction in discards has been shown to alter the ecosystem functioning of some seabird communities (e.g. increase predation on other species of birds; Votier et al., 2004). Considering that seabirds consume approximately 57-70% of discarded animals (Blaber and Wassenberg, 1989; Berghard and Rosner, 1992; Evans et al., 1994; Catchpole et al., 2006), the landing obligation is something which presents a potentially serious threat to some seabird communities.

In addition to the effects that a reduction in discards may have on seabird populations, the new policy may also have an impact on benthic and demersal species who consume discards on or near the seabed. In 2015, the ICES Working Group on the Ecosystem Effects of Fishing Activities (WGECO) was asked to address this issue: "Develop indicators of scavengers, examine their relation to discard amounts and evaluate the potential effect of a landing obligation on the benthic ecosystem". The increased focus on the effects of the landing obligation on scavengers is due to the potential creation of a food shortage for scavenger species that feed on discarded organisms. Furthermore, extracting additional biomass in the form of otherwise discarded organisms may have consequences for the populations from which they are extracted and secondary responses in the ecosystem (ICES, 2015).

The effect of this shortage depends on the ability of the scavengers to compensate by switching to other food sources and on the changes in conversion efficiency of their food. This may limit the direct effects on these species, but may also cause unpredictable cascading effects on other species through increased predation and/or competition. A wide range of other species has been identified to scavenge on discards, from marine mammals to benthos (Wassenberg and Hill, 1990; Svane et al., 2008).

Marine scavengers are defined as organisms which are able to detect carrion, usually by either distance or touch chemoreception, or both, deliberating to move toward it, and eventually consume either part or all of it (Britton and Morton, 1994). Scavenging can be considered as one form of feeding behaviour on a continuum which has fuzzy edges with other feeding behaviours such as predation and parasitism (Bengtson, 2002). Additionally, scavengers range from those that are close to obligate scavengers through to predators that will occasionally scavenge. Moleon et al. (2014) define a facultative scavenger as an animal "that scavenges at variable rates but that can subsist on other food resources in the absence of carrion", while an obligate scavenger as an animal "that relies entirely or near entirely on carrion as food resource". The animals that will be most affected by a landing obligation, and hence removal of discards, are likely to be those towards the obligate end of this continuum.

4.3.1 Identification of key scavengers

WGECO (2015) identified key scavengers from field studies which investigated the aggregation of organisms after presenting discards to them as bait. Scavenging organisms included species which were able to detect discards and move towards them for consumption. The key species were consequently mainly identified based on their numerical dominance and only indirectly based on their dependence on discards.

In European waters, most studies were conducted in the North Sea, the Irish Sea or the Clyde Sea. When considering the top five scavenging species based on the numbers attracted to sampling gear by discards, six taxa occurred in >2 studies: common whelk (*Buccinum undatum*), Hermit crab (*Pagurus bernhardus*), common sea star (*Asterias rubens*), Edible crab (*Cancer pagurus*), swimming crabs (*Liocarcinus sp.*), and common littoral crab (*Carcinus maenas*). The small number of *Cancer pagurus* is likely due to the sampling methods, as is the underrepresentation of the fish guilds (See methods section below). Several factors influenced the number of identified key scavengers. Background densities and their spatio-temporal variation are likely the most influential, but were not always registered.

Among fish species, the most obvious candidate would be the hagfish (https://www.youtube.com/watch?v=jz_mJ9AkTRo). In a baited camera study Martinez et al., (2011) found that hagfish (*Myxine glutinosa*) was the most abundant species attracted to bait. It should be noted however, that the other most common species were flatfish (mainly dabs *Limanda limanda*), whiting (*Merlangius merlangus*) and haddock (*Melanogramus aeglefinus*) emphasizing the continuum between predator and scavenger. In this context, even hagfish have been shown to occasionally act as predators (Zintzen et al., 2011).

Seasonal and diurnal feeding patterns may affect food partitioning. Ramsay et al., (1997) for instance illustrated that Liocarcinus sp. increase their scavenging activity during the night, while Nickell and Moore (1991) highlight that the monthly catch of Pandalina brevirostris and Ophiocomina nigra in the baited traps was correlated with variation in current speed over the spring/neap tidal cycle. Seasonal variation was not detected in Nickell and Moore (1991), but Groenewold and Fonds (2000) showed that the consumption rates varied due to temperature differences/seasons. Spatial variation in the segregation of food between species results from differences in scavenger assemblages by habitat type, and the resulting differences in competitive interactions (Ramsay et al., 1997). The spatio-temporal variability of species' distributions is some-thing which needs to be accounted for when developing indicators for species scavenging on discards.

4.3.2 Sampling methods

Several methods have been employed to study the consumption of discards on the seabed. In the Northeast Atlantic, baited traps have been use by a number of studies (Nickell and Moore, 1991; Moore and Howarth, 1996; Ramsay et al., 1997; Groenewold and Fonds, 2000; Bergmann et al., 2002; Castro et al., 2005; Catchpole et al., 2006). Baited cameras have also been used in a couple studies (Ramsay et al., 1997; Jenkins et al., 2004; Martinez et al., 2011), and, to a lesser extent, divers (Bergmann et al., 2002).

Several authors used a combination of Nephrops creels and funnel traps (Catchpole e et al., 2006; Nickell and Moore, 1991; Bergmann et al., 2002). Nickell and Moore (1991) mention that the use of baited traps or creels to catch commercially important scavengers is an established method of sampling smaller species (e.g. amphipods) for research (Forbes and Hanley, 1853; Holdsworth, 1874 in Edwards, 1979; Paul, 1973; Shulenberger and Barnard, 1976 ; Ingram and Hessler, 1983). Groenewold and Fonds (2000) initially tested 10 different types of traps, and concluded that transparent tube traps, Danish prawn traps, and small (transparent plastic) amphipod traps appeared to be most suitable (details in Groenewold, 1999; Lindeboom and de Groot, 1998).

In traps, the mesh size is the main factor in determining the abundance and diversity of species retained. Small-meshed funnel traps or amphipod traps usually retain the smaller scavengers (amphipods and isopods) while the larger meshed traps retain lager organisms (molluscs, crustaceans and in some cases, fish). This subdivision of species scavenging on discards may cause bias in their observed abundances and therefore it will be difficult to determine the relative importance of different scavenging size groups as consumers of discards.

Due to the small size of entrances often used in traps (25–70 mm) the abundance of larger and more mobile scavengers such as fish are most likely underestimated (Groenewold and Fonds 2000; Catchpole et al., 2006). Time-lapse camera observations can yield useful insights into on arrival times and residence time at food falls of these larger more mobile species (Kaiser and Hiddink, 2007) but they may not be optimal in obtaining accurate scavenger abundance of more mobile species. This is because time-lapse cameras only capture snippets of the feeding behaviour, thereby potentially missing the moments more mobile species are present. To obtain more accurate information on the scavenging behaviour of demersal fish Catchpole et al., (2006) examined the stomachs of demersal fish for evidence of discarded material (Wieczorek et al., 1999; ICES, 2000). With the introduction of cheaper underwater cameras the possibility to monitor the scavenging behaviour of demersal fish has increased.

4.3.3 DiscardLess sea trial in Kattegat

Due to technological advancements, observations of scavenging behaviour have become easier to obtain, especially for the more mobile species. As part of the DiscardLess and FP7 Benthis (www.benthis.eu) projects, DTU Aqua performed a sea trial outside of Varberg, Sweden in May 2015. A series of baited cameras were deployed to identify:

- Which benthic species scavenge on discards
- Scavenger succession. Who arrives first, who arrives last?
- Consumption rates. How much of the discard is eaten?
- Differences in species predating on discards across
    - Sediment type – muddy and rocky bottom
    - Species discarded

A string of 5 baited frames were attached 50 m apart (Figure 5). Each discard frame had a different type of discard (Nephrops, flatfish, round fish, creel bait, and a control to establish the background populations). The order of frames was randomised for each deployment. The stings were set perpendicular to the current to avoid bait plumes overlapping. Each frame was equipped with two GoPRO cameras. One camera was set to video mode to capture continuous footage for the first 2-4 hours; the other camera was set to time-lapse mode to capture an image every 5 minutes. The continuous footage was used to identify species, arrival times of different species (scavenger succession) while the time-lapse images were used to observe decay rates of the discards, as well as supplement species identification from the continuous footage over a longer time frame (~12-24 hours).

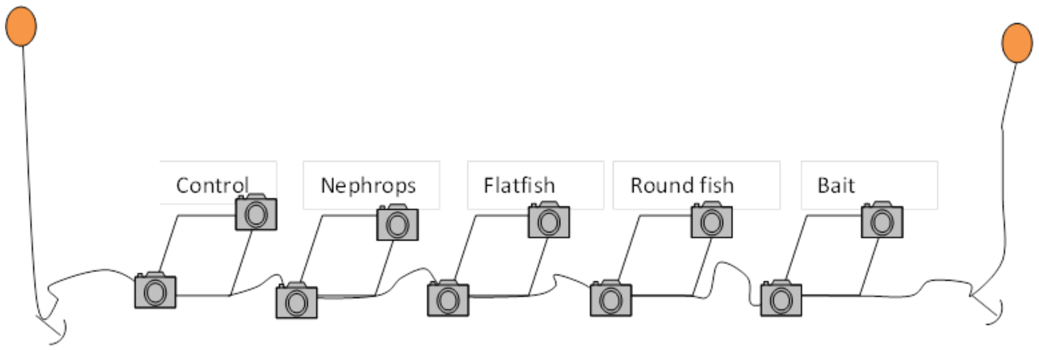

Figure 5: Experimental setup.

The data collected during the trial still needs to be thoroughly examined. However, preliminary analysis of the video observations showed that approximately 20 species were observed to be scavenging on the discards. These included wolffish (*Anarhichas lupus*; Figure 6), plaice (*Pleuronectes platessa*), dab (*limanda limanda*), whiting (*Merlangius merlangus*), Norway pout (*Trisopterus esmarkii*), hermit crab (*Pagurus bernhardus*), edible crab (*Cancer pagurus*), northern stone crab (Lithodes maja), brittle star (*Ophiuroidea spp.*), swimming crab (*Liocarcinus spp.*), whelks (*Neptunea antiqua* and *Buccinum undatum*), hagfish (*Myxine glutinosa*) and Norway lobster (*Nephrops norvegicus*).

Consumption rates were very dependent on the species discarded and the species scavenging. For example, flatfish species appeared to have difficulty penetrating the hard exoskeleton of Norway lobster while after the arrival of a wolfish there was not much left.

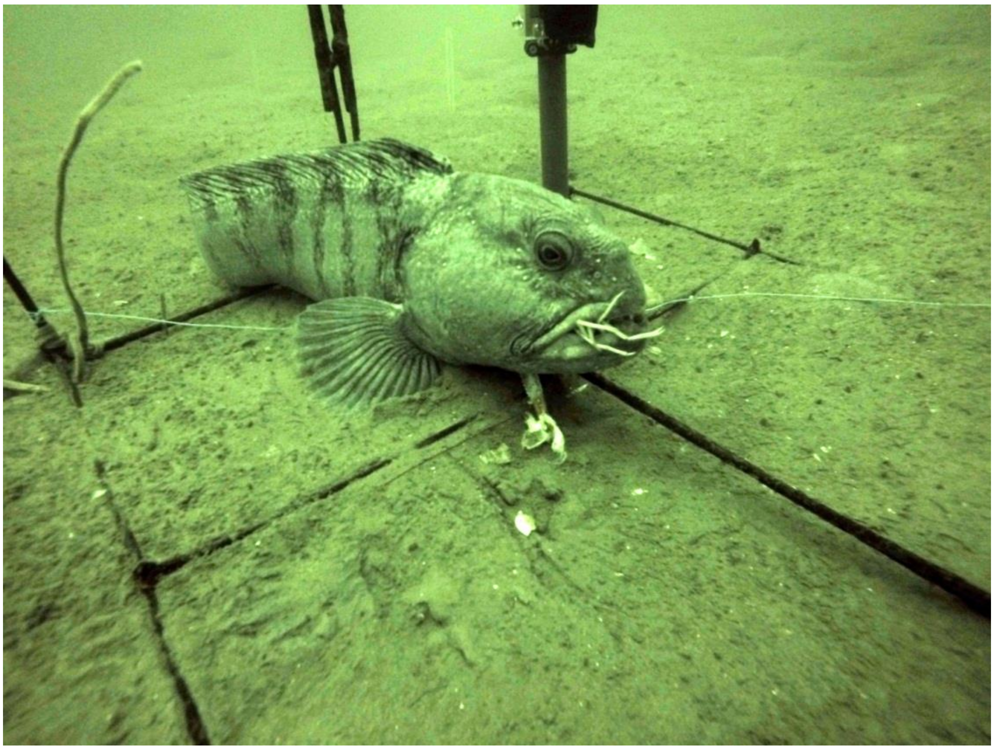

Figure 6: A time-lapse image from a baited frame of a wolfish (Anarhichas lupus) eating Norway lobster (Nephrops norvegicus).

5 The use of discard data in fish stock evaluation and management

5.1 The EU data collection framework and its use in ICES

Major progress has been achieved since the 2000s, in order to provide reliable discards estimates, and to account for them in management and advice. In EU the collection of discards data has been framed in the Data Collection Framework DCF (EU, 2008), and a number of bodies have been involved in coordinating the national sampling programs and sharing the data across countries. As such, fisheries data are considered both at the scale of the individual member state and at the regional scale across shared stocks. Much information regarding EU planning for data collection is gathered by the Joint Research Center (JRC) on http://datacollection.jrc.ec.europa.eu/. Progresses are also regularly reviewed by the STECF[1].

ICES is also actively involved in this process, being one of the most important end-users of data collected. ICES is now hosting the Regional DataBase FishFrame[2], which is a regionally coordinated database platform for fisheries assessments in the North Atlantic Ocean, the North Sea and the Baltic Sea. It supports the preparation and analysis of commercial catch data, mainly from EU countries on the basis of the DCF.

It must be kept in mind that the resources allocated by the EU and Member States into the monitoring and estimation of discards are important, but the number of trips that can be sampled by on-board observers remains limited because of the high cost of sampling programs. Hence, it is commonly admitted that less than 1% of the fishing trips are sampled. Therefore, the discard estimates used in the assessment and management advice for European stocks are considered as the best available knowledge, but they remain highly uncertain.

5.2 Extent of discards data included in assessment

This important effort has led to a continuous increase and improvement of the extent to which discards are now integrated in stock assessment in the ICES areas. The ICES database on stock assessments[3] records 78 stocks with analytical stocks, of which 26 had discards included in 2013 (Figure 7).

[1] http://stecf.jrc.ec.europa.eu/reports/dcf-dcr

[2] http://www.ices.dk/marine-data/data-portals/Pages/RDB-FishFrame.aspx

[3] http://www.ices.dk/marine-data/tools/Pages/stock-assessment-graphs.aspx. Extracted in August 2015, updated on 2014 assessments

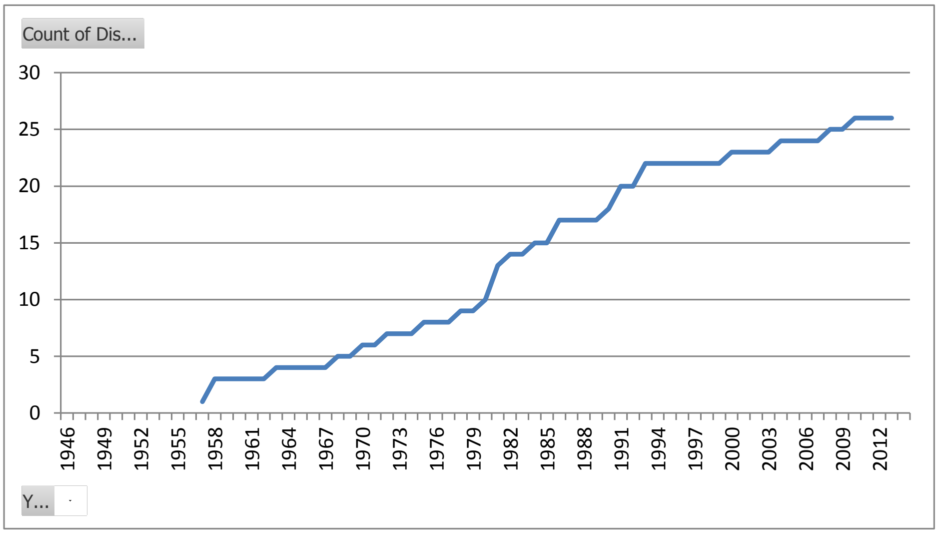

Figure 7: Number of stocks in ICES stock assessment database with discards tonnage available in the assessment.

Since 2013, ICES has generalised the basis of advice as being catch advice, by opposition to the previous landings advice. Advice sheets increasingly include a mention on discards estimates, at least qualitatively if not quantitatively. Quantitative estimates are either fully included in the assessment as explained above, if the time series is long and reliable enough, or used as a "top-up", where a recent estimate of the discarded proportion (usually a 3 year average) is added on the landings advice. If no quantitative estimate is available, mention is made of the best available knowledge, including the expected range of magnitude of the discarding. However, in 2015 this was still not generalised to all stocks yet. A overview made by the ICES secretariat in August 2015 indicated that out of 263 ICES stocks with an advice produced in 2014, 40 stocks had discards used in the assessment (increased to 47 stocks in 2015), 98 had discards characterised as negligible, and 108 had discards characterised as unknown or unquantified. The remaining 17 stocks had no mention of discards.

In 2015, ICES abandoned the wording of "landings" vs "discards", and classified the catches into equivalent categories of "wanted catches" vs. "unwanted catches".

An important aspect is that global discard information on EU Atlantic stocks is now available from two data sources, ICES and STECF-JRC. While both databases use the same raw data (DCF national sampling programs), they may nevertheless provide different discards estimates, as data are combined and raised differently, according to different strata and for different purposes.

In the context of the landings obligation, ICES and STECF made therefore the specific exercise of gathering and comparing their discard data used for assessment and management advice (STECF, 2013)[4]. A total of 85 stocks were identified by the Commission. Landings, Discards and Catch data were extracted from both the ICES and the JRC databases for all stocks, where available.

Based on the STECF data and ICES information, the 85 stocks were classified into three groups:-

[4] http://stecf.jrc.ec.europa.eu/documents/43805/610582/2013-11_STECF+13-23+-+Landing+obligation+in+EU+Fisheries-part1_JRC86112.pdf

Group I. 34 stocks which ICES indicates that discarding is considered negligible and STECF estimates that discarding is less than 10%.

Group II. 23 stocks for which detailed data on catch is available from both ICES and STECF

Group III. The remaining 28 stocks for which either ICES or STECF indicate that significant (>10%) discarding occurs but ICES did not present discard data in the advice sheets in 2013.

Group I. The 34 stocks are stocks with known or assumed low discards rates, mainly following species: anchovy, anglerfish, blue whiting, herring, horse mackerel, ling, mackerel, Nephrops, sole and Norway pout.

Group II. Included 23 stocks for which ICES publishes discard data in its advice and data is available in the STECF database. These stocks are mainly from following species: cod, haddock, hake, megrim, plaice, salmon and whiting.

For a number of these stocks, significant discrepancies were observed between ICES and STECF estimates of discards ratios for the period 2009-2012. The main reasons for such discrepancies were summarized as follows.

- Different methods are used from ICES and STECF in order to raise discard estimates when no information is available from a Member State.

- Several inconsistencies can be found in the management areas defined in the two datasets due to ICES practice of moving catch to better link area to stock and STECF area specification from DCF.

- For some stocks the ICES expert working groups are using official landings declarations considering this information as reliable to perform the assessment. In other cases, the landings figures are raised based on the experts' knowledge of the stock by adding unallocated values to obtain the so called ICES landings level. These ICES figures are used to in the stock assessment process. STECF uses only official submitted landings but does not carry out stock assessment.

- Only EU Members States are obliged to submit data to STECF whereas Norway, Faroes, Iceland and other countries provide data to ICES and can be a major contributor in some stock catches. Discarding ratios are different for some of these countries that do not submit data to STECF.

- Some discards information from some Member States in some years has been submitted to only one of the two datasets, but not both.

Nevertheless, STECF (2013) observed increased convergence between the two datasets over time. In 2012, differences between the estimated discards ratios were below 20% for all stocks, and below 10% for 16 stocks out of the 23.

Group III. For the remaining 28 stocks without discards included in the advice in 2013, STECF (2013) did not formulate generic conclusions, but drew ICES's attention on the availability on alternative discards estimates**.**

In the Mediterranean, the inclusion of discards data in the assessment and advice is done on a case-by-case basis. For several stocks, discards are included in the catch at age matrix, but for the others the advice is based on landings only, without a discards top-up.

6 Ecosystem modelling and the incorporation of discard-related processes into ecosystem models

The increasing demand for tools supporting ecosystem based management in the marine realm has been a major driver in the development of end to end modelling platforms (e.g. Rose et al 2010, Plaganyi, 2007). End-to-end models typically combine submodels of physicochemical oceanographic processes with descriptors of lower and higher-trophic-level organisms into a single modelling framework (Travers et al. 2009).They include important feedbacks among these three factors. End-to-end models also aim at including humans as members of the high trophic level community that react and adjust to changing conditions (Rose et al, 2010).

In the context of the Discardless project, such tools will allow us to study the interactions between i) the effects of the removal of the flow of discards to the ecosystem and ii) the effects on fish stocks of new fishing strategies and MSY-based exploitation patterns, in an integrated fashion. There are currently several established end to end models : Osmose, Atlantis, Ecospace (spatial version of the Ecopath with Ecosim, i.e. EwE), and the more recent StrathE2E. The four of them will be used in the Discardless project (although the EwE models will not all have the spatial/physical component developed in Ecospace).

The use of the deterministic fisheries dynamic simulation model ISIS-Fish in two of our case studies will also allow us to explore more specifically the spatial interaction between key fish stocks and their fishing fleets in the context of the LO implementation.

All of these models are presented below in further details with a special emphasis on how the discarding processes are modelled.

6.1 Ecopath with Ecosim (EwE)

6.1.1 Model description

Ecopath with Ecosim (EwE) is a food-web modelling facility that can be used to build trophic static mass-balanced snapshots (Ecopath) and to create temporal dynamic (Ecosim) of an ecosystem (Christensen and Pauly, 1992; Walters et al., 1997; Pauly et al., 2000; Walters et al., 2000; Christensen and Walters, 2004). EwE has been widely adopted all over the world and has led to some ground breaking ideas and results (Pauly and Christensen, 1995; Pauly et al., 1998; Watson and Pauly, 2001; Branch et al., 2010; Smith et al., 2011, Irigoien et al., 2014). However, some common mistake in EwE modeling have been identified and may be given due consideration (Ainsworth and Walters, 2015).

The basis for the parameterization of Ecopath models relies on two master equations, one describing the production term and the other the energy balance for each functional group (i.e. a defined biomass pool). The first master equation ensures a mass balance between groups and expresses production as a function of the catch, predation, net migration, biomass accumulation and other mortality (Eq. 1). The second master equation is based on the principle of conservation of matter within each group (Eq. 2), similar to the energy balance used in bioenergetics models (Christensen and Walters, 2004). Each group is parameterised with its biomass (B, t km^{-2}), production rate (P/B, yr^{-1}), consumption rate

(Q/B, yr^{-1}), the prey-predator interaction in the form of a diet composition (DC) table, ecotrophic efficiency (EE_i), the biomass accumulation rate (BA_i, yr^{-1}) and the net migration rate (E_i, yr^{-1}).

$$\boldsymbol{B\left(\frac{P}{B}\right)_i = Y_i + \sum_j B_j \left(\frac{Q}{B}\right)_j DC_{ij} + E_i\, BA_i + B_i \left(\frac{P}{B}\right)_i (1 - EE_i)} \quad (1)$$

$$\boldsymbol{Consumption\ (Q_i) = production\ (P_i) + respiration\ (Ri) + unassimilated\ food\ (Ui)} \quad (2)$$

Inherently being a static approach, Ecopath cannot simulate ecosystem changes over time. Therefore, the temporal dynamics of the ecosystem was assessed using Ecosim (Walters et al., 1997; Walters et al., 2000; Christensen and Walters, 2004). Ecosim calculates the Ecopath parameters for each time-step, taking into account changes in fishing effort, environmental factors or other parameters. Ecosim allows for the fitting of predicted biomasses to time series data. The resulting coupled differential equations used in Ecosim exist in the form of (Eq. 3)

$$\mathrm{dB_i/dt} = \mathrm{g_i} \sum\nolimits_j \mathrm{Q_{ji}} - \sum\nolimits_j \mathrm{Q_{ij}} + \mathrm{I_i} - (\mathrm{M0_i} + \mathrm{F_i} + \mathrm{e_i}), \quad (3)$$

where dBi/dt is the rate of change in biomass of group i during interval dt, g is the net growth efficiency, Fi is the fishing mortality rate, M0i is the natural mortality rate (excluding predation), ei is the emigration rate and Ii is the immigration rate.

The Q term refer to consumption by group i (Qji) and predation on i (Qij), and are calculated using the 'forage arena' concept where prey behaviour affects predation rates (Walters and Martell, 2004; Ahrens et al., 2012). Each prey group in each predator-prey interaction is divided into readily available (or vulnerable) and unavailable (or invulnerable) for predators. Therefore, the vulnerabilities of each prey to its predators are one of the key parameters of the foraging arena equations and heavily influence the behavior of the model (Ahrens et al., 2012). A low vulnerability indicates a bottom-up control while a high vulnerability indicates top-down Lotka-Volterra like control. Although intuitive, these parameters are still hard to estimate and validate.

A critical first step in the model construction consisted of grouping the species present in the ecosystem into functional groups (defined biomass pools), essentially biologically and ecologically defined groups with similarities in, amongst others, size, feeding habits and habitat. Model parameters, P/B, Q/B and P/Q can be estimated using empirical equations (Pauly, 1980; Palomares and Pauly, 1998; Ainsworth et al., 2006), or taken from literature, with preference to studies within model area or from similar areas. The model pedigree describing the origin and quality of each parameter can be documented and used to assign confidence intervals to the data using a sensitivity analysis (Pauly, 2000). A diet matrix has to be assembled using preferentially local literature on stomach content analyses, completed with other literature and adapted using empirical knowledge.

EwE has been recently used to assess the ecosystem status in support of the European Marine Strategy Framework Directive (eg. Heymans et al., 2011).

6.1.2 Implementations in Discardless

EwE models will be used in 5 of our Case studies: the Azores (Morato et al., submitted), the Eastern Mediterranean (the North Aegean Sea, Tsagarakis et al 2010), the Western Mediterranean (in the Balearic islands, Moranta et al. 2014), the Bay of Biscay (Lassalle et al, 2011), and the Celtic Sea (Lauria, 2012). These models are described in the attached factsheets.

6.1.3 Modelling of the discarding processes and of the fate of discards in EwE

Modelling the discarding processes and the fate of discards in the Ecopath with Ecosim (EwE) modelling approach is feasible since both landings and discards volumes can be specified in the baseline model (Ecopath), which allows the calculation of a total fishing mortality. The discards fate can then be directed towards a generic detritus functional group or a specific discards functional group, in a proportion regulated by the parameter "detritus fate" (0: all exported out of the system, 1 all flowed to the detritus group).

In the temporal dynamic module of EwE, Ecosim, a total fishing mortality per fleet is calculated when the model is driven by fishing effort or fishing mortality time series. This fishing mortality represents both landing and discards. Thus, in the dynamic part, the mortalities caused by landing and discards cannot be distinguished. The proportion of the total catch (resulting from the fishing mortality or fishing effort time series applied) going into the discards functional group or detritus group is then the same as in the Ecopath version of the model (where landings and discards were specified). This proportion is kept constant in the present released version of the software.

The discards groups in the Ecopath model can be consumed by any other group in the model and vulnerabilities to the different consumer groups to the discards can be specified as for any prey group.

At this stage, the modeling of discards in the EwE framework is limited because: (1) There is only a rudimentary way to change the discard survival rate in the static Ecopath model and this is not possible to change in the dynamic configuration; and (2) the proportion of the catch being discarded is constant in the dynamic configuration. A possibility to address the second issue is to separate each fleet in 2: one with a fishing mortality corresponding to the landings, and the other to the discards. However, this solution is limited in the way the ecosystem dynamics will be established and how the fleet dynamics will be captured. Additionally, there is no option to force discarding time series in Ecosim if discards time series are known. Potential developments of the software to overcome the identified limitations are feasible and are also under consideration. It is also worth noting that limited discard policy alternative calculations are already built into the Ecosim Management Strategy Evaluation (MSE) engine developed by CEFAS, and should not be too difficult to expose and properly integrate into the Ecosim user interfaces and computations in the future.

6.2 OSMOSE

6.2.1 Model description

OSMOSE (Object-oriented Simulator of Marine ecOSystems Exploitation model) is an individual-based model (IBM) which focuses on the dynamics of the fish community (Shin and Cury 2001, 2004; Shin et al., 2004). It represents fish individuals grouped into schools, which are characterized by their size, weight, age, taxonomy and geographical location (2D model), and which undergo major processes of fish life cycle (growth, reproduction, migration and mortality from predation, natural and starvation) and a fishing mortality distinct for each species. The central hypothesis of the model is that predation is opportunistic, based on spatial co-occurrence and size adequacy between a predator and its prey.The success of predation (amount of food eaten compared to food requirement) defines the amplitude of growth of fish, resulting in individual variability in size among a cohort due to feeding past The model needs basic parameters that are often available for a wide range of species, and which can be found in FishBase (e.g. von Bertalanffy parameters, condition factors, relative fecundity). Some parameters describing the predation process can be considered as fixed for all species (e.g., maximal ingestion rate, min/maximum thresholds for predator/prey size ratios). Whereas the initial version of the model used a carrying capacity for fish community (Shin et al. 2004), later developments allow a forcing of fish production by satellite-derived plankton data (Marzloff et al. 2009) or an explicit coupling with biogeochemical plankton model (Travers et al. 2009). The coupling process used to link OSMOSE to LTL (low trophic level) models (e.g. NPZD, BFM, ERSEM) is the predation process (Travers et al., 2009): the LTL model is used as a prey field for the HTL model (usually concentration of nitrogen/carbon converted into wet biomass) and the HTL model provides a field of predation mortality rate for the LTL model. Model outputs can be stored at the individual level, allowing to compute indicators (size-based, species-based and trophodynamics indicators) at different levels of aggregation: at the species level (mean size, mean size-at-age, max size, trophic level), and at the community level (slope and intercept of size spectrum, Shannon diversity index), in the catches and in the community (i.e. to be compared with sea survey data), spatially integrated or not. OSMOSE can be used to run scenarios of fishing pressure (including search for FMSY, no-take marine areas) and climate variation (via the LTL model), investigating the trophic functioning of the ecosystem and how indicators can be used to track ecosystem response to different pressure.

6.2.2 Implementations in Discardless

Osmose was implemented on the Eastern Channel area by explicitely modelling 14 fish species forced by phytoplankton and zooplankton fields modelled via the biogeochemical model ECOMARS-3D. This application is used to investigate how fishing and climate change affect the Eastern Channel ecosystem. More details on the model are provided in the Eastern Channel Factsheet.

6.2.3 Modelling of the discarding processes and of the fate of discards

In the current version of OSMOSE, fishing is modelled via a survival equation involving a fishing mortality (input parameter) applied to recruited fish (i.e. older than the recruitment age set as input parameter), resulting in a global catch biomass and abundance per species, without discard specification. Within the DiscardLess project, a PhD thesis has been initiated (from 2015 to 2017) to include a fisherman module to the OSMOSE model. This module would allow the representation of more details regarding how the fishing activity is modelled, including the specification of gears and the consideration of discards. As the model is size-based, the discard process will at least include discards of small-sized fish and possibly discard of fish of low market value. Details of how the discard process is modelled and implemented will be finalized in 2016.

6.3 StrathE2E

6.3.1 Model description

StrathE2E is a coupled ordinary differential equation (ODE) model of a marine food web, embedded in a geochemical and hydrodynamic representation of a regional sea domain (Heath, 2012; Heath et al, 2014a, Heath et al, 2014b). The currency of the model is molar nitrogen units. State variables represent the nitrogen mass of dissolved inorganic nutrients (nitrate and ammonia) in water column layers and sediment pore waters, various detritus categories including fishery discards, and all the living benthic and pelagic components of a shelf-sea food web grouped according to coarse feeding guilds, from phytoplankton to birds and mammals (Figure 8). The network of ODEs is solved using adaptive time stepping and the mass and fluxes between state variables are output at daily intervals over an unlimited number of simulation years. The equations are coded to represent geochemical processes, feeding uptakes, excretion, and advection and diffusion fluxes due to time varying natural oceanographic, environmental and anthropogenic factors. The model explicitly includes demersal and pelagic fisheries, and is particularly suitable for simulating the top-down and bottom-up cascading effects of changes in human activity such as harvesting rates, or environmental factors such as nutrient emissions.

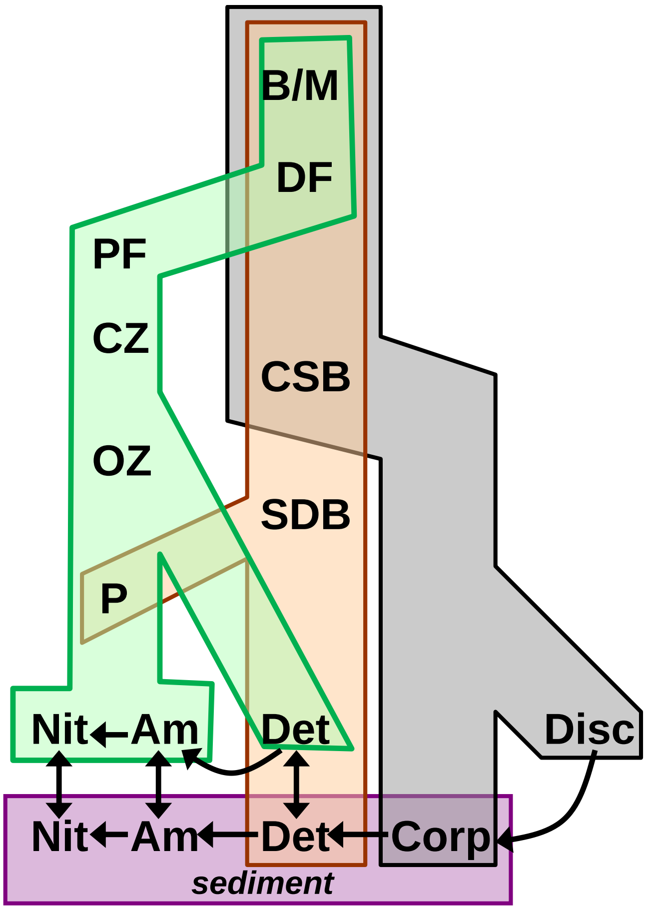

Figure 8: Simplified schematic of the food web compartments of the StrathE2E model. Taxa and non-living resources in the model form three interlinked food chain compartments: grey - scavenging; orange - benthic; green - pelagic. The purple compartment represents seabed sediment geochemistry. B/M - seabirds and marine mammals; DF - demersal fish (e.g. cod, haddock and plaice which feed mainly on other fish and benthos); PF - pelagic fish (e.g. herring, sprat and sandeel which feed mainly on plankton); CZ - carnivorous zooplankton; OZ - omnivorous zooplankton; P- phytoplankton; CSB - carnivorous/scavenging benthos; SDB - suspension/deposit feeding benthos; Nit - nitrate, Am- ammonia; Corp- corpses; Disc- fishery discards. Omnivory occurs within each compartment e.g. PF feed on both CZ and OZ; DF and PF are subdivided into larvae and adults; Nit, Am, Det and P in the water column are subdivided into surface and deep layers. Transformations between Disc, Corp, Det, Am and Nit are due to microbial degradation, mineralisation and nitrification processes. Fishery landings and denitrification represent export fluxes from the model, Water column classes of P, Nit, Am and Det are subject to hydrodynamic exchanges which generate net imports and exports depending on simulaed concentration gradients. The model also includes fluxes from living components to Am, Det, Corp and Disc due to excretion, defecation, death.

The model represents regionally averaged properties at scales such as, for example, the whole North Sea, and requires data on bathymetry, and at least monthly resolved time series of driving data including temperature, sea surface irradiance, turbidity, river inflows, and water transport and mixing rates derived from ocean circulation models. The model has been incorporated into a statistical fitting scheme (simulated annealing) which uses MCMC methodology to explore the parameter space by conducting thousands of simulation runs whilst randomly adjusting the parameter values, to seek the combination which provides the best match between the model results and assembled observations. These observations may include: nutrient concentrations, zooplankton abundances, fish and benthos survey data, fishery landings and discards, and bird and mammal diet compositions. Together these define the state of the ecosystem during the period corresponding to the environmental and human activity driving conditions applied to the model.

The model may be criticised for representing coarse groups of taxa rather than individual species, and only simulating dynamics at a coarse spatial scale. However, this is a conscious decision in order to produce fast computational speed (<1 sec per simulation year on a standard PC) and a minimal number of model parameters. Speed and minimal parameter count are key to enabling statistical fitting, which is lacking in the majority of other marine food web models. The ability to identify the best-fit parameter set yields a model producing results which are "as good as they can possibly be". The thesis is that coarse-scale results of this quality are preferable to more highly resolved outputs with little or no idea of how close they are to reality.

Existing implementations of this model are available for the North Sea (ICES area IV), the west of Scotland (ICES area VIa), and the Celtic Sea (ICES areas VII).

6.3.2 Implementations in Discardless

The simulated annealing procedure has been applied to a North Sea version of the model corresponding to ICES area IV. This has been peer reviewed and published (Heath 2012), and can be used online (http://www.mathstat.strath.ac.uk/outreach/e2e/). More details are provided in the North Sea factsheet.

6.3.3 Modelling of the discarding processes and of the fate of discards

In StrathE2E, catches of pelagic and demersal fish, and two classes of benthos are calculated from harvest ratios applied to the respective stock biomasses. Harvest ratios are derived from fleet-specific activity rates and catchability terms.

A proportion of the simulated catches of fish and benthos by each fleet is treated as 'landings' and is removed as an export flux from the model. The remainder of the catch is treated as discards, and this includes a) accidental or intentional (due to quota restrictions) spillage of marketable targeted catch from nets during gear recovery, b) throwing overboard of dead biomass of un-marketable species, and under-size or low value individuals of otherwise marketable species, and c) offal removed from the fish during gutting operations which is thrown overboard. In addition, though not normally regarded as a discard, fish which escape through net meshes but are damaged and do not survive, are functionally equivalent to discards.
The proportion of catches of pelagic fish and benthos classes which is discarded is a parameter in the model. However, for demersal fish the proportion discarded is parameterized as a function of adult demersal biomass to caricature the observed reduction in proportion of large fish in the stock, and hence in the catches, of groundfish communities. Discarded fish and benthos are all assumed dead for now, but a survival rate can easily be implemented.
Discards of fish and benthos form a potential food resource for demersal fish, birds and mammals in the model. They are also converted to corpses at a fixed daily rate representing settlement to the seabed, where they additionally form a food resource for scavenge-feeding benthos. As corpses, they slowly decay to detritus which is remineralised to ammonia, so that their nutrient content is recycled through the geochemical processes in the model.

6.4 Atlantis

6.4.1 Model description

Atlantis is a modelling framework intended for use in management strategy evaluation (MSE) studies (Fulton et al 2011 and ref. in). It therefore includes representations of each significant component of the adaptive management cycle, including the biophysical system, the human users of the system (industry), the three major components of an adaptive management strategy (monitoring, assessment and management decision processes) and socioeconomic drivers of human use and behaviour. Atlantis includes dynamic, two-way coupling of all these system components. The choice of formulation is an application-specific decision made by the user, who has the freedom to set complexity at any desired level. This can range from a small number of groups with simple trophic interactions and a Baranov catch equation to highly complex models with sophisticated stock structure, multiple fleets, detailed social and economic effort drivers and multiple management options. Below are some details about the modules used within this project, i.e. the biophysical and the industry and socio-economic modules.

The Atlantis biophysical submodel is a deterministic, spatially resolved, three-dimensional model, which is based on a system of irregular spatial polygons. This box representation facilitates tracking the flows of limiting nutrients (typically nitrogen and silica) through the main biological groups in the system (as defined by the user), with the system of differential equations typically solved on 12- or 24-h time steps (with finer adaptive substeps for high turn-over rate groups like plankton) using a simple forward difference integration scheme. The primary ecological processes modelled are consumption, production, waste production, movement and migration, predation, recruitment, habitat dependency and mortality. Ecological components are represented as either biomass pools (which are largely used for the lower trophic levels) or age-structure populations (typically for vertebrates) where the average size and condition of individuals in each age class are tracked in each box. Representation of the physical environment occurs within the polygonal boxes, matched to the major geographical and bioregional features of the marine system, coupled with an oceanographic transport model. Seabed type (proportions of soft, rough and flat) and features such as canyons are represented in each box, as well as the vertical temperature, salinity, pH and oxygen profiles, advective and diffusive flows and influence of eddies. The biological components may inhabit the substrate or any vertical layer of the water column according to environmental preferences.

The human impacts submodel deals primarily with the dynamics of fishing fleets – allowing for multiple fleets, each with its own characteristics (including gear selectivity, habitat association, targeting, effort allocation and management structures). The fleet dynamics model can be tailored to each fleet using formulations ranging from simple catch equations to forced effort, or catches, through to a quasi-agent- based approach. In the latter, subfleets (boats of similar size with common home ports, socioeconomic backgrounds or other aggregate behavioural feature) explicitly step through effort allocation decisions based on a memory of past conditions, current economic conditions, distance to fishing grounds, management regulations and social networks. The more complex variants can include explicit handling of taxes, markets, compliance decisions, exploratory fishing, fuel prices, employment, learning, information sharing, quota trading and investment/disinvestment.

There are currently 18 Atlantis models in use and more than 30 others under development across a range of scales and ecosystem types (Baltic Sea, North Sea, Strait of Sicily, Iceland, Great Lakes, Lake Victoria, Great Barrier Reef, Juan Fernandez Archipelago, Antarctica etc.) (http://atlantis.cmar.csiro.au/).

6.4.2 Implementations in Discardless

Atlantis was implemented on the Eastern Channel area (Girardin, 2015) to investigate the dynamics of the key Eastern English Channel ecosystem processes, with a particular focus on two commercial flatfish species, sole (*Solea solea*) and plaice (*Pleuronectes platessa*). More details on the model are provided in the Eastern Channel Factsheet.

6.4.3 Modelling of the discarding processes and of the fate of discards

In Atlantis, many options are available for the simulation of fishing. In the Eastern Channel Atlantis, the amount of catch is controlled by the interaction between the stocks (distribution, abundance and age/size distribution) and the fleets (effort, selectivity). The proportion of the catch that is discarded can then be computed in different ways, and a different setting can be selected for each stock/fishery couple. It can be a fixed proportion (per age class, or for all age classes under a certain size) of the catch (for each fishery and each stock). For stocks under quota, highgrading can be triggered when the cumulated landings are getting close to the quota (the threshold needs to be specified), and discarding of the whole catch can be triggered when the quota is reached.

A survival rate for discarded fish and invertebrates can be applied. The dead part of the discarded fish and invertebrates feeds into a "carrion" group which can be consumed by other groups and decay to detritus which is remineralised to ammonia, so that their nutrient content is recycled through the geochemical processes in the model.

6.5 ISIS-Fish

6.5.1 Model description

ISIS-Fish is a deterministic fisheries dynamic simulation model designed to investigate the consequences of alternative policies on the dynamics of resources and fleets for fisheries with mixed-species harvests (Mahévas and Pelletier 2004; Pelletier et al, 2009). It allows quantitative policy screening of combined management options, such as total allowable catch (TAC), effort control, licenses, gear restrictions, MPA, etc. Fishing mortality is the result of the interaction between the spatial distribution of population abundance resulting from the population submodel and the spatial distribution of fishing effort provided by the exploitation and management submodels at a monthly time-step (Figure 9). Fishing effort is standardized per métier and fleet according to gear selectivity and efficiency, ability to specifically target a species and technical efficiency. The effect of management measures can therefore be explicitly modelled either through modifications of the standardisation parameters for technical measures (e.g. change in the selectivity curve) or through modification of the level and spatio-temporal distribution of fishing time for seasonal closures or effort control for instance. Fisher's response to management may be accounted for by means of decision rules conditioned on population and exploitation variables or explicit dynamic model with endogenous (e.g. fish prices and variable costs) or exogenous variables. Discarding behaviour is implemented through decision rules (by default, as the consequence of catches under legal size or TAC reaching). The model is flexible in its spatial resolution and level of complexity to accommodate the specificities of mixed fisheries. It has been applied to the Bay of Biscay hake fishery (Drouineau et al., 2006) and pelagic fishery (Lehuta et al 2010, Lehuta et al 2013a, Lehuta et al 2013b), the European deep sea fishery (Marchal and Vermard 2013), the New Zealand Hoki fishery (Marchal et al 2009), the Tasmanian costal mixed fishery (Ziegler et al 2013), the cod fishery in the Baltic sea(Kraus et al 2008), and Mediteranean fisheries (Hussein et al 2011a, 2011b).

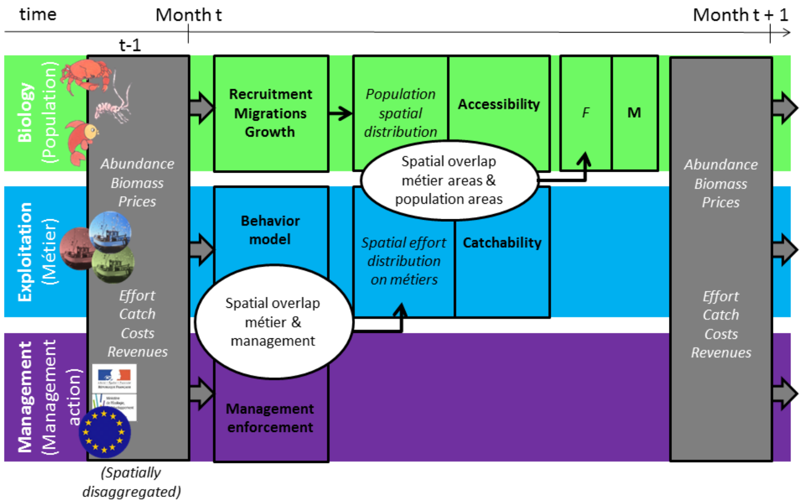

Figure 9: Flowchart of the events which occur at each time step in each sub-model (for each population, métier and management rule) and how they interact. Bolded words stand for processes and parameters, Italic represents dynamic state variables.

6.5.2 Implementation in Discardless

ISIS-Fish is implemented on the Eastern Channel area (Lehuta et al, 2015), and is being implemented in the Gulf of Lion. More details on these implementations are provided in the Eastern Channel and Western Mediterranean factsheets.

6.5.3 Modelling of the discarding processes and of the fate of discards

In ISIS-Fish, the amount of catch is determined by the age/length- , fleet-, area- specific fishing mortality. The separation of the catch between landings and discards is computed afterward according to explicit and flexible decision rules. In the current version of the model, in the **status quo** simulations, discards occur if:

- The **Quota** for a species is reached: Catches cumulate monthly in course of the year until the TAC of a species is reached. Thereafter the métier can still be practiced but the species which TAC is exhausted is totally discarded. Since the revenues of a métier are computed based on the landed quantities only, they can potentially be significantly impaired. The gravity model that drives fishermen behavior is then expected to redirect fishermen toward more profitable métiers.
- Fish under **minimum landing size** (equivalent to a minimum age (in month) given the model deterministic hypotheses on growth) are caught. For now, the model assumes a strict size threshold for discard. Data analyses are expected to allow challenging the current hypothesis with a distribution of discards across sizes to reflect both the diversity of reasons for discarding and the heterogeneity of size at age, especially for plaice in the case of the Eastern Channel implementation.

Under **landing obligation** the assumptions are changed:

- When the **TAC** is reached, the attractivity of all métiers catching the species is set to zero to reflect the impossibility to catch the species anymore. Exemptions can take place here, depending on the fleet or the métier or as a function of internal variables.
- If fish under **minimum conservation size** are caught they are landed but their price is set to zero to reflect the absence of commercialization opportunities. The gravity model as implemented, accounts for these extra non-commercial landings, and decreases the attractivity of the métier in consequence.

Other rules, such as highgrading behavior or fixed proportion of discards relative to catch could easily be implemented if needed.

An age-dependent survival rate for discarded fish can possibly be applied, when available (for now, the model assumes 0 for all species but scallops, for which the survival rate is 1). Given that there is no trophic relationship in the model, the dead part of the discards is not considered in the fishery dynamics any longer.

7 Conclusions

This extensive literature review has illustrated the complexity of the discard issue. Discard arise from complex interactions between numerous biological, ecological, economic, technical and regulatory considerations. As such, a great number of scientific disciplines within the broader field of marine and fisheries science can contribute with knowledge of relevance for the topic.

The causes and consequences of discards are getting increasingly well known and understood, and there is also a general perception that discard are an undesirable consequence; nevertheless, discard persist as solutions are difficult. There are many options to improve fisheries selectivity and better match catching capacity with fishing opportunities in order to reduce discards; nevertheless, it is understood that issues are largely fisheries specific and there are no unique simple solution to reduce discards without jeopardising the profitability of the fisheries in the short term; and incentives to do so have also largely lacked.

A crucial issue with discards is the importance of their monitoring, in order to include reliable quantitative estimates in stock assessment and management advice. Major progresses have been achieved in this domain for the European fisheries, which have contributed to better estimates of management targets. Paradoxally, a major concern with the implementation of a discard ban is the risk of poorer discard estimates and thus stock assessment if control and monitoring actions are unsufficient to ensure full compliance with the ban.

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